LOWER PASSAIC RIVER STUDY AREA
BASELINE ECOLOGICAL RISK ASSESSMENT

FINAL

Prepared for:

USEPA Region 2
as part of the 17-mile LPRSA Remedial Investigation/Feasibility Study

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Acronyms

ACR	acute-to-chronic ratio
A-D	Anderson-Darling
AE	assimilation efficiency
Ah	aryl hydrocarbon
AIC	Akaike's information criterion
AOC	Agreement and Order on Consent
AVS	acid volatile sulfide
AWQC	ambient water quality criteria
BBP	butyl benzyl phthalate
BEHP	bis(2-ethylhexyl) phthalate
BERA	baseline ecological risk assessment
BEST	Biomonitoring of Environmental Status and Trends
BHC	benzene hexachloride
BIC	Bayesian information criterion
BLM	biotic ligand model
BMF	biomagnification factor
BOD	biological oxygen demand
BTAG	Biological Technical Assistance Group
bw	body weight
CARP	Contaminant Assessment and Reduction Project
CBR	critical body residue
CCC	criterion continuous concentration
CDF	cumulative distribution function
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act
CF	conversion factor
cfs	cubic feet per second
CMC	criterion maximum concentration
COC	chemical of concern
COI	chemical of interest
COPEC	chemical of potential ecological concern

CPG	Cooperating Parties Group
CSM	conceptual site model
CSO	combined sewer overflow
CTR	critical tissue residue
DDD	dichlorodiphenyldichloroethane
DDE	dichlorodiphenyldichloroethylene
DDT	dichlorodiphenyltrichloroethane
DF	dietary fraction
DL	detection limit
DO	dissolved oxygen
DOC	dissolved organic carbon
DQO	data quality objective
dw	dry weight
EC10	concentration that causes a non-lethal effect in 10% of an exposed population
EC20	concentration that causes a non-lethal effect in 20% of an exposed population
EC50	concentration that causes a non-lethal effect in 50% of an exposed population
Eco-SSL	ecological soil screening level
ED10	dose that corresponds to a 10% increase in an adverse effect of an exposed population
EF	exceedance factor
EMAP	Environmental Monitoring and Assessment Program
EPC	exposure point concentration
ERA	ecological risk assessment
ERED	Environmental Residue Effects Database
ERM	effects range-median
ESP	ecological sampling program
ETM	estuarine turbidity maximum
FAV	final acute value
FCV	final chronic value
FFS	focused feasibility study

FIR	food ingestion rate
FMR	field metabolic rate
FS	feasibility study
FRV	final residue value
FSP	field sampling plan
FWM	food web model
GC	gas chromatography
GE	gross energy
GIS	geographic information system
HHRA	human health risk assessment
HPAH	high-molecular-weight polycyclic aromatic hydrocarbon
HQ	hazard quotient
HRGC	high-resolution gas chromatography
HRMS	high-resolution mass spectrometry
HSI	habitat suitability index
ID	identification
K-S	Kolmogorov-Smirnov
LC10	concentration that is lethal to 10% of an exposed population
LC20	concentration that is lethal to 20% of an exposed population
LC25	concentration that is lethal to 25% of an exposed population
LC50	concentration that is lethal to 50% of an exposed population
LCL	lower confidence limit
LD50	dose that is lethal to 50% of an exposed population
LOAEL	lowest-observed-adverse-effect level
LOE	line of evidence
LOEC	lowest-observed-effect concentration
LOEL	lowest-observed-effect level
LPAH	low-molecular-weight polycyclic aromatic hydrocarbon
LPR	Lower Passaic River
LPRSA	Lower Passaic River Study Area
LRC	low-resolution coring
MATC	maximum allowable toxicant concentration

mERMq	mean effects range-median quotient
ME	metabolizable energy
MLLW	mean lower low water
mPECq	mean probable effects concentration quotient
MRG	metal-rich granule
MS	mass spectrometry
MTLP	metallothionein-like protein
NCA	National Coastal Assessment
NCCA	National Coastal Condition Assessment
NFF	non-small forage fish
NJDEP	New Jersey Department of Environmental Protection
NJDFW	New Jersey Division of Fish and Wildlife
NJDOT	New Jersey Department of Transportation
NOAA	National Oceanic and Atmospheric Administration
NOAEL	no-observed-adverse-effect level
NOEC	no-observed-effect concentration
NOEL	no-observed-effect level
NY/NJ	New York/New Jersey
OC	organic carbon
OU	operable unit
PAH	polycyclic aromatic hydrocarbon
PAR	pathways analysis report
PCB	polychlorinated biphenyl
PCDD	polychlorinated dibenzo-*p*-dioxin
PCDF	polychlorinated dibenzofuran
PEC	probable effects concentration
PFD	problem formulation document
POC	particulate organic carbon
ppb	parts per billion
ppm	parts per million
ppth	parts per thousand
PRG	preliminary remediation goal

QA	quality assurance
QAPP	quality assurance project plan
QC	quality control
RAL	remedial action level
REMAP	Regional Environmental Monitoring and Assessment Program
RI	remedial investigation
RL	reporting limit
RM	river mile
SAV	submerged aquatic vegetation
SEM	simultaneously extracted metals
SFF	small forage fish
SI	sediment ingestion
SIM	selective ion monitoring
SIR	sediment ingestion rate
SLERA	screening-level ecological risk assessment
SMAV	species mean acute value
SMCV	species mean chronic value
SPI	sediment profile imaging
SQT	sediment quality triad
SSD	species sensitivity distribution
SSO	sanitary sewer overflow
SUF	site use factor
SVOC	semivolatile organic compound
SWO	stormwater outflow
T20	20% probability of observing toxicity
T50	50% probability of observing toxicity
TBT	tributyltin
TCDD	tetrachlorodibenzo-*p*-dioxin
TDS	total dissolved solids
TEF	toxic equivalency factor
TEQ	toxic equivalent
TMDL	total maximum daily load

TOC	total organic carbon
total DDx	sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT and 4,4′-DDT)
TPH	total petroleum hydrocarbons
TRV	toxicity reference value
TSS	total suspended solids
TSV	toxicity screening value
UCL	upper confidence limit on the mean
USACE	US Army Corps of Engineers
USEPA	US Environmental Protection Agency
USFWS	US Fish and Wildlife Service
VOC	volatile organic compound
Windward	Windward Environmental LLC
WIR	water ingestion rate
WOE	weight of evidence
WQC	water quality criteria
ww	wet weight

Executive Summary

This Final Baseline Ecological Risk Assessment for the Lower Passaic River Study Area, hereinafter referred to as the baseline ecological risk assessment (BERA), has been prepared as part of the Lower Passaic River Study Area (LPRSA) remedial investigation/feasibility study (RI/FS) for the 17.4-mi stretch of the Passaic River between Dundee Dam and Newark Bay. This BERA presents the results of the ecological risk assessment (ERA) prepared under US Environmental Protection Agency Region 2 (USEPA) oversight and direction, and was conducted in accordance with Section IX.37.d. of the May 2007 Administrative Settlement Agreement and Order on Consent (AOC) (USEPA 2007a). This final BERA has been amended to address comments, responses, and directives received from USEPA on January 2, 2019 (CDM 2019) and March 5, 2019 (USEPA 2019), and via additional communications between the Cooperating Parties Group (CPG) and USEPA from January through June 2019. .

ES.1 Conclusions

A multi-step process was used to identify preliminary chemicals of concern (COCs) that included the screening-level ecological risk assessment (SLERA) evaluations, which used conservative threshold values and maximum concentrations to identify a preliminary list of chemicals of potential ecological concern (COPECs) from chemicals of interest (COIs). Once COPECs had been identified, they were further evaluated using upper confidence limits and site-specific exposure assumptions. A range of threshold values were also used to assess potential risk, and a discussion of the uncertainty was presented. All COPECs with a hazard quotient (HQ) ≥ 1.0 based on a range of effect-level toxicity reference values (TRVs[1]) were identified as preliminary COCs.[2] The ERA of benthic invertebrates followed an approach to that of the surface water and tissue LOEs; however, the assessment of risk to community structure and

[1] Preliminary COCs were identified as those COPECs with HQs ≥ 1.0 based on any line of evidence (LOE) and effect-level concentration (i.e., HQ ≥ 1.0 based on a range of lowest-observed-adverse-effect levels [LOAELs] for tissue and diet LOEs, HQ ≥ 1.0 based on acute or chronic surface water TRVs; HQ ≥ 1.0 based on plant-specific sediment TRVs.)

[2] The New Jersey Department of Environmental Protection (NJDEP) acknowledges that the BERA for the 17-mile LPRSA RI/FS identifies unacceptable risk, and a remedial action to address the unacceptable risk is necessary. However, it is NJDEP's position that a single TRV set (no-observable-adverse-effect level [NOAEL] and LOAEL) that evaluates the more sensitive species and endpoints to characterize risk to invertebrates, fish, birds and wildlife should be selected in a BERA, not two sets of TRVs, as presented in this document. NJDEP's *Ecological Evaluation Technical Guidance*, published in August 2018 (NJDEP 2018), does not advocate the use of more than one set of TRVs for individual contaminant-receptor pairs. It is NJDEP's position that, for the LPRSA, use of one conservative TRV set derived for sensitive receptors and sensitive endpoints most clearly demonstrates the degree of risk for individual contaminant-receptor pairs and ensures protection of threatened, endangered, and species of special concern.

function was based on a sediment quality triad (SQT) analysis of sediment chemical concentration, sediment toxicity test, and benthic invertebrate community data. Preliminary COCs were not derived using the SQT analysis.

ES.1.1 Preliminary COCs

Preliminary COCs were identified using a range of TRVs (Table ES-1). HQs across receptors and LOEs for preliminary COCs are presented in Table ES-2.

Table ES-1. Summary of ecological COPECs and preliminary COCs

Selected Receptor Group and Species Evaluated	LOE	COPECs[a]	Preliminary COCs Using a Range of TRVs[b]
Protection and maintenance (i.e., survival, growth, and reproduction) of the benthic invertebrate community, both as an environmental resource in itself and as one that serves as a forage base for fish and wildlife populations			
Benthic invertebrate community	SQT (benthic community metrics; toxicity test data; surface sediment chemistry)	not identified using the SQT analysis	No preliminary COCs were identified using the SQT analysis; however, SQT sampling locations were identified as follows: • No, low, or likely low impacts (indicative of insignificant benthic invertebrate risk) relative to urban reference conditions were observed at ~37% of the 97 SQT locations. Medium, likely, or high impacts were observed at 63% of the 97 SQT locations. • Likely or high impacts were observed at ~31% of the 97 SQT locations. • At ~32% of the SQT locations, medium impacts were observed, suggesting moderate risk (an uncertain result due to confounding factors).
Benthic invertebrates (including benthic invertebrate community, macroinvertebrates, and mollusks)	surface water	cadmium, chromium, copper, lead, mercury, selenium silver, zinc, TBT (estuarine), anthracene, benzo(a)anthracene, benzo(a)pyrene, fluoranthene, pyrene, BEHP, BBP, total PCBs, 2,3,7,8-TCDD, 4,4′-DDE, 4,4′-DDT, dieldrin (estuarine), hexachlorobenzene, total chlordane, total DDx, cyanide	2,3,7,8-TCDD, copper, cyanide
Protection and maintenance (i.e., survival, growth, and reproduction) of healthy populations of benthic invertebrates (worms, blue crab and crayfish, and bivalve mussels) that serve as a forage base for fish and wildlife populations and as a base for sports fisheries			
Benthic invertebrates (worms, blue crab, and caged mussels)	tissue	arsenic, cadmium, chromium, cobalt, copper, lead, mercury, methylmercury, nickel, selenium, silver, vanadium, zinc, total LPAHs, total HPAHs, total PCBs, PCB TEQ-fish, 2,3,7,8-TCDD, PCDD/PCDF TEQ - fish, total TEQ - fish, dieldrin, heptachlor expoxide, total DDx	arsenic,[c] chromium,[c] copper,[c] lead,[c] methylmercury/mercury, nickel,[c] selenium, silver,[c] HPAHs, total PCBs, 2,3,7,8-TCDD, PCDD/PCDF TEQ - fish, total TEQ - fish, total DDx

Table ES-1. Summary of ecological COPECs and preliminary COCs

Selected Receptor Group and Species Evaluated	LOE	COPECs[a]	Preliminary COCs Using a Range of TRVs[b]
Protection and maintenance (i.e., survival, growth, and reproduction) of omnivorous, invertivorous, and piscivorous fish populations that serve as a forage base for fish and wildlife populations and as a base for sports fisheries			
Fish populations (mummichog/other forage fish, common carp, white perch, channel catfish, white catfish, brown bullhead, American eel, largemouth bass, smallmouth bass, and northern pike)	tissue	arsenic, cadmium, chromium, copper, lead, methylmercury, selenium, silver, zinc, total HPAHs, total LPAHs, total PCBs, PCB TEQ - fish, 2,3,7,8-TCDD, PCDD/PCDF TEQ - fish, total TEQ - fish, dieldrin, endosulfan I, total DDx	copper,[c] methylmercury/mercury, total PCBs, 2,3,7,8-TCDD, PCB TEQ - fish, PCDD/PCDF TEQ - fish, total TEQ - fish, total PCBs, dieldrin, total DDx
	diet	cadmium, chromium, cobalt, copper, mercury methyl mercury, nickel, selenium, vanadium, zinc, TBT, total PAHs, benzo(a)pyrene, total PCBs, PCT TEQ - fish, PCDD/PCDF TEQ-fish, total TEQ - fish, total DDx	cadmium, mercury, PCB TEQ - fish, PCDD/PCDF TEQ-fish, total TEQ - fish
	surface water	cadmium, chromium, copper, lead, mercury, selenium, silver, zinc, TBT (estuarine), anthracene, benzo(a)anthracene, benzo(a)pyrene, fluoranthene, pyrene, BEHP, BBP, total PCBs, PCB TEQ - fish, 2,3,7,8-TCDD, PCDD/PCDF TEQ - fish, total TEQ - fish, 4,4′-DDE, 4,4′-DDT, dieldrin (estuarine), hexachlorobenzene, total chlordane, total DDx, cyanide	copper and cyanide
	egg tissue (mummichog)	mercury, methylmercury, total PCBs, PCDD/PCDF TEQ - fish, total TEQ - fish	mercury, total PCBs
	mummichog egg count	none identified based on qualitative LOE	none identified based on qualitative LOE
	health assessment	none identified based on qualitative LOE	none identified based on qualitative LOE

Table ES-1. Summary of ecological COPECs and preliminary COCs

Selected Receptor Group and Species Evaluated	LOE	COPECs[a]	Preliminary COCs Using a Range of TRVs[b]
Protection and maintenance (i.e., survival, growth, and reproduction) of herbivorous, omnivorous, sediment-probing, and piscivorous bird populations			
Bird populations (spotted sandpiper, belted kingfisher, and great blue heron)	diet	cadmium, chromium, copper, lead, methylmercury, nickel, selenium, vanadium, zinc, total LPAHs, total HPAHs, total PCBs, PCB TEQ - bird, PCDD/PCDF TEQ - bird, total TEQ - bird, total DDx	copper, lead, methylmercury, total HPAHs, total PCBs, PCB TEQ - bird, PCDD/PCDF TEQ - bird, total TEQ - bird, total DDx
	egg tissue	methylmercury/mercury, total PCBs, PCB TEQ - bird, PCDD/PCDF TEQ - bird, total TEQ - bird, total DDx, dieldrin	total PCBs, PCDD/PCDF TEQ - bird, total TEQ - bird, PCB TEQ - bird, total DDx
Protection and maintenance (i.e., survival, growth, and reproduction) of aquatic mammal populations			
Mammal populations (river otter and mink)	diet	arsenic, cadmium, copper, lead, methylmercury/mercury, nickel, selenium, vanadium, and zinc, total HPAHs, total PCBs, PCB TEQ - mammal, PCDD/PCDF TEQ - mammal, total TEQ - mammal, dieldrin	total PCBs, PCB TEQ - mammal, PCDD/PCDF TEQ - mammal, total TEQ - mammal
Maintenance of the zooplankton community that serves as a food base for juvenile fish			
Zooplankton community	surface water	cadmium, chromium, copper, lead, mercury, selenium, silver, zinc, TBT (estuarine), anthracene, benzo(a)anthracene, benzo(a)pyrene, fluoranthene, pyrene, BEHP, BBP, total PCBs, 2,3,7,8-TCDD, 4,4′-DDE, 4,4′-DDT, dieldrin (estuarine), hexachlorobenzene, total chlordane, total DDx, cyanide	copper and cyanide
Protection and maintenance (i.e., survival, growth, and reproduction) of healthy amphibian and reptile populations			
Amphibians and reptile populations (multiple species represented)	surface water	chromium, copper, lead, mercury, nickel, silver, zinc	none identified

Table ES-1. Summary of ecological COPECs and preliminary COCs

Selected Receptor Group and Species Evaluated	LOE	COPECs[a]	Preliminary COCs Using a Range of TRVs[b]
Maintenance of healthy aquatic plant populations as a food resource and habitat for fish and wildlife populations			
Aquatic plant populations (multiple species represented)	sediment	antimony, arsenic, cadmium, chromium, cobalt, copper, lead, mercury, nickel, selenium, vanadium, zinc, and acenaphthene	chromium, copper, lead, mercury, selenium, vanadium, and zinc
	surface water	cadmium, chromium, copper, lead, mercury (estuarine), zinc, TBT, total PCBs (estuarine), 2,3,7,8-TCDD, 4,4'-DDE, cyanide (estuarine)	copper, zinc, TBT, cyanide

[a] COPECs are those COIs for which the maximum concentration exceeded its TSV in the SLERA. If a TSV was exceeded based on any species in a receptor group, it was retained as a COPEC for all species in that receptor group. COPECs for surface water are for both estuarine (RM 0 to RM 13) and freshwater (RM 4 to RM 17.4) unless noted otherwise.

[b] Preliminary COCs are those COPECs with HQs ≥ 1.0 based on any LOE and effect-level concentration (i.e., HQ ≥ 1.0 based on a ranges of LOAELs for tissue and diet LOEs, HQ ≥ 1.0 based on acute or chronic surface water TRVs; HQ ≥ 1.0 based on plant-specific sediment TRVs).

[c] Preliminary COCs for regulated metals based on the tissue residue LOE were based on EFs rather than HQs.

BBP – butyl benzyl phthalate
BEHP – bis(2-ethylhexyl) phthalate
COC – chemical of concern
COI – chemical of interest
COPEC – chemical of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
EF – exceedance factor
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
LOE – line of evidence
LPAH – low-molecular-weight polycyclic aromatic hydrocarbon
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
PCDD – polychlorinated dibenzo-*p*-dioxin
PCDF –polychlorinated dibenzofuran
RM – river mile
SLERA – screening-level ecological risk assessment
SQT – sediment quality triad
TBT – tributyltin
TCDD – tetrachlorodibenzo-*p*-dioxin
TEQ – toxic equivalent
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT and 4,4′-DDT)
TRV – toxicity reference value
TSV – toxicity screening value

Table ES-2. Summary of HQs and EFs for preliminary COCs

Preliminary COC	Risk Results (Range of LOAEL HQs/EFs[a])					
	Benthic Invertebrates	Fish	Birds	Mammals	Aquatic Plants	Zooplankton
Metals						
Arsenic	tissue: worm (2.2); blue crab (2.2)	*no unacceptable risk*	*no unacceptable risk*	*no unacceptable risk*	*no unacceptable risk*	*no unacceptable risk*
Cadmium	*no unacceptable risk*	diet: mummichog (1.3); common carp (1.2); white perch (1.1); white sucker (1.2); American eel (0.70–1.2)	*no unacceptable risk*	*no unacceptable risk*	*no unacceptable risk*	*no unacceptable risk*
Chromium	tissue: worm (6.0); mussel (3.7)	*no unacceptable risk*	*no unacceptable risk*	*no unacceptable risk*	sediment (160)	*no unacceptable risk*
Copper	surface water (estuarine: 0.14–2.7; freshwater: 0.034–1.0)	surface water (estuarine: 0.14–2.7; freshwater: 0.023-1.0)	diet: spotted sandpiper (0.30-3.6); great blue heron (0.029–1.3)	*no unacceptable risk*	sediment (2.4)	surface water (estuarine: 0.14–2.7; freshwater: 0.023–1.0)
	tissue: blue crab (2.1)	tissue: mummichog (2.1), other forage fish (2.7), white perch (9.3), American eel (1.7)			surface water (estuarine: 1.8)	
Lead	tissue: worm (0.16–2.5)	*no unacceptable risk*	diet: spotted sandpiper (0.20–10); belted kingfisher (0.015–1.1)	*no unacceptable risk*	sediment (2.3)	*no unacceptable risk*
Methylmercury/ mercury	tissue: blue crab: (1.3–1.5)	tissue: white catfish (0.71–1.1); American eel (0.74–1.1); largemouth bass (1.5–2.6); smallmouth bass (0.63–1.1)	diet: great blue heron (0.031–1.6); belted kingfisher (0.13–1.6)	*no unacceptable risk*	sediment (9.7)	*no unacceptable risk*
		diet: mummichog (1.3); common carp (1.1); white perch (1.3); white catfish (1.1); American eel (1.1–1.3)				
		Egg tissue: mummichog (0.11–1.1)				

Table ES-2. Summary of HQs and EFs for preliminary COCs

Preliminary COC	Risk Results (Range of LOAEL HQs/EFs[a])					
	Benthic Invertebrates	Fish	Birds	Mammals	Aquatic Plants	Zooplankton
Nickel	tissue: worm (12); mussel (6.0)	*no unacceptable risk*	*no unacceptable risk*	*no unacceptable risk*	*no unacceptable risk*	*no unacceptable risk*
Selenium	tissue worm (1.1); blue crab (1.5)	*no unacceptable risk*	*no unacceptable risk*	*no unacceptable risk*	sediment (1.8)	*no unacceptable risk*
Silver	tissue: blue crab (1.0)	*no unacceptable risk*	*not evaluated (no toxicity data available)*	*not evaluated (no toxicity data available)*	*no unacceptable risk*	*no unacceptable risk*
Vanadium	*no unacceptable risk*	*no unacceptable risk*	*no unacceptable risk*	*no unacceptable risk*	sediment (14)	*no unacceptable risk*
Zinc	*no unacceptable risk*	*no unacceptable risk*	*no unacceptable risk*	*no unacceptable risk*	sediment (3.1) surface water (estuarine; 21)	*no unacceptable risk*
Organometals						
TBT	*no unacceptable risk*	*no unacceptable risk*	*no unacceptable risk*	*no unacceptable risk*	surface water (estuarine: 1.1; freshwater: 50)	*no unacceptable risk*
PAHs						
HPAHs	tissue: worms (0.090–3.0)	*no unacceptable risk*	diet: spotted sandpiper (1.9–10)	*no unacceptable risk*	*no unacceptable risk*	*no unacceptable risk*

Table ES-2. Summary of HQs and EFs for preliminary COCs

Preliminary COC	Risk Results (Range of LOAEL HQs/EFs[a])					
	Benthic Invertebrates	Fish	Birds	Mammals	Aquatic Plants	Zooplankton
PCBs						
Total PCBs	tissue: worm (0.46–14), blue crab (0.67–21), mussels (0.046–1.4)	tissue: mummichog (0.16–1.1), other forage fish (0.14–1.0), common carp (1.4–9.8), white perch (0.66–4.7), channel catfish (0.45–3.2), brown bullhead (0.37–2.6), white catfish (0.89–6.4), white sucker (0.76–5.5), American eel (0.53–3.8), largemouth bass (2.1–15), northern pike (0.53–3.8), smallmouth bass (0.37–2.6) diet: northern pike (1.3) egg tissue: mummichog (2.2–18)	diet: spotted sandpiper (0.047–1.2), great blue heron (0.031–1.1) egg tissue: great blue heron (0.078–284); belted kingfisher (0.22–76)	diet: mink (0.94–3.1); river otter (2.6–3.7)	*no unacceptable risk*	*no unacceptable risk*
PCB TEQ	*no unacceptable risk*	tissue: common carp (0.037–2.4), white perch (0.018–1.2), channel catfish (0.015–1.0); white catfish (0.029–1.9), white sucker (0.027–1.8), largemouth bass (0.14–9.4), northern pike (0.019-1.3) diet: white perch (1.0); American eel (0.95–1.8); largemouth bass (1.6); smallmouth bass (1.5); northern pike (2.1)	diet: spotted sandpiper (0.073–3.9); great blue heron (0.030–1.6), belted kingfisher (0.10–1.5) egg tissue: great blue heron (0.56–36); belted kingfisher (0.46–12)	diet: mink (0.12–1.1); river otter (0.31–1.4)	*not evaluated*	*not evaluated*

Table ES-2. Summary of HQs and EFs for preliminary COCs

Preliminary COC	Risk Results (Range of LOAEL HQs/EFs[a])					
	Benthic Invertebrates	Fish	Birds	Mammals	Aquatic Plants	Zooplankton
PCDD/PCDFs						
2,3,7,8-TCDD	surface water (estuarine: 0.0028–4.3) tissue: worm (0.013–29); blue crab (0.019–44); mussel (0.00073–1.7)	tissue: mummichog (0.41–27), other forage fish (0.38–26), common carp (5.1–340), white perch (1.6–110), channel catfish: (0.80–53), brown bullhead (1.3–83), white catfish (1.8–120), white sucker (1.1–72), American eel (0.19–13), largemouth bass (1.5–100), northern pike (0.79–53), smallmouth bass (0.63–42)	*not evaluated*	*not evaluated*	*no unacceptable risk*	*no unacceptable risk*
PCDD/PCDF TEQ	tissue: worm (0.013–29), blue crab (0.021–48), mussel (0.00077–1.8)	tissue: mummichog (0.43–28), other forage fish (0.41–27), common carp (5.2–340), white perch (1.7–110) channel catfish (0.83–56), brown bullhead (1.3–89), white catfish (1.8–120), white sucker (1.1–72), American eel (0.20–13), largemouth bass (1.5–100), northern pike (0.83–56) smallmouth bass (0.63–42), diet**:** mummichog (200), common carp (200), white perch (170), channel catfish (190) white catfish (160), white sucker (190), American eel (180-190) largemouth bass (150) smallmouth bass (140), northern pike (200)	diet: spotted sandpiper (0.014–21), great blue heron (0.020–1.9), belted kingfisher (0.090–1.9) egg tissue: great blue heron (0.42–37), belted kingfisher (0.38–14)	diet: mink: (0.79–8.7), river otter (1.8–10)	*not evaluated*	*not evaluated*

Table ES-2. Summary of HQs and EFs for preliminary COCs

Preliminary COC	Risk Results (Range of LOAEL HQs/EFs[a])					
	Benthic Invertebrates	Fish	Birds	Mammals	Aquatic Plants	Zooplankton
Total TEQ	tissue: worm (0.013–30); blue crab (0.021–48); mussel (0.00077–1.8)	tissue: mummichog: (0.43–28), other forage fish: (0.41–27), common carp: (5.2–340), white perch: (1.7–110), channel catfish: (0.83–56), brown bullhead: (1.3–89), white catfish: (1.9–130), white sucker: (1.1–72), American eel: (0.21–14), largemouth bass: (1.5–100), northern pike: (0.92–61), smallmouth bass: (0.68–46)	diet: spotted sandpiper (0.089–25), great blue heron (0.044–3.5), belted kingfisher (0.18–3.1)	diet: mink (1.0–9.9), river otter (2.4–12)	*not evaluated*	*not evaluated*
		diet: mummichog (210); common carp (200); white perch (170); channel catfish (190); white catfish (160); white sucker (190); American eel (190-200); largemouth bass (150); smallmouth bass (140); northern pike (200)	egg tissue: great blue heron (1.0–74), belted kingfisher (0.85–23)			
Pesticides						
Total DDx	tissue: worm: (0.12–1.6), blue crab (0.52–6.8)	tissue: common carp (1.3–1.7)	diet: spotted sandpiper (0.018–1.4); great blue heron (0.020–2.4); belted kingfisher (0.066–1.8)	*no unacceptable risk*	*no unacceptable risk*	*no unacceptable risk*
			egg tissue: great blue heron (0.14–18); belted kingfisher (0.37–4.6)			
Dieldrin	*no unacceptable risk*	tissue: common carp (0.28–1.4), channel catfish (0.24–1.2), American eel (0.27–1.4), largemouth bass (0.20–1.0), northern pike (0.22–1.1)	*no unacceptable risk*	*no unacceptable risk*	*no unacceptable risk*	*no unacceptable risk*

Table ES-2. Summary of HQs and EFs for preliminary COCs

Preliminary COC	Risk Results (Range of LOAEL HQs/EFs[a])					
	Benthic Invertebrates	Fish	Birds	Mammals	Aquatic Plants	Zooplankton
Other						
Cyanide	surface water (estuarine: 1.3–4.1; freshwater: 0.23–1.0)	surface water (estuarine: 1.6–5.3)	*not evaluated*	*not evaluated*	surface water (estuarine: 2.0)	surface water (estuarine: 1.6–5.3)

Note – Preliminary COCs are identified as those COPECs with HQs ≥ 1.0 based on any LOE and effect-level concentration (i.e., HQ ≥ 1.0 based on a range of LOAELs for tissue and diet LOEs, HQ ≥ 1.0 based on acute or chronic surface water TRVs; HQ ≥ 1.0 based on plant-specific sediment TRVs).

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

COC – chemical of concern
COPEC – chemical of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
EF – exceedance factor
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
LOE – line of evidence
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
PCDD – polychlorinated dibenzo-*p*-dioxin
PCDF –polychlorinated dibenzofuran
TBT – tributyltin
TCDD – tetrachlorodibenzo-*p*-dioxin
TEQ – toxic equivalent
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT and 4,4′-DDT)
TRV – toxicity reference value

ES.1.2 Benthic invertebrate community risk conclusions

The following risks were identified for the benthic invertebrate community:

- **Benthic invertebrate community –** Among the 97 SQT locations, the weight of evidence (WOE) analysis resulted in 28 locations with low impacts, 18 locations with high impacts, and 51 locations with relatively uncertain medium impacts (Table ES-3, Figure ES-1). A site-specific post-hoc analysis was conducted to further evaluate stations initially categorized as medium impact due to high uncertainty associated with the individual LOEs. Based on the post-hoc analysis, several medium-impact locations were reclassified as likely low impact if the LPRSA SQT location was associated with low sediment chemistry[3] or negligible toxicity relative to the urban reference condition. Where habitat appeared to be suitable for invertebrates but toxicity and chemical contamination were evident, the WOE conclusion was altered from medium impact to likely impacted. Of the 97 SQT locations, 30 (31%) were placed in the likely and high-impact WOE categories, which indicate locations within the LPRSA where the benthic community is impacted relative to urban reference conditions. Thirty-six SQT locations (37%) were placed in the no, low, or likely low impact categories, indicating minimal impacts in the LPRSA community relative to urban reference conditions. Results remained relatively uncertain (i.e., medium impact) at 31 (32%) of the 97 SQT locations, although, for the sake of risk characterization, impacts at those SQT locations were considered to be moderate. Moderate impacts may be caused by exposures to chemical contaminants and exacerbated by habitat conditions. Thus, impacts (medium, likely, or high) were observed at 63% of SQT locations.

Table ES-3. Summary of WOE results by salinity zone compared with reference data representing urban habitats

Benthic Salinity Zone	N	No Impact		Low Impact		Medium Impact[a]						High Impact	
						Likely Low Impact		Medium Impact (Unchanged)		Likely Impacted			
		n	%	n	%	N	%	n	%	n	%	n	%
Upper estuarine (RM 0 to RM 4)	25	0	0%	13	52%	0	0%	5	20%	5	20%	2	8%
Fluvial estuarine (RM 4 to RM 13)	54	0	0%	14	26%	2	4%	26	48%	7	13%	5	9%
Tidal freshwater (RM 13 to RM 17.4)	18[b]	0	0%	1	6%	6	33%	0	0%	0	0%	11	61%

[3] Low sediment chemistry was defined as a mean probable effects concentration quotient (mPECq) < 0.5 at tidal freshwater locations or a mean effects range-median quotient (mERMq) < 0.361 at upper or fluvial estuarine locations.

Benthic Salinity Zone	N	No Impact		Low Impact		Medium Impact[a]						High Impact	
						Likely Low Impact		Medium Impact (Unchanged)		Likely Impacted			
		n	%	n	%	N	%	n	%	n	%	n	%
Site wide	97	0	0%	28	29%	8	8%	31	32%	12	12%	18	19%

[a] Medium-impact locations were re-evaluated using a post-hoc analysis; based on several factors, SQT locations were recategorized as likely low impact, likely impacted, or unchanged (medium impact) (Appendix B, Table B10)

[b] Of the 98 locations sampled in fall 2009 for sediment chemistry analyses and toxicity testing, benthic invertebrate communities were only analyzed at 97 locations. The WOE analysis was conducted at only the 97 locations for which all three types of SQT data were collected.

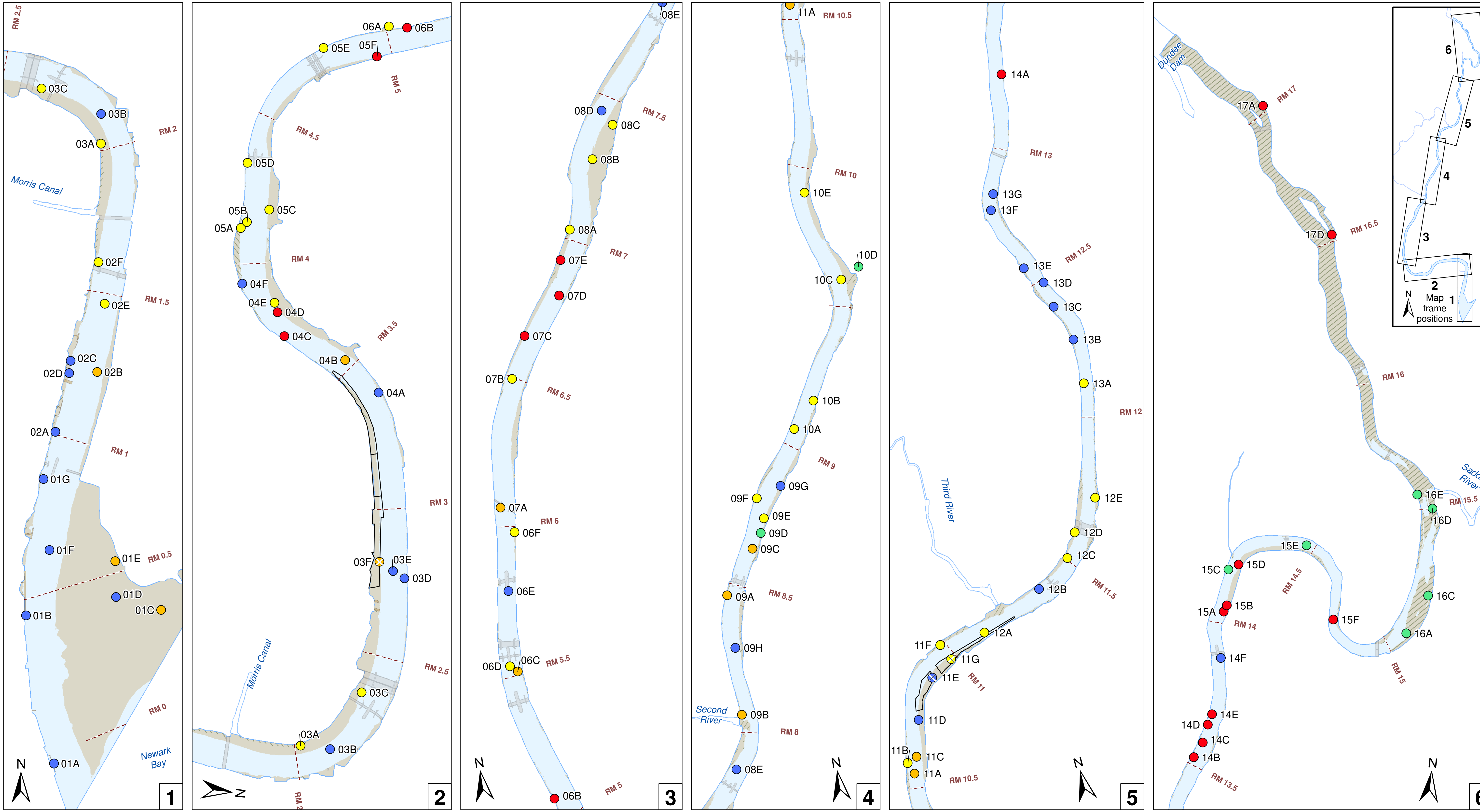
n – sample size (by conclusion)

N – sample size (by benthic salinity zone)

RM – river mile

SQT – sediment quality triad

WOE – weight of evidence



Windward environmental LLC

0 0.25 0.5 Kilometers

0 0.25 0.5 Miles

SQT weight of evidence
- High impact
- Likely impacted
- Medium impact
- Likely low impact
- Low impact

× Remediated location[a]
Dredge zone
River mile
LPRSA

Bridge
Abutment
Dock
Mudflat
Gravel with fines

[a]One sample was collected in the Lister Ave. dredge area at RM 2.8 and two were collected in the RM 10.9 dredge area.

Mudflat areas for ecological receptors are those areas where the river bottom slope is ≤ 6° and the depth is ≥ -2 ft MLLW, based on the 2007 CPG bathymetric survey.

Figure ES-1. Conclusion of weight of evidence analysis of SQT data from the LPRSA

Lower Passaic River Study Area
Baseline Ecological Risk Assessment

FINAL

Prepared by mikey 5/30/2019, W:\Projects\06-58-01 Passaic RI\Data\GIS\Maps_and_Analysis\BERA\Revised BERA 2016\6378_Benthic SQT urban WOE in the LPRSA_LSM_20160721.mxd

ES.1.3 Ecological risk drivers

The following preliminary COCs are recommended as risk drivers for further evaluation in the FS:

- 2,3,7,8-tetrachlorodibenzo-*p*-dioxin (TCDD)
- Polychlorinated dibenzo-*p*-dioxin/polychlorinated dibenzofuran (PCDD/PCDF) toxic equivalent (TEQ) (fish, bird, and mammal)
- Total TEQ (fish, bird, and mammal)
- Total polychlorinated biphenyls (PCBs)
- PCB TEQ (fish, bird, and mammal)
- Total DDx (sum of all six dichlorodiphenyltrichloroethane [DDT] isomers [2,4′-dichlorodiphenyldichloroethane (DDD), 4,4′-DDD, 2,4′-dichlorodiphenyldichloroethylene (DDE), 4,4′-DDE, 2,4′-DDT and 4,4′-DDT])

The above-listed risk drivers are based on effect-level HQs exceeding 1.0 for various ecological receptor groups and LOEs. Some LOEs are more certain than others and should be evaluated prior to making any management decisions. Table ES-4 presents a summary of the risk drivers and considerations for risk management decisions regarding the assumptions used to derive HQs.

Table ES-4. Considerations for risk management on ecological risk drivers

Risk Driver and LOE	LOAEL HQ range[a,b] Based on TRV-A[c]	LOAEL HQ range[a,b] Based on TRV-B[d]	Risk Management Considerations
Benthic invertebrate community	No risk drivers were identified using the SQT analysis; however, SQT locations with impacts were identified as follows: • No, low, or likely low impacts (indicative of insignificant benthic invertebrate risk) relative to urban reference conditions were observed at ~37% of the 97 SQT locations. Impacts were observed at 63% of SQT locations. • Likely or high impacts were observed at ~31% of the 97 SQT locations. • At ~32% of the SQT locations, risk was unclear (medium impact). Medium impacts suggest moderate chemical risk.		• The reference area chemistry and toxicity screens were conservative, which resulted in a dataset that may not represent realistic reference conditions. The quantitative analysis of uncertainty (Appendix P) provides an alternative screening process. • The sediment chemistry LOE was conservative and potentially unreliable for predicting actual effects in the LPRSA. The quantitative analysis of uncertainty (Appendix P) provides an alternative chemistry LOE. The multivariate analysis of SQT data (Appendix P) indicates that sediment chemical factors are related to benthic community impacts and exacerbated by habitat variables. • The comparison of LPRSA SQT data to non-urban reference data was less relevant than the comparison of LPRSA data to urban reference data. Effects in the LPRSA associated with its urban setting were not addressed by the comparison of LPRSA SQT data to non-urban reference data. • Medium-impact conclusions of the SQT WOE analysis were uncertain because of disagreement between or within LOEs. The quantitative analysis of uncertainty (Appendix P) attempts to address these uncertainties. Moderate effects are possible at medium-impact stations. • Impacts at freshwater LPRSA SQT locations LPRT17A and LPRT17D were potentially influenced (at least in part) by differences between habitat conditions immediately below Dundee Dam and those in the area above Dundee Dam. The area above the dam has finer sediments than the area just below, which is predominately composed of coarse sand and cobble. In general, such sediments are not expected to have elevated sediment contamination.
Total PCBs			
Benthic invertebrate tissue	0.046–0.67 (mussels, worm, and blue crab)	**1.4–21** (mussels, worm, and blue crab)	• TRV-A based on an SSD value less than lowest measured LOAEL; TRV-A results in HQs < 1.0 • TRV-B based on whole-body tissue concentrations interpolated from measured egg tissue concentrations

Table ES-4. Considerations for risk management on ecological risk drivers

Risk Driver and LOE	LOAEL HQ range[a,b]		Risk Management Considerations
	Based on TRV-A[c]	Based on TRV-B[d]	
Fish tissue	0.14–**2.1** (all LPRSA fish species evaluated)	**1.0**–**15** (all LPRSA fish species evaluated)	• TRV-A based on an SSD value less than lowest measured LOAEL • TRV-B based on changes in smolt seawater preference in Atlantic salmon • EPC for largemouth bass based on maximum tissue concentration due to sample size
Fish diet	**1.3** (northern pike)	ne	• LOAEL based on fecundity (number of eggs per female), but no significant reduction on egg weight or hatching rate was reported.
Fish egg	**2.2–3.6** (mummichog)	**11–18** (mummichog)	• TRV-A and TRV-B based on same literature source; TRV-A based on observed adverse effect on reproduction (reduced hatchability), and TRV-B based on reduced fecundity, but no effect on egg weight or hatchability • Mummichog egg concentration modeled using literature-based CFs and LPRSA mummichog-specific lipid content
Bird diet	0.031–0.70 (spotted sandpiper, great blue heron)	0.11–**1.2** (spotted sandpiper, great blue heron)	• TRV-A based on non-chicken reproduction; TRV-A results in HQs < 1.0 • TRV-B based on interpolated value from chicken hatchability data • Low HQs and HQs < 1.0 conflicted with bird egg LOE, which relied on literature-based BMFs to estimate egg tissue concentrations
Bird egg	0.078–**1.0** (great blue heron and belted kingfisher)	**1.0**–**284** (great blue heron and belted kingfisher)	• TRV-A based on non-chicken reproduction and limited dataset (two studies) • TRV-B based on interpolated value from chicken hatchability data • Uncertainty associated with use of literature-based BMFs to predict bird egg concentrations; species-specific BMF for heron and kingfisher based heron and kingfisher data, respectively, for comparison to TRV-A, and range of BMFs evaluated for comparison to TRV-B
Mammal diet	0.94–**3.2** (mink and river otter)	**1.1**–**3.7** (mink and river otter)	• TRV-A and TRV-B based on same literature source with slightly different ingestion rates and body weight assumptions used to derive TRV • HQ based on range of dietary scenarios and 2 exposure areas; HQs slightly lower based on site-wide exposure area (vs. > RM 10); mink HQs lower based on consumption of aquatic prey and assuming zero exposure from terrestrial prey

Table ES-4. Considerations for risk management on ecological risk drivers

Risk Driver and LOE	LOAEL HQ range[a,b]		Risk Management Considerations
	Based on TRV-A[c]	Based on TRV-B[d]	
PCB TEQ			
Fish tissue	0.014–0.037 (0.010-0.74) (common carp, white perch, channel catfish, white catfish, white sucker, largemouth bass, northern pike)	**1.0**–**9.4** (common carp, white perch, channel catfish, white catfish, white sucker, largemouth bass, northern pike)	• TRV-A based on SSD within range of measured LOAELs evaluated; TRV-A results in HQs < 1.0 • Alternative TRV-A based on SSD with relatively poor visual and statistical fit to the empirical data and likely over-predicts risk; alternative SSD less than lowest measured LOAEL • TRV-B based on interpolated larvae concentration from egg tissue
Fish diet	**1.5**–**2.1** (American eel - large; largemouth bass; smallmouth bass; northern pike)	Ne	• LOAEL TRV 2 orders of magnitude less than LOAELs reported for 2 other species
Bird diet	0.030–0.78 (all bird species evaluated)	0.13–**3.9** (all bird species evaluated)	• TRV-A and TRV-B based on same literature source based on weekly injection of pheasants; TRV-A results in HQs < 1.0 • TRV-B extrapolated from study using interspecies extrapolation factor of 5 • High variability of bird TEFs and differences among species sensitivities to dioxin-like compounds • Low HQs and HQs < 1.0 conflicted with bird egg LOE, which relied on literature-based BMFs to estimate egg tissue concentrations
Bird egg	0.46–**7.2** (great blue heron and belted kingfisher)	0.57–**36** (great blue heron and belted kingfisher)	• TRV-A based on SSD with no chicken reproduction data (SSD not expected to have changed significantly with inclusion of chicken data) • TRV-B based on SSD inclusive of chicken reproduction data • TEQ sensitivities varied with Ah receptor; chickens in high-sensitivity group and great blue heron in low-sensitivity group • Uncertainty associated with use of literature-based BMFs to predict bird egg concentrations; species-specific BMF for heron and kingfisher based on heron and kingfisher data, respectively, for comparison to TRV-A, and range of BMFs evaluated for comparison to TRV-B

Table ES-4. Considerations for risk management on ecological risk drivers

Risk Driver and LOE	LOAEL HQ range[a,b] Based on TRV-A[c]	LOAEL HQ range[a,b] Based on TRV-B[d]	Risk Management Considerations
Mammal diet	0.12–0.34 (mink and river otter)	0.49–**1.4** (mink and river otter)	• TRV-A based on mink fed laboratory-prepared diet and TRV-B based on mink fed field-collected carp; TRV-A results in HQs < 1.0 • HQ based on range of dietary scenarios and 2 exposure areas; HQs slightly lower based on site-wide exposure area (vs. > RM 10); mink HQs lower based on consumption of aquatic prey and assuming zero exposure from terrestrial prey
PCDD/PCDF and total TEQ			
Benthic invertebrate tissue	0.00077–0.021 (PCDD/PCDF TEQ; worm, blue crab and mussels) 0.00077–0.021 (total TEQ; worm, blue crab and mussels)	**1.8**–**48** (PCDD/PCDF TEQ; worm, blue crab and mussels) **1.8**–**48** (total TEQ; worm, blue crab and mussels)	• TRV-A based on injected (not measured) concentration in crayfish; TRV-A results in HQs < 1.0 • TRV-B based on uncontrolled field data and limited sample size (n = 1 tissue composite); LOAEL based on relative reduction at Arthur Kill site compared to Sandy Hook site • Evaluation as TEQ (based on fish TEFs) questionable for invertebrates because there was limited evidence for ligand activation of the Ah (dioxin) cellular receptor in these organisms (i.e., they were not susceptible to the dioxin-like effects reported for vertebrates) (Van den Berg et al. 1998).
Fish tissue	0.20–**5.2** (**1.0**–**27**) (PCDD/PCDF TEQ-fish; all fish species evaluated) 0.21–**5.2** (**1.1**–**27**) (total TEQ-fish; all fish species evaluated)	**13**–**340** (PCDD/PCDF TEQ-fish; all fish species evaluated) **14**–**340** (total TEQ-fish; all fish species evaluated)	• TRV-A based on SSD within range of measured LOAELs evaluated • Alternative TRV-A based on SSD with relatively poor visual and statistical fit to the empirical data and likely over-predicts risk; alternative SSD less than lowest measured LOAEL • TRV-B based on interpolated larvae concentration from egg tissue
Fish diet	**140**–**200** (PCDD/PCDF TEQ-fish; all fish species evaluated) **140**–**210** (total TEQ-fish; all fish species evaluated)	ne	• LOAEL TRV 2 orders of magnitude less than LOAELs reported for 2 other species

Table ES-4. Considerations for risk management on ecological risk drivers

Risk Driver and LOE	LOAEL HQ range[a,b]		Risk Management Considerations
	Based on TRV-A[c]	Based on TRV-B[d]	
Bird diet	0.014–**4.2** (PCDD/PCDF TEQ - bird; all bird species evaluated) 0.044–**5.0** (total TEQ - bird; all bird species evaluated)	0.071–**21** (PCDD/PCDF TEQ - bird; all bird species evaluated) 0.22–**25** (total TEQ - bird; all bird species evaluated)	• TRV-A and TRV-B based on same literature source based on weekly injection of pheasants • TRV-B extrapolated from study using interspecies extrapolation factor of 5 • High variability of bird TEFs and differences among species sensitivities to dioxin-like compounds
Bird egg	0.38–**7.5** (PCDD/PCDF TEQ - bird; great blue heron and belted kingfisher) **1.0**–**15** (total TEQ - bird; great blue heron and belted kingfisher)	0.43–**37** (PCDD/PCDF TEQ - bird; great blue heron and belted kingfisher) **1.0**–**74** (total TEQ - bird; great blue heron and belted kingfisher)	• TRV-A based on SSD with no chicken reproduction data (SSD not expected to have changed significantly with inclusion of chicken data) • TRV-B based on SSD inclusive of chicken reproduction data • TEQ sensitivities varied with Ah receptor; chickens in high-sensitivity group and great blue heron in low-sensitivity group • Species-specific BMF for heron and kingfisher based heron and kingfisher data, respectively
Mammal diet	0.79–**2.6** (PCDD/PCDF TEQ-mammal; mink and river otter) **1.0**–**2.9** (total TEQ-mammal; mink and river otter)	**3.2–10** (PCDD/PCDF TEQ-mammal; mink and river otter) **4.1**–**12** (total TEQ-mammal; mink and river otter)	• TRV-A based on mink fed laboratory-prepared diet and TRV-B based on mink fed field-collected carp • HQ based on range of dietary scenarios and 2 exposure areas; HQs slightly lower based on site-wide exposure area (vs. > RM 10); mink HQs lower based on consumption of aquatic prey and assuming zero exposure from terrestrial prey
Total DDx			
Benthic invertebrate tissue	0.15–0.62 (**1.6**–**6.8**) (worm and blue crab)	0.12–0.52 (worm and blue crab)	• TRV-A and alternative TRV-A based on SSD less than lowest measured LOAEL • Alternative TRV-A based on relatively poor visual and statistical fit to the empirical data and likely overestimates toxicity
Fish tissue	**1.3** (common carp)	**1.7** (common carp)	• TRV-A based on SSD less than lowest measured LOAEL evaluated • TRV-B based on SSD within range of measured LOAELs evaluated (which included TRVs based on field-collected organisms) • HQs < 1.0 for other 11 of 12 fish species evaluated

Table ES-4. Considerations for risk management on ecological risk drivers

Risk Driver and LOE	LOAEL HQ range[a,b]		Risk Management Considerations
	Based on TRV-A[c]	Based on TRV-B[d]	
Bird diet	0.018–0.26 (all bird species evaluated)	0.16–**2.4** (all bird species evaluated)	• TRV-A based on SSD within range of measured LOAELs evaluated; TRV-A results in HQs < 1.0 • TRV-B based on field study of eggshell thinning in pelicans • Low HQs and HQs < 1.0 conflicted with bird egg LOE, which relied on literature-based BMFs to estimate egg tissue concentrations
Bird egg	0.14–**1.8** (great blue heron and belted kingfisher)	0.19–**18** (great blue heron and belted kingfisher)	• TRV-A based on SSD not inclusive of chicken reproduction data • TRV-B based on SSD inclusive of chicken reproduction data • Uncertainty associated with use of literature-based BMFs to predict bird egg concentrations; species-specific BMF for heron based heron data, and species-specific BMF for kingfisher based on geomean of 5 species for comparison to TRV-A and range of BMFs evaluated for comparison to TRV-B

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

b HQs were based on LOAEL TRVs.

c TRVs were derived from the primary literature review.

d TRVs based on USEPA's revised draft of the LPR restoration project FFS (Louis Berger et al. 2014) or first draft of the LPR restoration project FFS (Malcolm Pirnie 2007b).

Ah – aryl hydrocarbon
BMF – biomagnification factor
CF – conversion factor
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
EPC – exposure point concentration
FFS – focused feasibility study
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
LOE – line of evidence
LPR – Lower Passaic River
LPRSA – Lower Passaic River Study Area
ne – not evaluated
PCB – polychlorinated biphenyl
PCDD – polychlorinated dibenzo-*p*-dioxin
PCDF –polychlorinated dibenzofuran
RM – river mile
SQT – sediment quality triad
SSD – species sensitivity distribution
TEF – toxic equivalency factor
TEQ – toxic equivalent
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT and 4,4′-DDT)
TRV – toxicity reference value
USEPA – US Environmental Protection Agency
WOE – weight of evidence

A number of preliminary COCs were not recommended as risk drivers to be carried forward to inform major risk management decisions. Preliminary COCs that were not retained as risk drivers were excluded primarily for two reasons:

- Background concentrations indicated that risks in the LPRSA would not be different or would be less than those in background (upstream or regional) areas.
- The LOE for which a LOAEL HQ was ≥ 1.0 could not reliably predict risks to a level appropriate for costly remedial decisions. This included the tissue residue LOE for metals[4] and the sediment LOE for aquatic plants.[5]

Eleven metals (arsenic, cadmium, chromium, copper, lead, mercury, nickel, silver, selenium, vanadium, and zinc), tributyltin (TBT), high-molecular-weight polycyclic aromatic hydrocarbons (HPAHs), dieldrin, and cyanide were not recommended as risk drivers based on background concentrations and/or the uncertainty of the LOE for remedial decisions.

- **Arsenic** – Arsenic was identified as a preliminary COC based on benthic invertebrate tissue LOE (worm HQ = 2.2 and blue crab HQ = 2.2). Arsenic was not recommended as a risk driver because of the uncertainty associated with the evaluation of regulated metals in tissue[6] as a LOE. In addition, the LPRSA exposure point concentration (EPC) for sediment (9.6 mg/kg) was less than regional background (i.e., Jamaica Bay and Mullica River/Great Bay) maximum concentrations (20.7 and 32.8 mg/kg at Jamaica Bay and Mullica River/Great Bay, respectively) and the upper confidence limit on the mean (UCL) for the Mullica River/Great Bay (12 mg/kg). However, the LPRSA EPC for sediment (9.6 mg/kg) was slightly greater than the UCL for Jamaica Bay (7.3 mg/kg) and above Dundee Dam (6.4 mg/kg).

[4] The use of a tissue residue approach for metals (except for methylmercury and selenium) was highly uncertain because of the wide range of strategies used by organisms to store, detoxify, and excrete bioaccumulated metals (e.g., fish and invertebrates may regulate their body burdens of some metals, although metals regulation, and the strategy thereof, is species and metal specific) (Adams et al. 2011; USEPA 2007e). The USEPA framework for metals risk assessment (USEPA 2007e) recommends against the use of a tissue residue approach, stating that the critical body residue (CBR) approach for metals "does not appear to be a robust indicator of toxic dose."

[5] The relevance of soil toxicity thresholds generated for agricultural crops to the sediment exposure of aquatic plants was unknown. Soil toxicity data were primarily from studies of agricultural crops and/or waste soils. Many of these soil toxicity data were based on a highly bioavailable chemical form that is not representative of natural soils or sediment (Efroymson et al. 1997).

[6] The use of a tissue residue approach for metals (except for methylmercury and selenium) was highly uncertain because of the wide range of strategies used by organisms to store, detoxify, and excrete bioaccumulated metals (e.g., fish and invertebrates may regulate their body burdens of some metals, although metals regulation, and the strategy thereof, is species and metal specific) (Adams et al. 2011; USEPA 2007e).

- **Cadmium** – Cadmium was identified as a preliminary COC based on the fish diet LOE (HQs for mummichog [*Fundulus heteroclitus*], common carp [*Cyprinus carpio*], white perch [*Morone americana*], white sucker [*Catastomus commersoni*], and American eel [*Anguilla rostrata*] ranged from 0.70 to 1.3). Cadmium was not identified as a preliminary COC for any other LOE or receptor group, and HQs for fish diet were just above 1.0 for several fish species. This identification was consistent with recommendations by USEPA (2007e). USEPA recommends a dietary assessment of inorganic metals for conservative screening purposes only, because the uptake by and toxicity of inorganic metals to fish can vary widely depending upon a number of factors, including (but not limited to) digestive physiology (e.g., gut residence time), food nutritional quality, distribution and chemical form of metals in prey tissue, and environmental conditions under which toxicity is evaluated (e.g., temperature).
- **Chromium** – Chromium was identified as a preliminary COC based on the benthic invertebrate tissue LOE (worm HQ = 6.0 and mussel HQ = 3.7) and aquatic plants and sediment LOE (HQ = 160). Chromium was not recommended as a risk driver based on the uncertainty of the aquatic plant assessment for sediment. This uncertainty was due to a screening level based on a highly bioavailable chemical form,[7] as well as the uncertainty associated with the evaluation of regulated metals in tissue.[8]
- **Copper** – Copper was identified as a preliminary COC based on the following LOEs: benthic invertebrate tissue (blue crab HQ = 2.1), fish tissue (mummichog, other forage fish, white perch, and American eel HQs ranged from 1.7 to 9.3), bird diet (sandpiper and great blue heron [*Ardea herodias*] HQs ranged from 0.029 to 3.6), surface water (benthic invertebrate, fish, zooplankton, and aquatic plant estuarine and freshwater HQs ranged from 0.14 to 2.7 and from 0.023 to 1.0, respectively), and sediment for aquatic plant populations (HQ = 2.4). Copper was not recommended as a risk driver for the following reasons:
  - Uncertainty associated with the evaluation of regulated metals in tissue.[9]

[7] The relevance of soil toxicity thresholds generated for agricultural crops to the sediment exposure of aquatic plants was unknown. Soil toxicity data were primarily from studies of agricultural crops and/or waste soils. Many of these soil toxicity data were based on a highly bioavailable chemical form that is not representative of natural soils or sediment (Efroymson et al. 1997).

[8] The use of a tissue residue approach for metals (except for methylmercury and selenium) was highly uncertain because of the wide range of strategies used by organisms to store, detoxify, and excrete bioaccumulated metals (e.g., fish and invertebrates may regulate their body burdens of some metals, although metals regulation, and the strategy thereof, is species and metal specific) (Adams et al. 2011; USEPA 2007e).

[9] The use of a tissue residue approach for metals (except for methylmercury and selenium) was highly uncertain because of the wide range of strategies used by organisms to store, detoxify, and excrete bioaccumulated metals (e.g., fish and invertebrates may regulate their body burdens of some metals,

- Uncertainty associated with the aquatic plant assessment for sediment due to a screening level based on a highly bioavailable chemical form.[10]
- Evaluation of background. Dissolved estuarine surface water LPRSA EPCs for copper (2.61 µg/L) were less than the maximum (3.36 µg/L) and UCL (2.7 µg/L) background surface water concentrations above Dundee Dam. Sediment LPRSA EPCs for copper (170 mg/kg) were less than or similar to maximum (209 mg/kg) and UCL (150 mg/kg) background sediment concentration above Dundee Dam.

- **Lead** – Lead was identified as a preliminary COC based on the benthic invertebrate tissue LOE (worm HQs ranged from 0.16 to 2.5), bird diet LOE (spotted sandpiper [*Actitis macularia*] HQs ranged from 0.20 to 10, and belted kingfisher (*Ceryle alcyon*) HQs ranged from 0.015 to 1.1), and sediment LOE for aquatic plant populations (HQ = 2.3). Lead was not recommended as a risk driver based on benthic invertebrate tissue due to uncertainty associated with the evaluation of regulated metals in tissue,[11] uncertainty of the aquatic plant assessment for sediment due to a screening level based on a highly bioavailable chemical form,[12] and the background evaluation. The LPRSA EPC for lead in sediment (270 mg/kg) was less than the UCL (440 mg/kg) background concentration above Dundee Dam.
- **Methylmercury/mercury** – Methylmercury/mercury was identified as a preliminary COC based on the following LOEs: benthic invertebrate tissue (blue crab HQs ranged from 1.3 to 1.5), fish tissue (white catfish [*Ameiurus catus*], American eel, largemouth bass [*Micropterus salmoides*], and smallmouth bass [*Micropterus dolomieu*] HQs ranged from 0.63 to 2.6), fish diet (mummichog, common carp, white perch, white catfish, and American eel HQs ranged from 1.1 to 1.3), fish egg tissue (mummichog HQs ranged from 0.11 to 1.1), bird diet (great blue heron and kingfisher HQs ranged from 0.031 to 1.6), and sediment

although metals regulation, and the strategy thereof, is species and metal specific) (Adams et al. 2011; USEPA 2007e).

[10] The relevance of soil toxicity thresholds generated for agricultural crops to the sediment exposure of aquatic plants was unknown. Soil toxicity data were primarily from studies of agricultural crops and/or waste soils. Many of these soil toxicity data were based on a highly bioavailable chemical form that is not representative of natural soils or sediment (Efroymson et al. 1997).

[11] The use of a tissue residue approach for metals (except for methylmercury and selenium) was highly uncertain because of the wide range of strategies used by organisms to store, detoxify, and excrete bioaccumulated metals (e.g., fish and invertebrates may regulate their body burdens of some metals, although metals regulation, and the strategy thereof, is species and metal specific) (Adams et al. 2011; USEPA 2007e).

[12] The relevance of soil toxicity thresholds generated for agricultural crops to the sediment exposure of aquatic plants was unknown. Soil toxicity data were primarily from studies of agricultural crops and/or waste soils. Many of these soil toxicity data were based on a highly bioavailable chemical form that is not representative of natural soils or sediment (Efroymson et al. 1997).

for aquatic plant populations (HQ = 9.7). Methylmercury and mercury were not recommended as risk drivers for the following reasons:

- Evaluation of background. The sediment LPRSA EPC for mercury (2,900 µg/kg) was less than the UCL (2,910 µg/kg) background sediment concentration above Dundee Dam. In addition, LPRSA methylmercury fish tissue EPCs were less than maximum concentrations for 7 of 10 species above Dundee Dam, and similar to or less than UCLs for 3 of 4 fish species for which UCLs above Dundee Dam could be calculated. Mummichog LPRSA EPCs for methylmercury were less than UCLs in mummichog from Jamaica Bay/Lower Harbor.
- Uncertainty is associated with the aquatic plant assessment for sediment due to a screening level based on a highly bioavailable chemical form.[13]
- Uncertainty associated with the bird diet. The TRV resulting in HQs > 1.0 was derived using an interspecies extrapolation factor of 3 (assumed mallards [*Anas platyrhynchos*] were 3 times less sensitive than the selected avian species evaluated), and was based on exposure to methylmercury dicyandiamide, a fungicide that is not a form of mercury expected to be associated with the LPRSA.

- **Nickel** – Nickel was identified as a preliminary COC based on the benthic invertebrate tissue LOE (worm HQ = 12 and blue crab HQ = 6.0). Nickel was not recommended as a risk driver based on the uncertainty associated with the evaluation of regulated metals in tissue.[14]
- **Silver** – Silver was identified as a preliminary COC based on the benthic invertebrate tissue LOE (blue crab HQ = 1.0). Silver was not recommended as a risk driver based on uncertainty associated with the evaluation of regulated metals in tissue.[15]

[13] The relevance of soil toxicity thresholds generated for agricultural crops to the sediment exposure of aquatic plants was unknown. Soil toxicity data were primarily from studies of agricultural crops and/or waste soils. Many of these soil toxicity data were based on a highly bioavailable chemical form that is not representative of natural soils or sediment (Efroymson et al. 1997).

[14] The use of a tissue residue approach for metals (except for methylmercury and selenium) was highly uncertain because of the wide range of strategies used by organisms to store, detoxify, and excrete bioaccumulated metals (e.g., fish and invertebrates may regulate their body burdens of some metals, although metals regulation, and the strategy thereof, is species and metal specific) (Adams et al. 2011; USEPA 2007e).

[15] The use of a tissue residue approach for metals (except for methylmercury and selenium) was highly uncertain because of the wide range of strategies used by organisms to store, detoxify, and excrete bioaccumulated metals (e.g., fish and invertebrates may regulate their body burdens of some metals, although metals regulation, and the strategy thereof, is species and metal specific) (Adams et al. 2011; USEPA 2007e).

- **Selenium** – Selenium was identified as a preliminary COC based on the benthic invertebrate tissue (worm HQ = 1.1 and blue crab HQ = 1.5) and aquatic plant sediment (HQ = 14) LOEs. Selenium was not recommended as a risk driver based on the uncertainty of the aquatic plant assessment for sediment. This uncertainty was due to a screening level based on a highly bioavailable chemical form.[16] In addition, selenium was not recommended as a risk driver based on a comparison to background; the LPRSA sediment concentration (0.93 mg/kg) was less than the UCL and maximum above Dundee Dam (27 and 2.7[17] mg/kg, respectively) and the UCL from Jamaica Bay (1.4 mg/kg).

- **Vanadium** - Vanadium was identified as a preliminary COC based on the sediment LOE for aquatic plants (HQ = 14). Vanadium was not recommended as a risk driver based on the uncertainty of the aquatic plant assessment for sediment. This uncertainty was due to a screening level based on a highly bioavailable chemical form.[18]

- **Zinc** – Zinc was identified as a preliminary COC based on the LOEs for sediment for aquatic plants (HQ = 3.1) and surface water for aquatic plants (HQ = 21). Zinc was not recommended as a risk driver based on the uncertainty of the aquatic plant assessment for sediment. This uncertainty was due to a screening level based on a highly bioavailable chemical form.[19] In addition, zinc was not recommended as a risk driver based a comparison to background; LPRSA estuarine and freshwater surface water EPCs for dissolved zinc (8.5 and 7.5 µg/L, respectively) were less than the background maximum dissolved zinc concentration above Dundee Dam (9.8 µg/L). In addition, zinc concentrations in surface water based on general surface water criteria for the evaluation of other aquatic receptor groups (i.e., invertebrates, fish, and zooplankton) resulted in HQs < 1.0.

- **TBT** – TBT was identified as a preliminary COC based on aquatic plant populations (surface water HQs ranged from 1.1 to 50). TBT was not

[16] The relevance of soil toxicity thresholds generated for agricultural crops to the sediment exposure of aquatic plants was unknown. Soil toxicity data were primarily from studies of agricultural crops and/or waste soils. Many of these soil toxicity data were based on a highly bioavailable chemical form that is not representative of natural soils or sediment (Efroymson et al. 1997).

[17] Maximum background concentrations were derived excluding outliers. UCL background concentrations were derived including all data. Details on the background evaluation are provided in Appendix J of this BERA.

[18] The relevance of soil toxicity thresholds generated for agricultural crops to the sediment exposure of aquatic plants was unknown. Soil toxicity data were primarily from studies of agricultural crops and/or waste soils. Many of these soil toxicity data were based on a highly bioavailable chemical form that is not representative of natural soils or sediment (Efroymson et al. 1997).

[19] The relevance of soil toxicity thresholds generated for agricultural crops to the sediment exposure of aquatic plants was unknown. Soil toxicity data were primarily from studies of agricultural crops and/or waste soils. Many of these soil toxicity data were based on a highly bioavailable chemical form that is not representative of natural soils or sediment (Efroymson et al. 1997).

recommended as a risk driver based on the background evaluation; surface water EPCs for TBT were represented by maximum concentrations (0.026 µg/L) and detection limits (DLs) (0.05 µg/L) in the LPRSA. The maximum LPRSA TBT concentrations were less than the DL for background surface water above Dundee Dam (0.05 µg/L), and the LPRSA DLs were equal to background DLs from above Dundee Dam. In addition, TBT had a low detection frequency in the surface water of the LPRSA (0 to 1%).

- **HPAHs -** Total HPAHs were identified as a preliminary COC based on the benthic invertebrate tissue LOE for worms (HQs ranged from 0.090 to 3.0) and the bird diet LOE for spotted sandpiper (HQs ranged from 1.9 to 10 by reach; HQ = 4.5 site wide). Total HPAHs were not recommended as a risk driver based on the background evaluation; the LPRSA sediment EPC (46,000 µg/kg) was less than both the EPC and the maximum sediment concentration above Dundee Dam (300,000 and 73,300[20] µg/kg, respectively). No background invertebrate tissue data were available for comparison to LPRSA invertebrate concentrations, so there was some uncertainty with this evaluation.
- **Dieldrin** – Dieldrin was identified as a preliminary COC based on the fish tissue LOE for several fish species: common carp, channel catfish (*Ictalurus punctatus*), American eel, largemouth bass, and northern pike (*Esox lucius*) (HQs ranged from 0.20 to 1.4). The two TRVs used to determine the HQs were derived from the same study (Shubat and Curtis 1986). The higher LOAEL TRV was based on unadjusted data from the 16-week study wherein reduced growth of rainbow trout (*Oncorhynchus mykiss*) was observed, and the lower LOAEL TRV was based on 96-hr LC50 (concentration that is lethal to 50% of an exposed population) data adjusted using extrapolation factors. Given that the HQs were relatively low based on the LOAEL TRV that was adjusted using extrapolation factors, remedial action based on these predicted risks was not recommended. In addition, dieldrin was not recommended as a risk driver based on the background evaluation; the LPRSA sediment EPC (8.3 µg/kg) was less than the EPC above Dundee Dam (17 µg/kg).
- **Cyanide** – Cyanide was identified as a preliminary COC based on surface water (for invertebrate populations [estuarine and freshwater HQs ranged from 1.3 to 4.1 and from 0.23 to 1.0, respectively], fish and zooplankton populations [estuarine HQs ranged from 1.6 to 5.3], and aquatic plant populations [estuarine HQ = 2.0]). Cyanide was not recommended as a risk driver due to its low detection frequency in surface water in the LPRSA; less than 6% of samples in the estuarine portion had detected concentrations of cyanide.

[20] Maximum background concentrations were derived excluding outliers. UCL background concentrations were derived including all data. Details on the background evaluation are provided in Appendix J of this BERA.

ES.2 SITE BACKGROUND INFORMATION

The LPRSA is a large, complex site encompassing the lower 17.4 mi of the Passaic River (river mile [RM] 0 to RM 17.4), from Newark Bay to Dundee Dam, located within a highly urbanized, industrial, and intensely developed region of northern New Jersey. Adjacent land use is predominantly industrial in the lower river (near Newark Bay), and becomes more commercial and recreational near RM 4, and more residential above RM 5. Land use is increasingly residential and recreational above RM 8.

The LPRSA has been industrialized and urbanized for more than two centuries. Beginning with cotton mills concentrated along the river, the LPRSA watershed grew to include manufactured gas plants; petroleum refineries; tanneries; ship building facilities; smelting facilities; pharmaceutical, electronic product, dye, paint, pigment, paper, and chemical manufacturing plants; and other industrial activity facilities (Shear et al. 1996; Malcolm Pirnie 2007b). Major population centers such as Paterson and Newark transformed the watershed with a mix of residential, commercial, and industrial uses. This mixture of activity, as in many other urban river systems, has subjected the LPRSA to a broad range of contaminant loadings and non-chemical stressors from multiple sources over a long period of time. Its distinguishing factor, however, is elevated levels of 2,3,7,8-TCDD, which is atypical of other urban sites. The Lister Avenue site, which was a significant source of 2,3,7,8-TCDD, was identified as operable unit (OU)-1 of the Diamond Alkali Superfund site.

The objective of this BERA is to identify potentially unacceptable risks posed by site-related chemicals to ecological receptors in the LPRSA.[21] The baseline risk assessments for the LPRSA, which include this BERA and the separately prepared human health risk assessment (HHRA), were performed within the context of the larger New York/New Jersey (NY/NJ) Harbor Estuary, which has also undergone significant industrial and urban development. The NY/NJ Harbor Estuary includes a network of waterways, including rivers (e.g., Passaic, Hackensack, Hudson, Elizabeth, Rahway, and Raritan Rivers), tidal straits (e.g., Kill van Kull and Arthur Kill), and bays (e.g., New York, Raritan, and Newark Bays) that are tidally influenced. These baseline risk assessments consider background and reference conditions and site-specific habitat characteristics. This BERA provides the information necessary to assess

[21] An unacceptable risk, which may or may not be linked to ecologically significant adverse effects at the population or community level, equates to potential adverse risk to ecological receptors; an unacceptable risk is identified when an HQ is found to be greater than or equal to one. As described in USEPA guidance (USEPA 1997b), an HQ less than one indicates that the contaminant alone is unlikely to cause adverse ecological effects, and it does not indicate the absence of ecological risk; rather, it should be interpreted based on the severity of the effect reported and the magnitude of the calculated quotient. Those chemicals found to contribute to unacceptable risk (i.e., an HQ greater than or equal to one) in this BERA are considered potential COCs. An evaluation of the chemical-species pairs with regard to background and the uncertainties associated with the assessment are discussed in this BERA when determining potential risk drivers.

current risks for ecological receptors, develop remedial goals in the FS, and make sound risk management decisions related to the protection of ecological receptors.

Consistent with USEPA guidance (2002d, 2005a), an evaluation of contaminated sediment sites should utilize a risk-based framework that is iterative and as site specific as possible given the available data. A key component of assessing the potential risks in the LPRSA was to use the data and information from the LPRSA RI and from recent site-specific studies to inform risk management decisions. The use of site-specific information is consistent with principles articulated by the National Academy of Sciences (NRC 2001) and USEPA guidance (2002d, 2005a) concerning risk management decisions at contaminated sediment sites.

ES.3 Ecological Setting

The ecological setting of the LPRSA is typical of urban systems, with severely reduced habitat quality and increased urban inputs, and has been extensively described previously (Germano & Associates 2005; Iannuzzi et al. 2008; Iannuzzi and Ludwig 2004; Ludwig et al. 2010; Windward and AECOM 2009; Baron 2011).To determine which organisms to assess for potential ecological risk, it is critical to understand this setting and habitat types within and adjacent to the river.

The Lower Passaic River (LPR) has been industrialized and urbanized for more than two centuries. As in many other urban river systems, a mixture of activity has subjected the LPR to a broad range of contaminant loadings from multiple sources (e.g., untreated industrial and municipal wastewater, combined sewer overflow (CSOs)/stormwater outflow (SWOs), direct runoff, and atmospheric deposition) for many years. The quality of the ecological habitat within the LPRSA has been severely impaired. The historical and current industrial use and development of the shoreline (particularly in the lower portion of the LPRSA) have reduced available shoreline habitats to largely marginal quality. Urbanization has also altered the physical characteristics of the LPRSA. Most tidal marshes, wetlands, and mudflats have been filled in or dredged, which, along with the hardened shoreline, has gradually transformed the LPRSA into a highly channelized river. The LPR shoreline can be divided into the following general areas based on habitat and vegetation: 1) a lower portion (primarily below RM 8) that is largely characterized by a developed shoreline with structures abutting industrial properties; and 2) an upper portion (generally above RM 8) that is characterized by mixed vegetation abutting roads, parks, and residential properties. Access to a significant portion of the west bank of this upper portion of the river is limited by State Route 21.

In addition to the physical disturbance from urban development, the LPRSA and its ecological community are influenced by a variety of environmental factors, including episodic fluctuations in salinity, increased turbidity from natural and anthropogenic inputs, depressed dissolved oxygen (DO) due to urban conditions, considerable organic and nutrient inputs, variations in sediment grain size and organic carbon (OC) content, and the impact of invasive and/or non-native species, as well as chemical stressors. These chemical and non-chemical stressors are present under natural and urban conditions and adversely impact the ecology. These chemical and non-chemical stressors are enhanced by anthropogenic activities such as shoreline development, channelization, CSO/SWO discharge, and urban runoff or extreme weather conditions (e.g., hurricanes and droughts).

ES.4 Problem Formulation

The problem formulation for a BERA provides the roadmap for conducting the assessment and provides a basis for dialogue with stakeholders. The problem formulation was developed and approved by USEPA in 2009 (Windward and AECOM 2009). A baseline risk assessment incorporates as much site-specific data and information as possible and is essential for developing remedial goals that are site specific and will support sound risk management decisions for the LPRSA.

The problem formulation presents the LPRSA ecological CSM, including potential exposure pathways, exposure media, and receptors. In the problem formulation, species or representative species per feeding guild are selected for evaluation of that particular feeding guild. For example, the spotted sandpiper was selected to represent probing invertivorous birds that may forage in the LPRSA.

Assessment endpoints (the attribute[s] to be protected) and measurement endpoints were selected for each receptor group and feeding guild. The assessment endpoints and receptor species are presented below:

- **Benthic invertebrate community –** Protection and maintenance (i.e., survival, growth, and reproduction) of the benthic invertebrate community, both as an environmental resource in itself and as one that serves as a forage base for fish and wildlife populations.
- **Blue crab –** Protection and maintenance (i.e., survival, growth, and reproduction) of healthy populations of blue crab[22] that serve as a forage base for fish and wildlife populations and as a base for sports fisheries.
- **Mollusks –** Protection and maintenance (i.e., survival, growth, and reproduction) of healthy mollusk populations.
- **Fish –** Protection and maintenance (i.e., survival, growth, and reproduction) of omnivorous, invertivorous, and piscivorous fish populations that serve as a forage base for fish and wildlife populations and as a base for sports fisheries.

[22] Crayfish were identified in the PFD as representing freshwater macroinvertebrate populations. However, blue crab will be the only species used to represent the macroinvertebrate population, for both the estuarine and freshwater portions of the LPRSA, because of the limited number of crayfish collected in the freshwater portion.

- **Birds –** Protection and maintenance (i.e., survival, growth, and reproduction[23]) of herbivorous, omnivorous,[24] sediment-probing, and piscivorous bird populations.
- **Mammals –** Protection and maintenance (i.e., survival, growth, and reproduction) of aquatic mammal population.
- **Zooplankton –** Maintenance of the zooplankton community that serves as a food base for juvenile fish.
- **Amphibians/Reptiles –** Protection and maintenance (i.e., survival, growth, and reproduction) of healthy amphibian and reptile populations.
- **Aquatic plants –** Maintenance of healthy aquatic plant populations as a food resource and habitat for fish and wildlife populations

Representative species were identified for each assessment endpoint. Empirical data (i.e., surface water, surface sediment, toxicity tests, community data, invertebrate tissue and fish tissue) were used to determine the potential for unacceptable risk to species that may utilize the LPRSA.

ES.5 SLERA

A SLERA was prepared as part of this BERA (Appendix A). Media-specific (i.e., sediment, surface water, tissue, and diet) COIs were defined as those chemicals detected in a given exposure media (i.e., sediment, surface water, or tissue) or in prey tissue of the diet of fish, bird, or mammal receptors. COIs were screened against conservative toxicity screening levels (TSVs) to identify COPECs per medium, as available. Each COI with a maximum concentration equal to or exceeding the TSV was identified as a COPEC that was evaluated further in this BERA. COIs for which a TSV could not be selected were also retained for further evaluation in this BERA.

ES.6 RISK ANALYSIS

The risk analysis was conducted by first determining the potential for exposure, then performing an effects assessment, and finally conducting a risk characterization to estimate the potential for risk. COPECs with HQs ≥ 1.0 based on LOAEL TRVs (using a range of TRVs) were identified as preliminary COCs.[25]

[23] Few aquatic birds currently use the LPRSA for breeding because of habitat constraints. The reproduction assessment endpoint for birds will evaluate whether existing chemical concentrations would impact reproduction if suitable habitat were present.

[24] Consistent with the PFD, omnivorous birds were not identified in the CSM as a feeding guild to be quantitatively evaluated. A representative species was not selected because the evaluation of other avian feeding guilds (i.e., sediment-probing and piscivorous birds) will be protective of omnivorous birds.

[25] Any HQs ≥ 1.0 were identified as a preliminary COC.

ES.6.1 Exposure assessment

For the exposure assessment, each receptor group representing each assessment endpoint is evaluated in terms of exposure to surface sediment and surface water in the LPRSA. For some receptors such as benthic invertebrates, the exposure is described in terms of distribution and co-occurrence with potential COPECs. For other receptor groups such as avian receptors, exposure areas are determined in the LPRSA based on the species utilization and habitat requirements. EPCs were represented by a UCL (e.g., 95% UCL[26]) concentration using USEPA's ProUCL® statistical package (Version 5.0.00) (USEPA 2013d).

ES.6.2 Effects assessment

In the effects assessment, the COPECs are evaluated for the potential for adverse effects for each assessment endpoint. Each COPEC is evaluated for adverse effects based on survival, growth, or reproduction. For benthic invertebrates, the reliability of COPECs is evaluated in terms of the potential for adverse effects on benthic organisms. The results of the effects assessment are TRVs, either as concentrations or dose levels that are used as thresholds in the risk characterization for determination of the potential for unacceptable risk. A range of TRVs was evaluated. TRV selection was based on a comprehensive review of the primary literature and an assessment of acceptability. TRVs were also selected from previous documents developed by USEPA Region 2 for the LPRSA; specific documents from which USEPA recommended TRVs include:

- USEPA's revised draft of the LPR restoration project focused feasibility study (FFS) (Louis Berger et al. 2014)
- USEPA's first draft of the LPRSA FFS (Malcolm Pirnie 2007b)
- USEPA's LPR pathways analysis report [PAR] (Battelle 2005)

These TRVs are consistent with the comments, responses, and directives received from USEPA on June 30, 2017 (USEPA 2017b), September 18, 2017 (USEPA 2017e), July 10, 2018 (USEPA 2018), January 2, 2019 (CDM 2019), and March 5, 2019 (USEPA 2019); during face-to-face meetings or conference calls on July 24, September 27, October 3, November 6, 2017, July 24, 2018, and August 16, 2018; and via additional deliverables and communications between the CPG and USEPA from August through December 2017, July through September 2018, and January through June 2019.

[26] There are cases (e.g., when data are highly skewed) in which USEPA's ProUCL® software recommends a 97.5 or 99% UCL, rather than the 95% UCL.

ES.6.3 Risk characterization, uncertainty analysis, and identification of preliminary COCs and risk drivers

In the risk characterization, the estimation of risk is determined by comparing the COPEC concentration or dose level developed in the exposure analysis to the adverse-effects-level TRVs developed in the effects assessment. The result of this comparison is quantified as HQs, which are estimated for each TRV selected (as defined in ES 6.2). COPECs with HQs ≥ 1.0 (based on a range of LOAEL TRVs) in at least one LOE were identified as preliminary COCs.

The results of this BERA will be used in the FS as a tool for risk managers to make potential remedial decisions for the LPRSA. Determining the potential for unacceptable ecological risk at the population level provides information regarding decisions to be made in the FS or other programmatic environmental management changes. The TRVs used to evaluate risk to various ecological receptors are organism-level effects, including those that affect particular attributes of organisms within a population, such as survival, growth, and reproduction. Survival, growth, and reproduction of individual organisms may, in turn, affect populations of those organisms depending upon the magnitude and severity of the effect. However, population-level effects—such as size or density of population, population growth, or population survival—are more direct measures of influence on the population as a whole. Since BERAs evaluate ***populations*** as assessment endpoints, not individuals, a number of other factors—including the potential magnitude and severity of the effect, the ecological significance of the risk to the population, and the certainty of the assessment—should be evaluated to determine if a risk driver should be used to develop preliminary remediation goals (PRGs) or remedial action levels (RALs). In addition, uncertainty of assumptions surrounding exposure and effects data used to derive risk estimates should be assessed as part of the evaluation of risk management decisions.

The following preliminary COCs were recommended as risk drivers for further evaluation in the FS:

- 2,3,7,8-TCDD
- PCDD/PCDF TEQ (based on fish - TEQ, bird - TEQ, and mammal - TEQ)
- Total TEQ (based on fish - TEQ, bird - TEQ, and mammal - TEQ)
- Total PCBs
- PCB (based on fish - TEQ, bird - TEQ, and mammal - TEQ)
- Total DDx

The above-listed risk drivers are based on varying receptors and LOEs. Some LOEs are more certain than others and should be evaluated prior to making any management decisions (see Table ES-4).

ES.6.3.1 Benthic invertebrates

The assessment of risk to benthic invertebrates included the evaluation of three receptor groups: benthic invertebrate community, macroinvertebrates (blue crabs), and mollusks.

Benthic Invertebrate Community

The assessment of risk to the benthic invertebrate community was conducted using several LOEs. The benthic infaunal invertebrate community, sediment toxicity, and sediment chemistry data (collectively referred to as the SQT), as well as surface water and tissue data, were evaluated in order to characterize risk. The SQT data were evaluated as independent LOEs. A WOE analysis combined the three SQT LOEs into a single location-by-location characterization of risk. The WOE approach was adapted from other studies (Bay et al. 2007; Bay and Weisberg 2012; McPherson et al. 2008). To the extent possible, each LOE and WOE analysis was conducted by comparing LPRSA data to either urban or non-urban reference area data.[27] The LPRSA toxicity test and benthic invertebrate community metric data were compared to reference area data (in addition to comparisons of LPRSA toxicity test data to negative control results). Based on statistical evaluations, sediment chemistry data in the LPRSA were not strongly related to biological responses in benthic invertebrates, particularly benthic community structure. Regardless, sediment chemistry data were evaluated as an independent LOE and incorporated into the WOE analysis.

For the SQT analysis, USEPA identified Jamaica Bay as the estuarine reference area representing an urban habitat, with available SQT data collected and analyzed by others. Similarly, USEPA identified the area upstream of Dundee Dam (Windward 2012a) as a freshwater reference area representing urban habitat, with the reference dataset collected by CPG in the fall of 2012. Mullica River and Great Bay were also identified by USEPA as non-urban reference areas; however, acceptable SQT data were available from only estuarine portions of Mullica River and Great Bay; this data was collected and analyzed by others.

The WOE analysis of the SQT data indicated that LPRSA benthic infaunal invertebrate communities were impacted relative to the selected urban reference areas at 18 of the 97 individual locations in the LPRSA. The SQT data from 28 SQT locations indicated that impacts were low at those locations relative to other urbanized systems (Table ES-3; Figure ES-1). Of the 97 SQT locations, 51 were initially categorized as having a medium impact, which was an uncertain outcome caused by conflicting LOEs or disagreement among components of LOEs (i.e., toxicity was inconsistent across test endpoints and/or decreased community metrics [relative to the reference condition] were inconsistent across metrics). Medium impacts may also, in some cases, be driven by a moderate degree of chemical effects. A post-hoc analysis of medium-

[27] Acceptable non-urban, freshwater SQT reference data were not available to compare with LPRSA data from the tidal freshwater zone.

impact locations using additional site-specific data resulted in 20 locations being recategorized, 8 as likely low impact and 12 as likely impacted. This meant that 31% of SQT locations had high impacts or were likely impacted relative to urban reference conditions, whereas 37% had no, low, or likely low impacts. Impacts at the remaining 32% of LPRSA SQT locations were uncertain but considered to be moderate.

When comparing upper and fluvial estuarine LPRSA SQT locations to non-urban reference conditions, risks were marginally greater for those salinity zones than were risks based on comparing LPRSA SQT locations to urban reference conditions (Tables ES-3 and ES-5). Given that non-urban conditions do not take into account the possible effects on invertebrates of stressors associated with urbanization, the increase in calculated risks is to be expected. Reference conditions are meant to represent the site but for the release of site-related hazardous materials, so the use of a non-urban reference condition to characterize risks for the LPRSA (an urban system) is less relevant than the use of an urban reference condition.

Table ES-5. Summary of WOE results by salinity zone compared with reference data representing non-urban habitats

Benthic Salinity Zone	N	No Impact		Low Impact		Medium Impact: Likely Low Impact		Medium Impact: Medium Impact (Unchanged)		Medium Impact: Likely Impacted		High Impact	
		n	%	n	%	n	%	n	%	n	%	n	%
Upper estuarine (RM 0 to RM 4)	25	0	0%	12	48%	0	0%	4	16%	6	24%	3	12%
Fluvial estuarine (RM 4 to RM 13)	54	0	0%	8	15%	5	9%	23	43%	9	17%	9	17%
Both estuarine zones (RM 0 to RM 13)	79	0	0%	20	25%	5	6%	27	34%	15	19%	12	15%

n – sample size (by category)
N – sample size (by benthic salinity zone)
RM – river mile
WOE – weight of evidence

Worms

The potential for risk to worms was characterized using LPRSA bioaccumulation tissue and water chemistry.[28] Worm tissue and surface water concentrations were compared to TRVs to derive risk estimates (HQs) in the risk characterization.

COPECs were evaluated in both surface water and worm tissue. Based on this assessment, 3 preliminary COCs were identified with HQs ≥ 1.0 for surface water (2,3,7,8-TCDD, copper, and cyanide), and 11 preliminary COCs were identified for worm tissue (2,3,7,8-TCDD, PCDD/PCDF TEQ - fish, total TEQ - fish, total HPAHs,

[28] The evaluation of the benthic invertebrate community and sediment is considered to be protective of worm populations.

total PCBs, total DDx, selenium, and four regulated metals [arsenic, chromium, lead, and nickel]).

Macroinvertebrates (Blue Crab)

The potential for risk to blue crab was characterized using LPRSA tissue and water chemistry.[29] Blue crab tissue and surface water concentrations were compared to TRVs to derive risk estimates (HQs) in the risk characterization.

COPECs were evaluated in surface water and tissue. Based on this assessment, 10 preliminary COCs were identified with HQs ≥ 1 in blue crab tissue (2,3,7,8-TCDD, PCDD/PCDF TEQ - fish, total TEQ - fish, total PCBs, total DDx, methylmercury/mercury, selenium, and three regulated metals [arsenic, copper, and silver]), and 3 preliminary COCs were identified in surface water (2,3,7,8-TCDD, cyanide, and copper).

Mollusks

The potential for risk to mollusks was characterized using LPRSA tissue and water chemistry.[30] *In situ* mussel tissue and surface water concentrations were compared to TRVs to derive risk estimates (HQs) in the risk characterization.

COPECs were evaluated in surface water and mussel tissue. Based on this assessment, six preliminary COCs were identified in mussel tissue (total PCBs, 2,3,7,8-TCDD, PCDF/PCDD TEQ - fish, total TEQ - fish, and two regulated metals [chromium and nickel]), and three preliminary COCs were identified in surface water (2,3,7,8-TCDD, copper, and cyanide).

ES.6.3.2 Fish

The potential for risk to a number of fish species representing various feeding guilds (i.e., benthic omnivores [mummichog, other forage fish, and common carp], invertivores [white perch, channel catfish, brown bullhead (*Ameiurus nebulosus*), white catfish, and white sucker], and piscivores [American eel, largemouth bass, northern pike, and smallmouth bass]) was characterized using multiple LOEs. Fish tissue, dietary doses, surface water, and modeled fish egg concentrations were compared to TRVs to derive risk estimates (i.e., HQs) in the risk characterization. In addition, several qualitative LOEs were evaluated that involved the assessment of LPRSA data for mummichog egg counts and gross external and internal health observations.

[29] For the sediment LOE, the evaluation of the benthic invertebrate community and sediment is considered to be protective of macroinvertebrates populations.

[30] The evaluation of the benthic invertebrate community and sediment is considered to be protective of mollusk populations.

COPECs with HQs ≥ 1.0 based on LOAEL TRVs (using a range of TRVs) were identified as preliminary COCs. The following preliminary COCs were identified:

- **Benthic omnivorous fish populations**
    - **Mummichog and other forage fish** – five preliminary COCs (total PCBs, 2,3,7,8-TCDD, PCDD/PCDF TEQ - fish, total TEQ - fish, and one regulated metal [copper]) were identified based on the tissue LOE; four preliminary COCs (cadmium, mercury, PCDD/PCDF TEQ - fish, and total TEQ - fish) were identified based on the dietary LOE. Total PCBs and methylmercury/mercury were identified as preliminary COCs for mummichog based on the egg tissue LOE.
    - **Common carp** – seven preliminary COCs (2,3,7,8-TCDD, PCDD/PCDF TEQ - fish, total TEQ - fish, total PCBs, PCB TEQ - fish, dieldrin, and total DDx) were identified based on the tissue LOE, and four preliminary COCs (cadmium, mercury, PCDD/PCDF TEQ - fish, and total TEQ - fish) were identified based on the dietary LOE.
- **Invertivorous fish populations**
    - **White perch** – six preliminary COCs (2,3,7,8-TCDD, PCDD/PCDF TEQ - fish, total TEQ - fish, total PCBs, PCB TEQ fish, and one regulated metal [copper]) were identified based on the tissue LOE, and five preliminary COCs (cadmium, mercury, PCDD/PCDF TEQ - fish, PCB TEQ - fish, and total TEQ - fish) were identified based on the dietary LOE.
    - **Channel catfish** – six preliminary COCs (2,3,7,8-TCDD, PCDD/PCDF TEQ - fish, total TEQ - fish, total PCBs, PCB TEQ - fish, and dieldrin) were identified based on the tissue LOE, and two preliminary COCs (PCDD/PCDF TEQ - fish and total TEQ - fish) were identified based on the dietary LOE.
    - **Brown bullhead** – four preliminary COCs (2,3,7,8-TCDD, PCDD/PCDF TEQ - fish, total TEQ - fish, and total PCBs) were identified based on the tissue LOE.
    - **White catfish** – six preliminary COCs (mercury, 2,3,7,8-TCDD, PCDD/PCDF TEQ - fish, total TEQ - fish, total PCBs, and PCB TEQ - fish) were identified based on the tissue LOE, and three preliminary COCs (mercury, PCDD/PCDF TEQ - fish, and total TEQ - fish) were identified based on the dietary LOE.
    - **White sucker** – five preliminary COCs (2,3,7,8-TCDD, PCDD/PCDF TEQ - fish, total TEQ - fish, total PCBs, and PCB TEQ - fish) were identified based on the tissue LOE, and three preliminary COCs (cadmium, PCDD/PCDF TEQ - fish, and total TEQ - fish) were identified based on the dietary LOE.

- **Piscivorous fish populations**
  - **American eel** – seven preliminary COCs **(**2,3,7,8-TCDD, PCDD/PCDF TEQ - fish, total TEQ - fish, total PCBs, methylmercury, dieldrin, and one regulated metal [copper]) were identified based on the tissue LOE, and five preliminary COCs (cadmium, mercury, PCB TEQ - fish, PCDD/PCDF TEQ - fish, and total TEQ - fish) were identified based on the dietary LOE.
  - **Largemouth bass** – seven preliminary COCs **(**2,3,7,8-TCDD, PCDD/PCDF TEQ - fish, total TEQ - fish, methylmercury, total PCBs, PCB TEQ, and dieldrin) were identified as based on the tissue LOE, and three preliminary COCs (PCB TEQ - fish, PCDD/PCDF TEQ - fish, and total TEQ - fish) were identified based on the dietary LOE.
  - **Northern pike** – six preliminary COCs (2,3,7,8-TCDD, PCDD/PCDF TEQ - fish, total TEQ - fish, total PCBs, PCB TEQ, and dieldrin) were identified based on the tissue LOE, and three preliminary COCs (PCB TEQ - fish, PCDD/PCDF TEQ - fish, and total TEQ - fish) were identified based on the dietary LOE.
  - **Smallmouth bass** – five preliminary COCs (mercury, 2,3,7,8-TCDD, PCDD/PCDF TEQ - fish, total TEQ - fish, and total PCBs) were identified based on the tissue LOE, and three preliminary COCs (PCB TEQ - fish, PCDD/PCDF TEQ - fish, and total TEQ - fish) were identified based on the dietary LOE.

Of the 28 COPECs evaluated in surface water, HQs were < 1.0 for 26. Two COPECs (cyanide and copper) were identified as preliminary COCs with HQs ≥ 1.0.

ES.6.3.3 Aquatic birds

The potential for risk to three bird species (spotted sandpiper, great blue heron, and belted kingfisher) was characterized using LPRSA tissue, sediment, and water chemistry to estimate dietary doses. In addition, risks to great blue heron and belted kingfisher were characterized using chemical concentrations in bird egg tissue as a secondary LOE. Dietary doses and modeled bird egg concentrations were compared to a range of TRVs to derive risk estimates (HQs) in the risk characterization.

Sixteen COPECs were evaluated for aquatic avian dietary exposure and seven COPECs were evaluated using the bird egg tissue LOE. The following preliminary COCs were identified with HQs ≥ 1.0 (based on a range of LOAEL TRVs):

- **Spotted sandpiper** – eight preliminary COCs (PCDD/PCDF TEQ - bird, total TEQ, copper, lead, total HPAHs, total PCBs, PCB TEQ - bird, and total DDx) were identified based on the dietary LOE.
- **Great blue heron –** seven preliminary COCs **(**PCDD/PCDF TEQ - bird, total TEQ - bird, copper, methylmercury, total PCBs, PCB TEQ - bird, and total DDx)

were identified based on the dietary LOE, and five preliminary COCs (PCDD/PCDF TEQ - bird, total TEQ - bird, total PCBs, PCB TEQ, and total DDx) were identified based on the egg tissue LOE.

- **Belted kingfisher** – six preliminary COCs (PCDD/PCDF TEQ - bird, total TEQ - bird, lead, methylmercury, PCB TEQ - bird, and total DDx) were identified based on the dietary LOE, and five preliminary COCs (PCDD/PCDF TEQ - bird, total TEQ - bird, total PCBs, PCB TEQ - bird, and total DDx) were identified based on the egg tissue LOE.

ES.6.3.4 Aquatic mammals

The potential for risk to mammals was characterized using LPRSA tissue, sediment, and water chemistry to estimate dietary doses of COPECs for two mammal species (i.e., river otter [*Lontra canadensis*] and mink [*Neovison vison*]). Dietary doses were compared to a range of TRVs to derive risk estimates (HQs) in the risk characterization.

Fifteen COPECs were evaluated for aquatic mammals. Four preliminary COCs (PCDD/PCDF TEQ - mammal, total TEQ - mammal, total PCBs, and PCB TEQ - mammal) were identified with HQs ≥ 1.0 (based on a range of LOAEL TRVs) for river otter and mink.

ES.6.3.5 Zooplankton

The potential for unacceptable risk to zooplankton was characterized using LPRSA water chemistry. Surface water concentrations were compared to TRVs intended to be protective of a variety of aquatic organisms to derive risk estimates (HQs) in the risk characterization.

Twenty-five COPECs were evaluated in surface water, two of which had HQs ≥ 1.0 (copper [estuarine surface water] and cyanide) and were identified as preliminary COCs.

ES.6.3.6 Amphibians/reptiles

The potential for unacceptable risk to amphibians and reptiles was characterized using LPRSA surface water chemistry. Surface water concentrations were compared to amphibian-specific TRVs based on the literature to derive risk estimates (HQs) in the risk characterization. Limited amphibian- and reptile-specific water toxicity data are available, so the evaluation of risks to amphibians and reptiles is limited and uncertain.

No preliminary COCs were identified for amphibians and reptiles, because all seven surface water COPECs (chromium, copper, lead, mercury, nickel, silver, and zinc) evaluated had HQs < 1.0.

ES.6.3.7 Aquatic plants

The potential for unacceptable risk to aquatic plants was characterized using LPRSA surface sediment and surface water chemistry. Surface water and sediment data were compared to media-specific TRVs to derive risk estimates (HQs). The paucity and questionable applicability of both exposure and effects data—especially for the sediment evaluation, which was based on terrestrial plants and soil—reduce the level of certainty of the quantitative estimates of risk to the aquatic plant community.

Seven preliminary COCs (chromium, copper, lead, mercury, selenium, vanadium, and zinc) were identified for aquatic plants based on the sediment LOE, and four preliminary COCs (copper, zinc, TBT, and cyanide) were identified for aquatic plants based on the surface water LOE.

1 Introduction

This Final 3 Baseline Ecological Risk Assessment for the Lower Passaic River Study Area, hereinafter referred to as the baseline ecological risk assessment (BERA), has been prepared as part of the Lower Passaic River Study Area (LPRSA) remedial investigation/feasibility study (RI/FS) of the 17.4-mi stretch of the Passaic River between Dundee Dam and Newark Bay. This BERA presents the results of the ecological risk assessment (ERA) prepared under US Environmental Protection Agency Region 2 (USEPA) oversight and direction, and was conducted in accordance with Section IX.37.d. of the May 2007 Settlement Agreement (USEPA 2007a). This final BERA has been amended to address comments, responses, and directives received from USEPA on January 2, 2019 (CDM 2019), March 5, 2019 (USEPA 2019) and via additional communications between the CPG and USEPA from January through June 2019.

Developing a site-specific BERA is particularly important in an urban setting such as the LPRSA. The LPRSA is a large, complex site encompassing the lower 17.4 mi of the Passaic River (river mile [RM] 0 to RM 17.4), from Newark Bay to Dundee Dam, within a highly urbanized and developed region of northern New Jersey. Adjacent land use is predominantly industrial in the lower river (near Newark Bay); it becomes more commercial, residential, and recreational near RM 4, and increasingly residential and recreational above RM 8.

The Lower Passaic River (LPR) has been industrialized and urbanized for more than two centuries, having served as the receiving environment for industrial and municipal waste discharges since the 19th century. The LPRSA is located within what was one of the major centers of the American Industrial Revolution. Beginning with cotton mills concentrated along the river, the LPR watershed grew to include manufactured gas plants; petroleum refineries; tanneries; ship building facilities; smelting facilities; pharmaceutical, electronic product, dye, paint, pigment, paper, and chemical manufacturing plants; and other industrial activities facilities (Shear et al. 1996; Malcolm Pirnie 2007b). Major population centers such as Paterson and Newark transformed the watershed with a mix of residential, commercial, and industrial uses. This mixture of activity, as in many other urban river systems, has subjected the LPR to a broad range of contaminant loadings from multiple sources (e.g., untreated industrial and municipal wastewater, combined sewer overflow [CSOs]/stormwater outflows [SWOs], direct runoff, and atmospheric deposition) for a long time. Its distinguishing factor is elevated levels of 2,3,7,8-tetrachlorodibenzo-*p*-dioxin (TCDD), which is atypical among urban sites.

The LPR first became the focus of RIs because of contamination resulting from discharges from the former manufacturing facility located at 80 and 120 Lister Avenue in Newark, New Jersey. These properties (hereinafter referred to as the Lister Avenue site) were operated by various companies for more than 100 years. Chemical manufacturing and compounding occurred at the Lister Avenue site from the 1940s through the 1960s (Bopp et al. 1991; Bopp et al. 1998; Chaky 2003; Lillienfeld and Gallo 1989). Kolker Chemical Company (later acquired by Diamond Alkali Company in 1951) manufactured dichlorodiphenyltrichloroethane (DDT) and pesticides at the Lister Avenue site in the 1940s. Between 1951 and 1969, Diamond Alkali Company (subsequently known as the Diamond Shamrock Chemicals Company) manufactured chemicals such as pesticides and phenoxy herbicides, including the primary components used to make the military defoliant Agent Orange. The property was used by subsequent owners until 1983, when high levels of 2,3,7,8-TCDD were detected in on- and off-site soils and groundwater. Based on these findings, USEPA added the Diamond Alkali site (also referred to as the Lister Avenue site) to the National Priorities List in September 1984. The Lister Avenue site was a significant source of 2,3,7,8-TCDD and DDT to the LPRSA (some investigators have concluded that it was the dominant source of 2,3,7,8-TCDD to the river) (Bopp et al. 1991; Bopp et al. 1998; Chaky 2003; Hansen 2002). The property itself was identified as operable unit (OU)-1 of the Diamond Alkali Superfund site. Subsequent investigations in the Passaic River and Newark Bay have been undertaken as additional OUs.

Urbanization has altered the physical characteristics of the LPRSA. Most tidal marshes, wetlands, and mudflats have been filled in or dredged, gradually transforming the LPR into a highly channelized river, the lower 8 mi of which are dominated by hardened shorelines (e.g., sheet pile, riprap, and wood pilings) (Malcolm Pirnie 2007a). Land use developments have altered the ecology and limited human uses of the river and shoreline. Currently, most (approximately 70%) of the riverbank along the lower portion of the LPRSA (from RM 1 to RM 7) is composed of bulkhead and/or riprap and supports a limited amount of vegetation (Windward 2014b). The upper portion of the LPRSA riverbank (from RM 7 to RM 17.4) is primarily dominated by mixed vegetation, generally over steep banks.

The objective of this BERA is to identify unacceptable risks posed by site-related chemicals to ecological receptors in the LPRSA.[31] The baseline risk assessments for the LPRSA, which include this BERA and the separately prepared human health risk assessment (HHRA), were performed within the context of the larger New York/New Jersey (NY/NJ) Harbor Estuary, which has also undergone significant industrial and

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[31] An unacceptable risk, which may or may not be linked to ecologically significant adverse effects at the population or community level, equates to potential adverse risk to ecological receptors; an unacceptable risk is identified when an HQ is found to be greater than or equal to one. As described in USEPA guidance (USEPA 1997b), an HQ less than one indicates that the contaminant alone is unlikely to cause adverse ecological effects, and it does not indicate the absence of ecological risk; rather, it should be interpreted based on the severity of the effect reported and the magnitude of the calculated quotient. Those chemicals found to contribute to unacceptable risk (i.e., an HQ greater than or equal to one) in this BERA are considered potential COCs. An evaluation of the chemical-species pairs with regard to background and the uncertainties associated with the assessment are discussed in this BERA when determining potential risk drivers.

urban development. The NY/NJ Harbor Estuary includes a network of waterways, including rivers (e.g., Passaic, Hackensack, and Hudson Rivers), tidal straits (e.g., Kill van Kull and Arthur Kill), and bays (e.g., New York Harbor and Raritan and Newark Bays), that are tidally influenced. These baseline risk assessments, therefore, consider background and reference conditions and site-specific habitat characteristics. This BERA therefore provides the information necessary for assessing current risk to ecological receptors, developing remedial goals in the FS, and making sound risk management decisions related to the protection of ecological receptors.

Documents that have been prepared to support the preparation of this BERA include:

- *Lower Passaic River Restoration Project Draft Field Sampling Plan, Volume 2* (Malcolm Pirnie et al. 2006)
- *Lower Passaic River Restoration Project Pathways Analysis Report* (Battelle 2005)
- *LPRSA Human Health and Ecological Risk Assessment Streamlined 2009 Problem Formulation* (Windward and AECOM 2009), hereafter referred to as the problem formulation document (PFD)
- Revised *LPRSA Toxicity Reference Value Deliverable* (Appendix A, Attachment A3)
- *Data Usability and Data Evaluation Plan for the LPRSA Risk Assessments* (Windward and AECOM 2015)

Table 1-1 presents a list of CPG's quality assurance project plans (QAPPs) and data reports that support this BERA.

Table 1-1. List of QAPPs and data reports

QAPPs			Associated Data Reports		
Document	**Date Approved by USEPA**	**Reference**	**Document**	**Date Approved by USEPA**	**Reference**
Quality Assurance Project Plan: RI Low Resolution Coring/Sediment Sampling, Rev. 4	October 20, 2008	ENSR/AECOM (2008)	Revised Low Resolution Coring Report	July 20, 2015	AECOM (2014a)
Quality Assurance Project Plan: Fish and Decapod Crustacean Tissue Collection for Chemical Analysis and Fish Community Survey	August 6, 2009	Windward (2009a)	Fish and Decapod Field Report for the Late Summer/Early Fall 2009 Field Effort	September 14, 2010	Windward (2010c)
			2009 Fish and Blue Crab Tissue Chemistry Data Report for the Lower Passaic River Study Area	pending approval; submitted November 23, 2015	Windward (2018b)
Quality Assurance Project Plan: Surface Sediment Chemical Analyses and Benthic Invertebrate Toxicity and Bioaccumulation Testing	October 8, 2009	Windward (2009b)	Fall 2009 Benthic Invertebrate Community Survey and Benthic Field Data Collection Report for the Lower Passaic River Study Area	January 6, 2014	Windward (2014a)
			Fall 2009 Sediment Toxicity Test Data for the Lower Passaic River Study Area	pending approval; submitted November 20, 2015	Windward (2018f)
			2009 and 2010 Sediment Chemistry Data for the Lower Passaic River Study Area	July 20, 2015	Windward (2015a)
			2009 Bioaccumulation Tissue Chemistry Data for the Lower Passaic River Study Area	August 7, 2018	Windward (2018a)
Winter 2010 Fish Community Survey, Addendum to the Quality Assurance Project Plan: Fish and Decapod Crustacean Tissue Collection for Chemical Analysis and Fish Community Survey, Addendum No. 1	January 25, 2010	Windward (2010j)	Fish Community Survey and Tissue Collection Data Report for the Lower Passaic River Study Area 2010 Field Efforts	July 20, 2011	Windward (2011c)
Late Spring/Early Summer 2010 Fish Community Survey, Addendum to the Quality Assurance Project Plan: Fish and Decapod Crustacean Tissue Collection for Chemical Analysis and Fish Community Survey, Addendum No. 3	June 22, 2010	Windward (2010e)			

Table 1-1. List of QAPPs and data reports

QAPPs			Associated Data Reports		
Document	**Date Approved by USEPA**	**Reference**	**Document**	**Date Approved by USEPA**	**Reference**
Quality Assurance Project Plan/Field Sampling Plan Addendum, Remedial Investigation Water Column Monitoring/Physical Data Collection for the Lower Passaic River, Newark Bay, and Wet Weather Monitoring, Rev. 4	March 2010	AECOM (2010a)	Physical Water Column Monitoring Sampling Program Characterization Summary	pending approval; submitted March 2014	AECOM (2019a)
Spring and Summer 2010 Benthic Invertebrate Community Surveys, Addendum to the Quality Assurance Project Plan: Surface Sediment Chemical Analyses and Benthic Invertebrate Toxicity and Bioaccumulation Testing, Addendum No. 1	May 17, 2010	Windward (2010i)	Spring and Summer 2010 Benthic Invertebrate Community Survey Data for the Lower Passaic River Study Area	January 15, 2014	Windward (2014c)
Late Spring/Early Summer 2010 Fish Tissue Collection, Addendum to the Quality Assurance Project Plan: Fish and Decapod Crustacean Tissue Collection for Chemical Analysis and Fish Community Survey, Addendum 4	June 21, 2010	Windward (2010f)	2010 Small Forage Fish Tissue Chemistry Data for the Lower Passaic River Study Area	pending approval; submitted November 23, 2015	Windward (2018c)
Avian Community Survey, Addendum to the Quality Assurance Project Plan: Fish and Decapod Crustacean Tissue Collection for Chemical Analysis and Fish Community Survey, Addendum No. 2	August 9, 2010	Windward (2010a)	Avian Community Survey Data Report for the Lower Passaic River Study Area Summer and Fall 2010	August 8, 2011	Windward (2011a)
			Avian Community Survey Data Report for the Lower Passaic River Study Area Winter and Spring 2011	pending approval; submitted November 20, 2015	Windward (2019e)
Collection of Surface Sediment Samples Co-Located with Small Forage Fish Tissue Samples, Addendum to the Quality Assurance Project Plan: Surface Sediment Chemical Analyses and Benthic Invertebrate Toxicity and Bioaccumulation Testing, Addendum No. 2	August 13, 2010	Windward (2010b)	2009 and 2010 Sediment Chemistry Data for the Lower Passaic River Study Area	July 20, 2015	Windward (2015a)

Table 1-1. List of QAPPs and data reports

QAPPs			Associated Data Reports		
Document	Date Approved by USEPA	Reference	Document	Date Approved by USEPA	Reference
Habitat Identification Survey, Addendum to the Quality Assurance Project Plan: Surface Sediment Chemical Analyses and Benthic Invertebrate Toxicity and Bioaccumulation Testing, Addendum No. 3	September 13, 2010	Windward (2010d)	Habitat Identification Survey Data Report for the Lower Passaic River Study Area Fall 2010 Field Effort	January 6, 2014	Windward (2014b)
Caged Bivalve Study, Addendum to the Quality Assurance Project Plan: Surface Sediment Chemical Analyses and Benthic Invertebrate Toxicity and Bioaccumulation Testing, Addendum No. 4	March 2, 2011	Windward (2011b)	2011 Caged Bivalve Study Data for the Lower Passaic River Study Area	pending approval; submitted November 23, 2015	Windward (2019a)
Quality Assurance Project Plan, Lower Passaic River Study Area River Mile 10.9 Characterization, Rev. 2	August 2011	AECOM (2011)	River Mile 10.9 Characterization Program Summary, Lower Passaic River Study Area	pending approval; submitted April 19, 2012	CH2M HILL and AECOM (Draft)
Quality Assurance Project Plan, Lower Passaic River Restoration Project, Low Resolution Coring Supplemental Sampling Program, Rev. 3	June 2012	AECOM (2012a)	Low Resolution Coring Supplemental Sampling Program Characterization Summary	pending approval; submitted August 2013	AECOM (2013a)
Quality Assurance Project Plan/Field Sampling Plan Addendum, RI Water Column Monitoring/Small Volume Chemical Data Collection, Rev. 2	2011	AECOM (2012c)	Small Volume Chemical Water Column Monitoring Sampling Program Characterization Summary	pending approval; submitted February 2014	AECOM (2019b)
Quality Assurance Project Plan, Lower Passaic River Restoration Project, RI Water Column Monitoring/High Volume Chemical Data Collection, Rev. 2	December 2012	AECOM (2012b)	High Volume Chemical Water Column Monitoring Sampling Program Characterization Summary	pending approval; submitted February 2014	AECOM (2014b)
Summer and Fall 2012 Dissolved Oxygen Monitoring Program, Addendum to the Quality Assurance Project Plan: Remedial Investigation Water Column Monitoring/Physical Data Collection for the Lower Passaic River, Newark Bay, and Wet Weather Monitoring, Addendum No. 1	August 6, 2012	Windward (2012c)	Dissolved Oxygen Monitoring Program Data Report for the Lower Passaic River Study Area: Summer and Fall 2012	pending approval; submitted November 23, 2015	Windward (2018e)

Table 1-1. List of QAPPs and data reports

QAPPs			Associated Data Reports		
Document	Date Approved by USEPA	Reference	Document	Date Approved by USEPA	Reference
Background Tissue Addendum to the Quality Assurance Project Plan: Fish and Decapod Crustacean Tissue Collection for Chemical Analysis and Fish Community Survey, Addendum No. 5	October 10, 2012	Windward (2012b)	2012 Fish Tissue Survey and Chemistry Background Data for the Lower Passaic River Study Area	pending approval, submitted July 22, 2015	Windward (2019c)
Background and Reference Conditions Addendum to the Quality Assurance Project Plan: Surface Sediment Chemical Analyses and Benthic Invertebrate Toxicity and Bioaccumulation Testing, Addendum No. 5	October 26, 2012	Windward (2012a)	2012 Benthic Invertebrate Community Reference Data for the Lower Passaic River Study Area	pending approval; submitted August 26, 2013	Windward (2019b)
			2012 Sediment Toxicity Reference Data for the Lower Passaic River Study Area	pending approval; submitted October 22, 2013	Windward (2018d)
			2012 Sediment Chemistry Background Data for the Lower Passaic River Study Area	pending approval; submitted October 30, 2013	Windward (2019d)

QAPP – quality assurance project plan
USEPA – US Environmental Protection Agency
RI – remedial investigation
Windward – Windward Environmental LLC

The remainder of this document is organized as follows.

- Section 2 presents the environmental setting of the LPRSA, including physical and habitat characteristics (i.e., environmental factors and ecological habitat); benthic community characteristics; surveys of fish and decapod communities; bird community; mammalian community; amphibian and reptile communities; and threatened, endangered, and special status species.
- Section 3 presents a summary of the updated problem formulation, including the assessment endpoints, risk questions, and measurement endpoints used in the evaluation of risks, consistent with the PFD (Windward and AECOM 2009). Section 3 also presents a summary of the updated ecological conceptual site model (CSM) based on site-specific surveys conducted by CPG.
- Section 4 presents a summary of the data quality objectives (DQOs), data used, and data reduction rules.
- Section 5 presents a summary of the screening-level ecological risk assessment (SLERA), which was used to develop the list of chemicals of potential ecological concern (COPECs) for further evaluation in this BERA.
- Section 6 presents the benthic invertebrate assessment, including a description and evaluation of the lines of evidence (LOEs) for the benthic assessment, risk characterization for each LOE, and the final weight of evidence (WOE) approach and final conclusions of the potential for unacceptable risk to the benthic community.
- Sections 7 through 12 present assessments for the remaining receptor groups that were evaluated, including a description and evaluation of each LOE, a risk characterization for each LOE, and the final conclusions of the potential for unacceptable risk to receptor populations or communities in the LPRSA. The specific sections are as follows:
    - Section 7. Fish Assessment
    - Section 8. Bird Assessment
    - Section 9. Mammal Assessment
    - Section 10. Zooplankton Assessment
    - Section 11. Amphibian and Reptile Assessment
    - Section 12. Aquatic Plant Assessment
- Section 13 presents the summary and risk conclusions per receptor and assessment endpoint, the preliminary chemicals of concern (COCs), and the final conclusions and risk drivers for consideration in the FS.

These sections are supported by the following appendices:

- Appendix A. LPRSA Screening Level Ecological Risk Assessment
- Appendix B. Benthic Data Calculation Files
- Appendix C. BERA EPC Values
- Appendix D. Derivation of Surface Water TRVs for the BERA
- Appendix E. Methods Used to Derive LPRSA BERA Tissue and Dietary TRVs Based on the General Literature
- Appendix F. Toxicity Profiles
- Appendix G. HQ Calculations
- Appendix H. Sensitivity Analysis of Risk Estimates for Mink and River Otter
- Appendix I. Mink Habitat Analysis
- Appendix J. Derivation of Background Concentrations
- Appendix K. BERA Data
- Appendix L. Background and Reference Area Data
- Appendix M. LPRSA Benthic Species List
- Appendix N. Risk Assessment of Amphibians/Reptiles
- Appendix O. Risk Assessment of Aquatic Plants
- Appendix P. Sediment Quality Triad Lines of Evidence for the Baseline Ecological Risk Assessment of LPRSA Benthic Invertebrates
- Appendix Q. Lower Passaic River Study Area Upper 9-Mile Evaluation Ecological Risk Assessment

2 Ecological Setting

The ecological setting of the LPRSA is typical of urban systems, with severely reduced habitat quality and increased urban inputs, and has been extensively described previously (Germano & Associates 2005; Iannuzzi et al. 2008; Iannuzzi and Ludwig 2004; Ludwig et al. 2010; Windward and AECOM 2009; Baron 2011). To determine which organisms to assess for potential ecological risk, it is critical to understand this setting and the habitat types within and adjacent to the river.

As presented in Section 1, the LPR has been industrialized and urbanized for more than two centuries. As in many other urban river systems, a mixture of activity has subjected the LPR to a broad range of contaminant loadings from multiple sources (e.g., untreated industrial and municipal wastewater, CSOs/SWOs, direct runoff, and atmospheric deposition) for a long time. The LPR's distinguishing factor is elevated levels of 2,3,7,8-TCDD, which is atypical among urban sites. The Lister Avenue site, which was a significant source of 2,3,7,8-TCDD, was identified as OU-1 of the Diamond Alkali Superfund site.

While the above describes the chemical inputs, this section discusses the overall environmental setting of the LPRSA (Section 2.1), including environmental factors and habitat, as well as various species that are present in the LPRSA (Sections 2.2 to 2.7). This ecological information was used in the development of receptors for this BERA.

2.1 Environmental Setting

The quality of the ecological habitat within the LPRSA has been severely impaired. The historical and current industrial uses and residential development of the shoreline (particularly in the lower portion of the LPRSA) have reduced available shoreline habitats to largely marginal quality. Urbanization has also altered the physical characteristics of the LPRSA. Most tidal marshes, wetlands, and mudflats have been filled in or dredged, thus gradually transforming the LPR into a highly channelized river. The LPRSA shoreline can be divided into the following general areas based on habitat and vegetation: 1) a lower portion (primarily below RM 8) that is largely characterized by a developed shoreline marked by bulkhead and riprap abutting industrial properties; and 2) an upper portion (generally above RM 8) that is characterized by mixed vegetation abutting roads, parks, and residential properties. Access to the west bank of the upper portion of the river is limited by State Route 21, which abuts the LPR.

Urbanization within the watershed of the LPRSA has resulted in extensive habitat loss, namely of wetlands, small tributaries, submerged aquatic vegetation (SAV), and emergent woodlands. Furthermore, hydrologic alterations (e.g., dredging and hardened shorelines) to the LPRSA and its tributaries have resulted in significant changes to aquatic vegetated habitat within the LPRSA. The loss of wetlands, in particular, likely contributed to declines in the richness of avian and mammalian fauna within the LPRSA (Parsons 1993; Burger 1993). Shoreline habitat is limited in the LPRSA due to the physical development associated with urbanization along the banks of the river, particularly in the lower portion of the LPRSA (below RM 8).

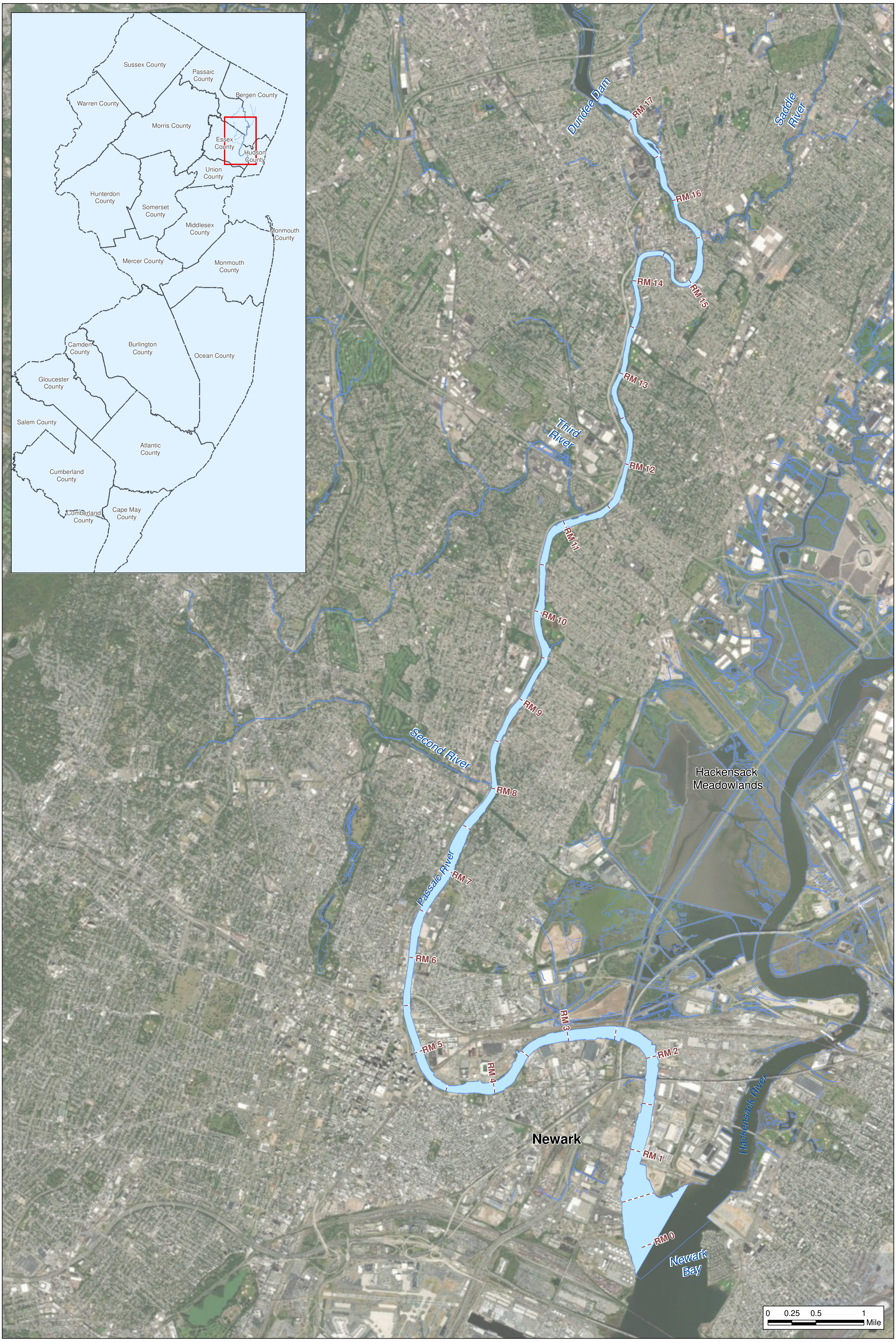
Approximately 88% of the wetlands in the area of the LPR and Newark Bay were lost between 1816 and 1997, a reduction from 24,728 to 2,921 ac (IntraSearch 1999). Of the wetlands within the LPR watershed alone (historically estimated as 7,400 ac), only 84 ac remain, less than 1% of the original wetlands (Peet and Johnson 1996). Most of the marshes lost were either drained with large ditches, blocked with dikes, or filled in order to "reclaim" the lands for development or to control the local mosquito population, and some marshes were used as landfills (Iannuzzi et al. 2002). The rubbish dumped into the wetlands is thought to have contributed to the spread of avian botulism and the subsequent decline of wading birds in the area (Brydon 1968).

In 1858, Dundee Dam and associated locks were constructed at RM 17.4. The dam created an impoundment, called Dundee Lake, just upstream of the dam, and greatly altered the downstream freshwater flows. Reduced freshwater flows resulted in an increase in salinity downstream from the dam, reducing the available habitat for freshwater plants, fish, and invertebrate species. The dam itself blocked the upstream migration of various fish species to spawning habitat.

The LPRSA was first dredged for commercial navigation in 1874 (USACE 2010). In 1884, construction began on a federal navigation channel of varying depth extending from the mouth of the river (RM 0) to the Eighth Street Bridge in Wallington, New Jersey (RM 15.4). The channel was subject to numerous deepening and maintenance dredging activities over its first 50 years of existence. The dredging allowed for commercial shipping and the docking of deeper-draft ships in the lower section of the LPRSA. Between 1874 and 1983, approximately 20 million cy of sediment were dredged from the LPRSA by the US Army Corps of Engineers (USACE) in order to provide for vessel passage (Iannuzzi et al. 2002). No new channel construction was authorized after 1932, but the existing channel was maintained for nearly 50 years (USACE 2010). The navigation channel between RM 0 and RM 1.5 was last dredged in 1983, but the area between RM 2.5 and RM 6.8 has not been dredged (to maintain the shipping channel) since 1949 (Chant et al. 2011). Frequent and intense sediment disturbance from dredging caused significant declines in SAV, as well as perturbations of LPRSA communities that are supported by SAV (Iannuzzi et al. 2002).

2.1.1 Environmental factors

The LPRSA is a large, complex site located within a highly urbanized and developed region of northern New Jersey (Figure 2-1). The LPRSA is a partially mixed estuary with circulation and salinity patterns that are mainly controlled by a dynamic balance between the freshwater flow from upstream and the brackish tidal inflow from Newark Bay.



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Orthophoto: National Agricultural Imagery Program, 7/29/2015, USDA-FSA and ESRI

Figure 2-1. Lower Passaic River Study Area (LPRSA)

Lower Passaic River Study Area Baseline Ecological Risk Assessment

FINAL

Prepared by : mikey 5/30/2019; W:\Projects\06-58-01 Passaic RI\Data\GIS\Maps_and_Analysis\BERA\Revised BERA 2016\5724_LPRSA site_LSM_20131028.mxd

The LPRSA receives freshwater discharges from above Dundee Dam; three tributaries (Saddle River at approximately RM 15.5, Third River at approximately RM 11.2, and Second River at approximately RM 8.1); and to a lesser extent, smaller tributaries, CSOs, SWOs, permitted municipal and industrial discharges, and direct runoff. Groundwater inflow to the LPRSA is small, estimated to make up < 2% of the total freshwater entering the LPRSA over the Dundee Dam (Malcolm Pirnie 2007a).

The LPRSA and its ecological community are influenced by a variety of factors, including chemical contamination, alterations to the salinity regime, turbidity, organic inputs, dissolved oxygen (DO), and invasive and/or non-native species. The chemical history of the LPRSA is discussed in Section 1, and potential ecological risks resulting from exposure to chemical contaminants in the LPRSA are presented in Sections 6 through 12 of this document. Several of the non-chemical factors are known to influence the quality of habitat for aquatic species. Furthermore, a number of these factors are interrelated. For example, fine sediment is more easily resuspended than coarse sediment and therefore contributes to increased turbidity. Fine sediment is also related to increased concentrations of organic carbon (OC) and inorganic nutrients such as nitrogen and phosphorus. The turbid mixing of fine sediment and organic material can cause suspended sediment to aggregate with other particles, form flocculants with dissolved materials, and settle out into a thin and easily resuspended layer of "fluff," which provides a substrate and food source for benthic invertebrates and other aquatic species. The contributing factors of resuspension and high turbidity (also referred to as non-chemical stressors because of their ability to adversely impact the ecology of a system) are present under natural or urban conditions and augmented by anthropogenic activities (e.g., shoreline development, channelization, sanitary sewer overflow [SSO]/CSO discharge and urban runoff) or during extreme weather conditions (e.g., hurricanes and droughts). The biological community in the LPRSA is composed of many species that are present as a result of the conditions there; for example, species that inhabit the upper estuary and fluvial estuary tolerate a wide range of salinity, turbidity, DO, and OC. Anthropogenic alterations to the aquatic environment (e.g., channelization, SSO/CSO discharge) can strengthen these environmental drivers, causing stress to the biological community in excess of what would occur without the influence of urban development.

Seasonal and daily fluctuations in DO and invasive and/or non-native species can also cause stress or otherwise impact the structure of biological communities. DO is related to many other environmental factors (e.g., autotrophic productivity, biological oxygen demand (BOD), salinity, water temperature, presence of metals) that can influence the production and consumption of oxygen by organisms as well as through redox reactions with chemicals (e.g., oxidation of metals). Invasive and/or non-native species can cause shifts in biological communities through the displacement and exclusion of native species. All of these environmental factors are discussed in detail in the following subsections.

2.1.1.1 Salinity

Salinity is a key driver of the environmental setting of the LPRSA, particularly due to the tidal nature of the river. The LPRSA is commonly evaluated based on three general salinity zones: upper estuarine, fluvial estuarine, and tidal freshwater. The salinity zones were developed using information on the movement of the salt wedge, and the evaluation of salinity data collected at the time of sampling. The following three zones were defined:

- Upper estuarine zone – RM 0 to RM 4
- Fluvial estuarine zone – RM 4 to RM 13
- Tidal freshwater zone – RM 13 to RM 17.4

Interstitial and overlying salinity information were used to refine the upper and lower boundaries (i.e., RM 4 and RM 13, respectively) of the fluvial estuarine zone. Interstitial salinities measured in 2009 did not exist below 5 parts per thousand (ppth) below RM 3.95 or above 0.5 ppt above RM 12.43 (Windward 2015a). Overlying surface water salinities measured during low-flow conditions (similar to conditions present prior to 2009 sampling) did not exist below 5 ppth below RM 4.5 (AECOM 2012c) or above 0.5 ppth above RM 12.8 (Windward 2018e). Thus, the fluvial estuarine zone was determined to be from approximately RM 4 to RM 13.

The salinity zones are shown in Figure 2-2.

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0 0.25 0.5 Mile

0 0.25 0.5 0.75 Kilometers

N

- Upper estuarine
- Fluvial estuarine
- Tidal freshwater
- River mile
- Bridge
- Abutment
- Dock
- Mudflat
- Gravel with fines
- LPRSA
- River

Mudflat areas for ecological receptors are those areas where the river bottom slope is ≤ 6° and the depth is ≥ -2 ft MLLW, based on the 2007 CPG bathymetric survey.

Figure 2-2. General salinity zones for the LPRSA
Lower Passaic River Study Area
Baseline Ecological Risk Assessment

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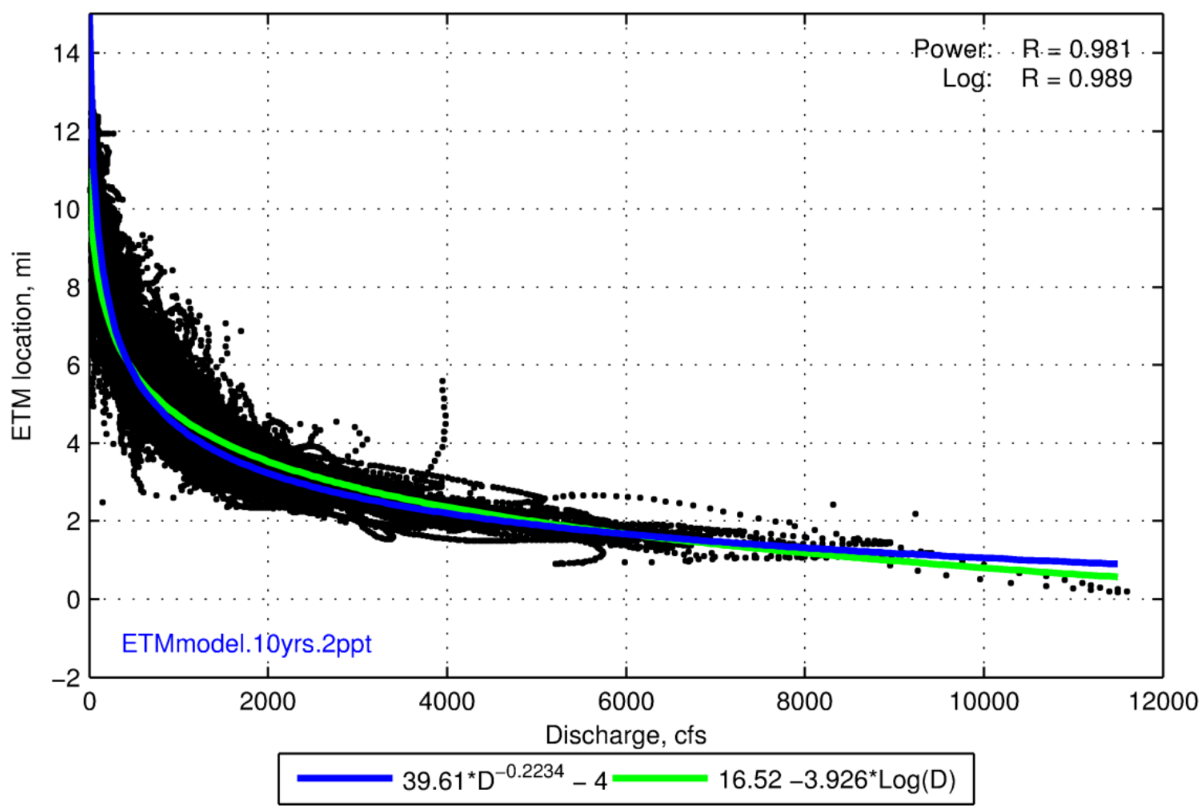
Prepared by mikey 5/30/2019; W:\Projects\06-58-01 Passaic RI\Data\GIS\Maps_and_Analysis\BERA\Revised BERA 2016\5878_Malcolm Pirnie salinity in the LPRSA_LSM_20160624.mxd

The designations for the salinity zones in the LPRSA are qualitative because the location of the interface between fresh and saline waters (also referred to as the "salt wedge")[32] is influenced by freshwater and tidal flows, as well as system geometry. The salt wedge in the LPRSA migrates between 2.5 and 4.5 mi each day (Moffatt & Nichol 2013), typically moving several miles during each tidal cycle (Malcolm Pirnie 2007a; Canizares et al. 2009). The salt wedge can travel as far upstream as RM 14 under extreme low-flow conditions (Sea Engineering and HDR|HydroQual 2011). Figure 2-3 shows the location of the salt front (defined as located at the point of 2 ppth salinity at the bottom of the water column) as a function of river discharge at Dundee Dam. The location of the salt front was computed by Moffat & Nichol (2013) using a hydrodynamic simulation model developed by USEPA Region 2 (HydroQual 2008) with hydrodynamic data from 1995 through 2004.



Source: Moffatt & Nichol (2013)

Figure 2-3. Salt wedge location as a function of discharge at Dundee Dam

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32 The salt wedge is the boundary in an estuary between freshwater and salt water that is formed by the net downstream flow of freshwater. Salt water is denser than freshwater, and therefore remains deeper in the water column as it moves upstream from a river mouth. Freshwater, being less dense, floats above the salt water layer as it moves downstream toward the river mouth. As the two layers mix, a wedge shape is formed in the salt water intrusion (when visualizing the river laterally).

The location of the salt wedge is above RM 5 when discharge at Dundee Dam is below the annual average of 1,300 cubic feet per second (cfs) (based on data collected between 1900 and 2012) (USGS 2014). The salt wedge is pushed further downriver with increasing flows and is located further upriver during low-flow conditions.

The location of the salt wedge typically coincides with the location of the estuarine turbidity maximum (ETM), an area of relatively high suspended sediment concentrations. The ETM is a product of the resuspension of sediment from turbulence created at the front of the tidal current as it pushes the salt wedge upriver beneath the freshwater flowing downriver, and the flocculation of dissolved material as it comes into contact with the salt wedge (Chant et al. 2011; Dyer 1988; Dyer 1997 as cited in Moffatt & Nichol 2013). The ETM migrates up and downstream, both seasonally and daily, due to tidally influenced movement of the salt wedge. The ETM is therefore not a single point in space, but is integrated over several miles, appearing spatially as a turbidity gradient that decreases with distance both upstream and downstream from the salt wedge.

Salinity is the primary influence on benthic community structure in the LPRSA (see Section 2.2 for further discussion). In addition, daily and seasonal variations of salinity in the fluvial estuarine zone can have a significant impact upon biological communities in the LPRSA. The benthic invertebrate community, for example, is influenced by salinity in the interstitial and overlying water, which may vary differently from salinity in the water column. Thus, salinity zones (different from those shown in Figure 2-2) have been developed for the purpose of evaluating the benthic invertebrate community (Section 2.2.1). In addition, some species of fish found in the LPRSA appear to be excluded from certain portions of the LPRSA because of the salinity gradient (e.g., channel catfish [*Ictalurus punctatus*]); those species tolerant of brackish salinities (e.g., white perch [*Morone americana*]) are found throughout much or all of the LPRSA. Salinity tolerances in many species (e.g., American eel [*Anguilla rostrata*]) vary by life stage, such that adults migrate downstream into the estuary, whereas spawning and rearing occurs in freshwater, or vice versa. Therefore, the generalized salinity zones described above are not ecologically relevant for all receptor groups, and the use of receptor-specific zones for the assessment of ecological risk (e.g., benthic invertebrate-specific salinity zones in Section 2.2.1) is warranted.

2.1.1.2 Sediment grain size

Fine sediment and flocculent material are considered non-chemical impacts on habitat characteristics that can influence ecosystems (Relyea et al. 2000, 2012). A relatively high proportion of sediment in urban watersheds is derived from anthropogenic sources, particularly road runoff, stormwater, and sewage (Taylor and Owens 2009; Owens et al. 2011; Owens et al. 2001; Walsh et al. 2005). Urbanization has also been shown to significantly influence the sediment-associated chemical concentrations of fine-grained (< 63 µm) river sediment deposits (Droppo et al. 2002; Walsh et al. 2005; Meharg et al. 2003). Fine-grained surficial deposits are easily eroded from the channel bed and resuspended in the water column (Droppo et al. 2002), so they act as a sink for contamination and as a pathway for chemical transport (as suspended sediment).

Grain size for LPRSA sediments, measured as the percentage of fine-grained sediment mass within each sample, ranges from 0 to 97.5%, and the percentage of sediment composed of gravel ranges from 0 to 37.9% (Appendix K). Figure 2-4 shows the large variability in the fraction of fine-grained sediment in the LPRSA and that, in general, sediment tends to be coarser above RM 13 (i.e., above the influence of the salt wedge and ETM), and finer near the mouth of the LPRSA. Grain sizes within the middle of the LPRSA, where the salt wedge migrates both seasonally and daily, vary substantially.

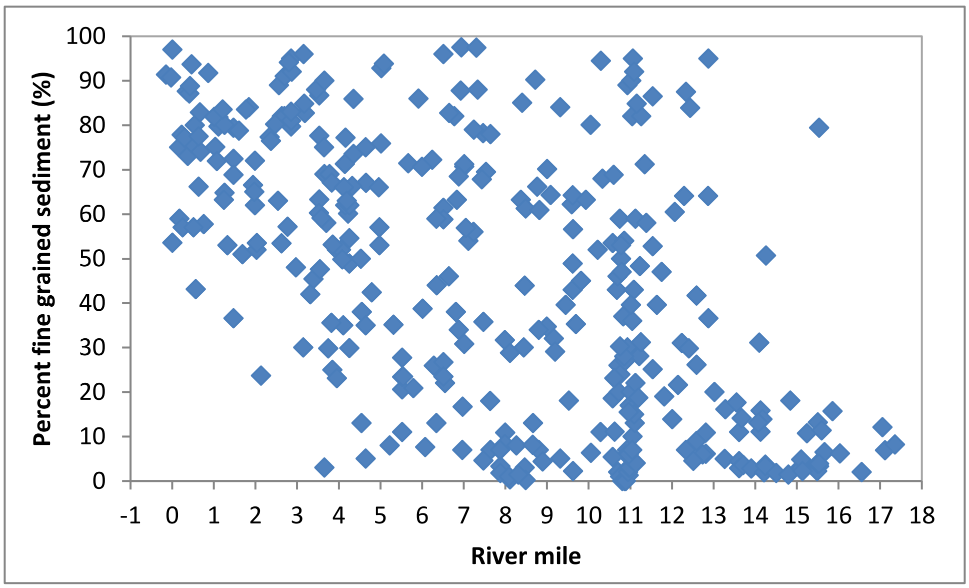

Figure 2-4.Spatial gradient of the fraction of fine-grained sediment in the LPRSA

2.1.1.3 Turbidity

Both natural and anthropogenic inputs of sediments and dissolved organic matter are also sources of turbidity in the LPRSA. These sources include, but are not limited to, soil erosion, urban runoff, SSO/CSO/SWO discharge, river discharge stage, daily tidal exchange, and excessive algal growth.

During recent surface water monitoring events in the LPRSA (AECOM 2012c), total suspended solids (TSS), which is an analogous measure of turbidity, were measured at concentrations above the New Jersey Department of Environmental Protection (NJDEP) surface water quality standard of 40 mg/L applicable to the LPRSA upstream of the confluence with Second River (NJDEP 2008b).[33,34] It is not clear whether TSS becomes sufficiently concentrated to render the LPRSA downstream of Second River unsuitable for its designated uses as defined by NJDEP (2008b).[35]

2.1.1.4 Organic inputs

Sediment profile imaging (SPI) conducted in 2005 (Germano & Associates 2005) indicated that the LPRSA system is highly enriched by organic inputs such as leaf litter, SSO/CSO inputs, and urban runoff. This was confirmed by additional sampling; organic debris (e.g., leaf litter) was observed in many of the recently collected sediment samples (Windward 2014a, c, 2019b). The amount of total organic carbon (TOC) in the sediment directly influences the benthic community structure and function (Pearson and Rosenberg 1978; Borja et al. 2008; Carvalho et al. 2005; Carvalho et al. 2011). Although organic matter is an important food source for benthic organisms, too much organic matter can cause changes in the benthic community structure (affecting species richness and abundance) (Diaz and Rosenberg 1995) through the depletion of oxygen and the buildup of toxic biological waste products, such as ammonia. Previous studies have indicated that TOC in excess of 3.5% may result in significantly decreased benthic diversity (Hyland et al. 2005), and that TOC in excess of 10% can result in "severe effects" (Persaud et al. 1993). The TOC in the LPRSA was found to be as high as 24% (with a mean value of approximately 4%) (Windward 2015a; AECOM 2014a).

2.1.1.5 Nutrient inputs

There are various sources of nutrients (e.g., phosphorus and nitrogen) in urban settings including, but not limited to, urban runoff (Foster and Charlesworth 1996; Owens et al. 2001) and SWO or SSO/CSO discharges (Droppo et al. 2002). Nutrients are quickly taken up by aquatic autotrophs (e.g., algae) or accumulated in sediment or the fluff layer (Section 2.1.1.6). In the LPRSA, sediment concentrations of nitrogen and phosphorus are strongly negatively correlated with sediment grain size (Windward 2015a), indicating that the distribution of nutrients in sediment is closely related to physical factors (e.g., flow, scour and deposition, and the influence of tides and salinity on sediment transport and flocculation).

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[33] The LPRSA upstream of Second River is classified as FW2-NT waters, and downstream of Second River is classified as SE3 waters.

[34] TSS values in water samples collected during 2011 and 2012 LPRSA chemical water column monitoring events ranged from 2.7 to 221 mg/L; 20% of the samples (40 of the 200 water samples) had TSS greater than 40 mg/L.

[35] A TSS criterion for SE3 waters is not clear, in that it stipulates that the water body not be unsuitable for designated uses, which include secondary contact recreation, maintenance and migration of fish populations, migration of diadromous fish, maintenance of wildlife, and any other reasonable uses (N.J.A.C. 7:9B-1.12(f)) (NJDEP 2008b).

Nutrients are known to represent stressors within urbanized rivers and estuaries (Carpenter et al. 1998; Savage et al. 2002). Phosphorus, in particular, has been identified as an aquatic stressor in freshwater portions of the Passaic River (NJDEP 2008a). A total maximum daily load (TMDL) for phosphorus was adopted by NJDEP in 2008 for the freshwater, non-tidal portion of the Passaic River Basin upstream of Dundee Dam. This TMDL was adopted to meet the Surface Water Quality Standards pursuant to the Water Quality Planning Act (N.J.S.A. 58:11A-7) and the Statewide Water Quality Management Planning rules (N.J.A.C. 7:15-6.3(a)), and in compliance with Sections 305(b) and 303(d) of the Clean Water Act (NJDEP 2008a). Excess phosphorus (i.e., concentrations greater than the Surface Water Quality Standards) can lead to excess primary productivity (e.g., algal growth) and associated changes in pH and DO concentrations, which can cause additional stress and adverse effects on the aquatic community.

Phosphorus (and other nutrients) may also contribute to the general environmental stress in the LPRSA, given the urban nature of the study area, the abundant sources of nutrients (e.g., SSO/CSO, non-point source runoff) (Carpenter et al. 1998), and the fact that phosphorus has been identified as a pollutant of concern upstream of the study area. During recent surface water monitoring events (AECOM 2012c), phosphorus was measured in the LPRSA at concentrations above the NJDEP criterion applicable to the LPRSA upstream of the confluence with Second River (NJDEP 2008b)[36] (0.1 mg/L). Phosphorus measured in the LPRSA in 2011 and 2012 ranged from 0.094 to 0.721 mg/L.

2.1.1.6 Fluff layer

A fluff layer consists of unconsolidated sediment that overlies a less erodible (consolidated) bed of sediment. The fluff layer is easily erodible sediment deposited during slack water circumstances and resuspended during flood or ebb tides. The fluff layer includes flocculent material that is prevalent in urban systems (Droppo et al. 2002) and is created when various types of particles in water aggregate (Droppo et al. 1997; Droppo et al. 1998). The aggregation of particles influences the hydrodynamic properties of the particles, in particular the settling velocity and sorption capacity of the composite particles, both of which, in turn, influence the transport and storage of the fluff layer (Droppo 2001; Droppo et al. 1998; Droppo et al. 2002). Inputs of fine particles, coupled with inputs high in organic content (i.e., TOC and sewage) in urban areas, indicate that flocculation processes are likely to be important within urban rivers (Droppo et al. 2002).

Episodic blooms of phytoplankton also contribute to the fluff layer. Chlorophyll-*a* data, which is a surrogate for the measurement of phytoplankton, collected in the

[36] The LPRSA upstream of Second River is classified as FW2-NT waters, and downstream of Second River is classified as SE3 waters. No phosphorus criterion has been set for SE3 waters (NJDEP 2008b).

LPRSA in 2011 and 2012 (ddms 2013a, b, c, d, e) (Figure 2-5) demonstrate that the system undergoes periodic blooms of phytoplankton.

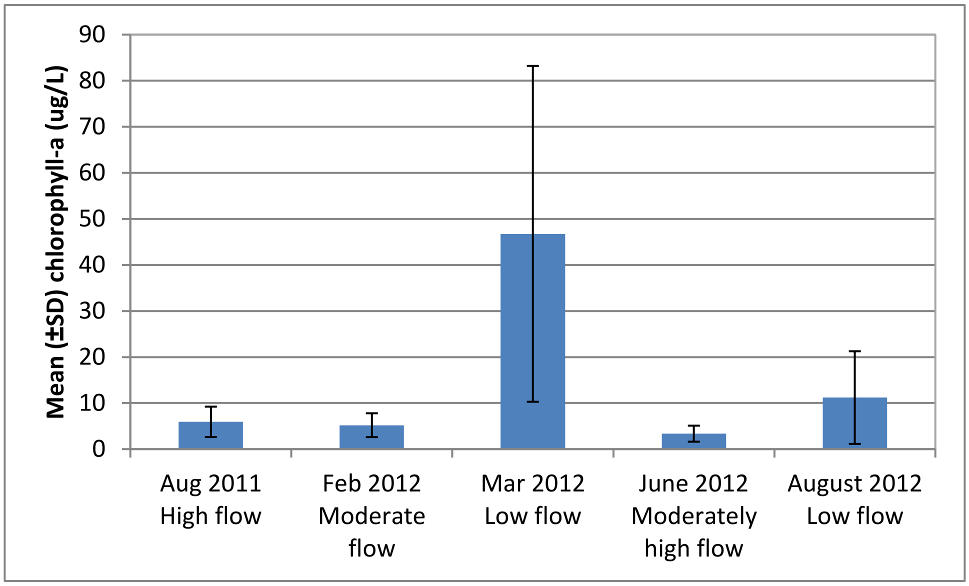
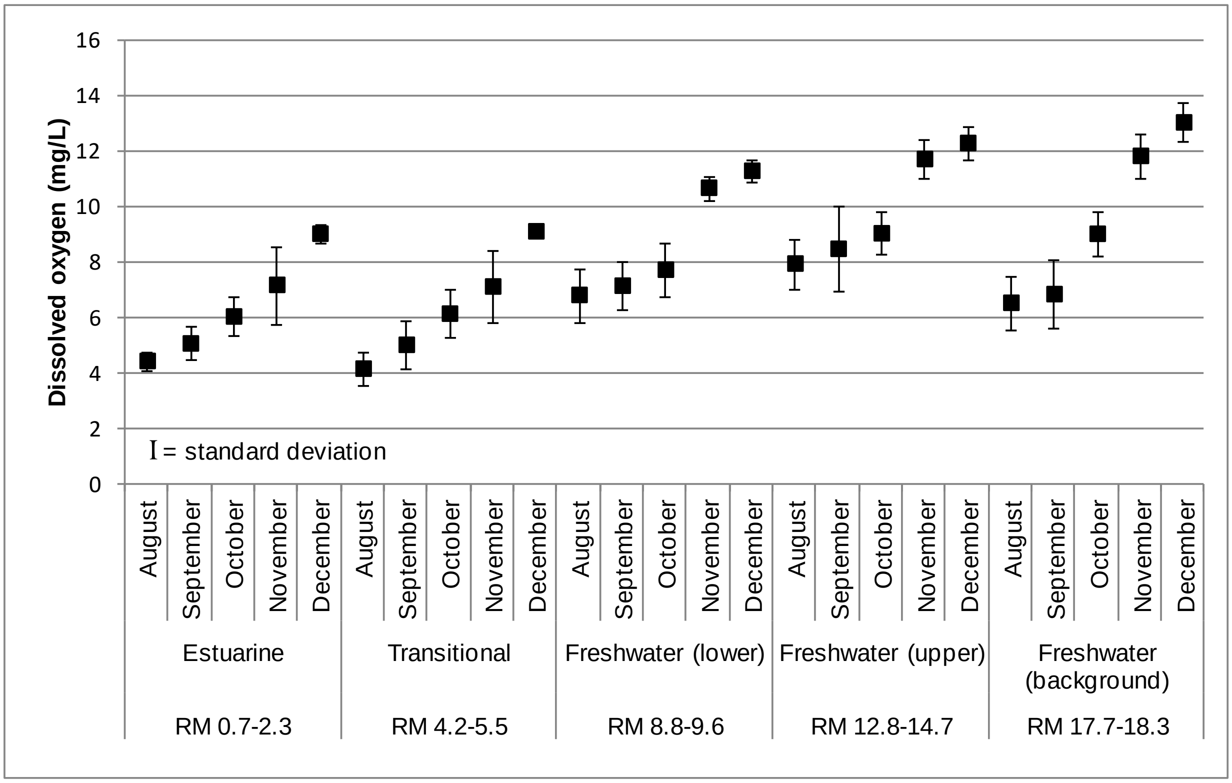


Figure 2-5. Average chlorophyll *a* concentrations measured in the LPRSA during five water sampling events between August 2011 and August 2012

Fine-grained sediment and the surficial fluff layer are important habitat characteristics that influence the benthic invertebrate community in the LPRSA, as well as contaminant uptake in benthic invertebrates and at higher trophic levels. For example, fine-grained sediments provide a substrate in which invertebrates burrow and live (Esselink and Zwarts 1989; Kristensen and Kostka 2004), as well as a food source for deposit feeders, and fluff is also consumed by invertebrates as a food source. In addition to being physically unstable (i.e., easily disturbed benthic invertebrate habitat), fine-grained and fluff layer sediment generally has a high organic content (Droppo et al. 2002), both increasing the sorption of higher organic and inorganic contaminant concentrations (Droppo 2001; Droppo et al. 2002; Droppo et al. 1998; Droppo et al. 2006), and providing a source of nutrients to deposit feeders and detritivores (e.g., filter-feeders), which may increase the chemical exposures of benthic invertebrates. The generation of fluff is often enhanced in urban systems, wherein inorganic and organic materials entering the aquatic system via urban runoff and CSO/SWO inputs provide a substrate for the flocculation of fluff material.

2.1.1.7 Dissolved oxygen

Based on data collected in the LPRSA and above Dundee Dam, DO has, at times, been depressed (Windward 2018e). DO concentrations as low as 1.70 mg/L have been recorded in surface water above Dundee Dam, and DO concentrations as low as 3.25 mg/L have been recorded in the LPRSA. Some historical accounts of DO concentrations are even lower: DOI (1969) reported very low DO (i.e., 0 mg/L at one sampling location in the upper estuary) as well as high BOD and fecal coliform in the upper estuary. These findings were spatially consistent with a large number of industrial and municipal outfalls. More recent average monthly DO concentrations, measured between August and December 2012, are shown in Figure 2-6. The figure shows that DO is lower in the LPRSA during the summer and higher during the fall. Additionally, DO tends to be higher (regardless of season) further upstream.



Source: Windward (2018e)

Figure 2-6. Mean monthly dissolved oxygen concentrations in the Lower Passaic River Study Area and background freshwater area

Concentrations of DO less than 5 mg/L can act as stressors on benthic communities and fish. Hypoxia is a stressful condition that may change the physiology of benthic organisms. Fish and benthic organisms may exhibit behavioral responses, such as avoidance of certain areas, reduced burrowing depths (for benthic invertebrates), metabolic depression, and/or growth reduction (Diaz and Rosenberg 1995; Riedel et al. 1997; Villnäs et al. 2012; Riedel et al. 2008; Vaquer-Sunyer and Duarte 2008). Chemical concentrations at elevated levels can also cause avoidance and other behavioral changes (Oakden et al. 1984; Keilty et al. 1988).

Physical, chemical, and biological processes may influence DO. Nutrient loading, seasonal temperature fluctuations, and algae/macrophyte communities, as well as salinity, also affect DO. Temperature and salinity are among the physical factors known to have an effect. The solubility of oxygen in water decreases as temperature and salinity increase. Oxygen is consumed by organisms in the system (e.g., bacteria, benthic invertebrates, and fish) according to the BOD and through redox reactions with certain chemicals (e.g., reduced metal species) according to chemical oxygen demand. If the biological and chemical oxygen demand in the system are greater than the amount of oxygen supplied (through autotrophic productivity or physical mixing), then hypoxia or anoxia can occur.

Periodically depressed DO concentrations in the LPRSA may have been the result of several biotic (e.g., BOD) and abiotic (e.g., temperature and salinity) factors. Available data do not include BOD, so the relationship between BOD and DO cannot be determined at present. Higher salinity and lower DO concentrations were observed at monitoring locations in estuarine waters than at locations in freshwater areas (Windward 2018e). However, salt water has a lower saturation level for DO than does freshwater. A tide-related drop in DO from saturated freshwater to saturated salt water is not an ecological concern, as organisms that live in the transition zone are adapted to such changes.

2.1.1.8 Invasive and/or non-native species

The presence of invasive and/or non-native species has impacted both the physical characteristics and the biological community of the LPRSA. The primary concern regarding the introduction or invasion of non-native species is that native species may not be able to compete successfully for necessary resources with adaptable, non-native species (Carey and Wahl 2010). The result can be localized paucity or extinction of sensitive species (Colnar and Landis 2007; Buhle and Ruesink 2009; Jarv et al. 2011; Miller et al. 2010).

Riparian vegetation in the LPRSA includes both native and non-native plant species; only 20 to 57% of herbaceous plant species and 60 to 80% of shrubs observed along the LPRSA during the 2007 and 2008 vegetation surveys were native species (USACE et al. 2008). Invasive species in the LPRSA, such as purple loosestrife (*Lythrum salicaria*) and Japanese knotweed (*Fallopia japonica,* syn. *Polygonum cuspidatum*), can displace native plants. Invasive species have become widely distributed due to the lack of natural predators and diseases that kept them in check in their original habitat, which allows invasive species to grow and persist at very high rates and densities in their new environment (Van Clef 2009).

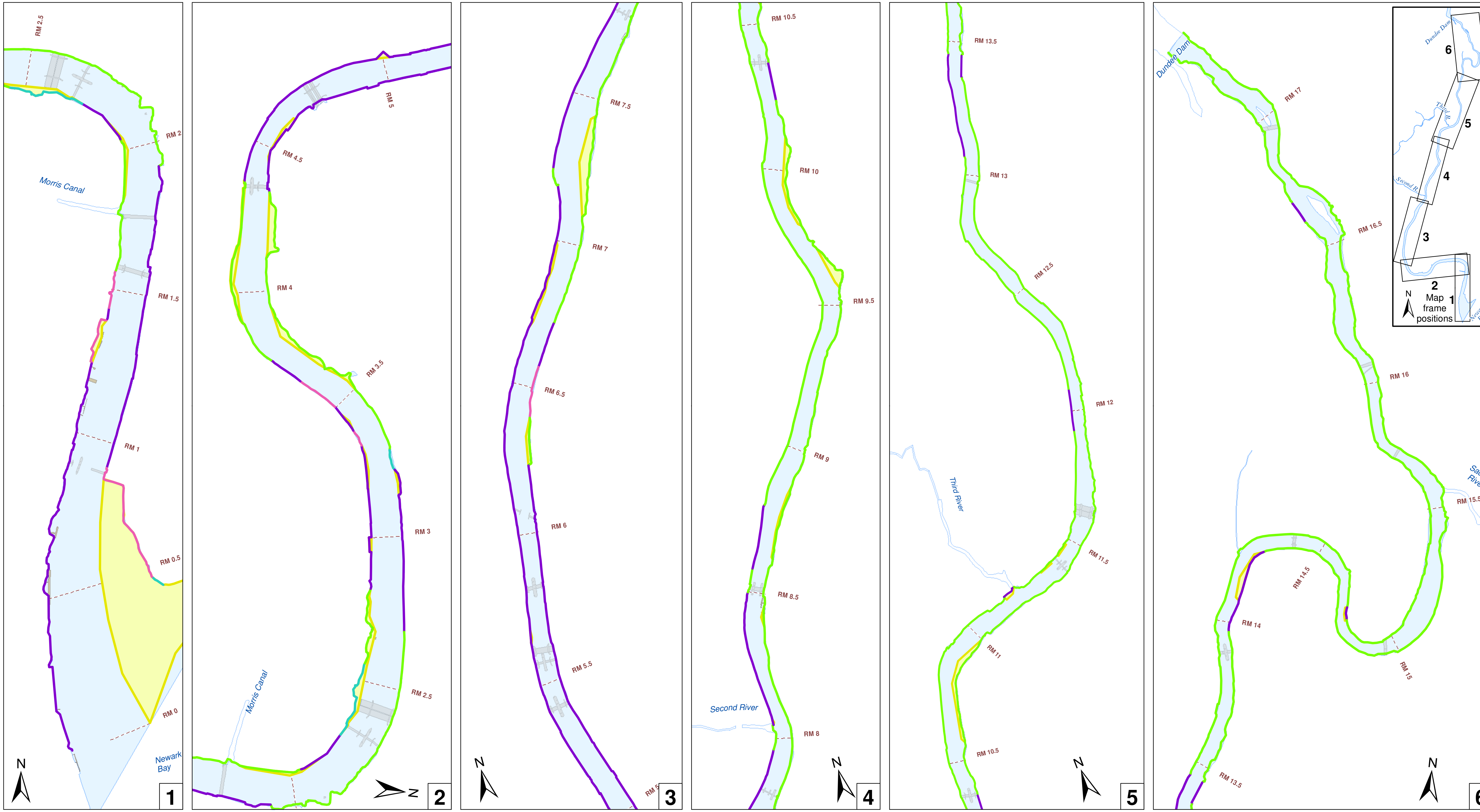
Several non-native species have been intentionally introduced to the LPRSA as game fish or to support game fish in New Jersey (Van Clef 2009). One non-native fish species with the potential to impact the LPRSA system is the common carp (*Cyprinus carpio*), which has adapted to the conditions observed in the LPRSA. It should also be noted that catfish, through their behavior and use of bedded sediment as habitat, can also disturb sediment, as can other benthic feeding native species such as suckers. In the LPRSA, common carp accumulate substantial mass and are widely distributed (from approximately RM 5.5 to RM 17.4, as well as above Dundee Dam and within the tributaries) (Windward 2010j, c, 2014b, 2012b; Do 2013). The common carp has been linked to observable adverse effects on aquatic habitats and the sustainability of those habitats for both aquatic and terrestrial wildlife (Kloskowski 2011; Kloskowski et al. 2010; Bajer et al. 2009; Wahl et al. 2011; Roozen et al. 2007; Miller and Crowl 2006). The degree to which carp affect habitat conditions or other species in the LPRSA is unknown, but the potential exists for carp to cause localized stress of fish, invertebrates, and aquatic vegetation (e.g., due to temporarily increased turbidity, physical disturbance of sediments, or competition).

2.1.2 Habitat

As described, the abundance of complex and functional ecological habitats is limited in the LPRSA. Degraded habitat may adversely affect the health, abundance, diversity, and reproductive success of biological populations. General habitat areas have been identified in several habitat and vegetation surveys that have been conducted in the LPRSA since 1999 (Iannuzzi and Ludwig 2004; USACE et al. 2008; Windward 2014b).

2.1.2.1 General LPRSA habitat

Figure 2-7 provides a general description of the types of habitat present along the LPRSA shoreline; general shoreline habitat within the LPRSA is categorized based on sediment grain size, bathymetry (i.e., mudflats), and the type of riparian vegetation (aquatic or mixed vegetation) or man-made structures (i.e., riprap and bulkhead) along the banks of the LPRSA (i.e., as far as 100 m from either bank). Currently, most (approximately 70%) of the riverbank along the lower portion of the LPRSA (approximately RM 7 and below) consists of bulkhead and/or riprap and supports a limited amount of vegetation (Windward 2014b). The upper portion of the LPRSA riverbank (above RM 8) is dominated by mixed vegetation, generally over steep banks. Natural habitat areas along the shoreline, including wetland and mudflat habitats, are limited to small patches or isolated areas. Available mudflats provide key foraging habitat for shorebirds, and the nearshore shallow areas provide key foraging areas for small forage fish (SFF) and other prey species. Avian use of the LPRSA is limited by habitat availability, as observed in recent avian community surveys (Windward 2019e, 2011a), and as reported by Ludwig et al. (2010). Table 2-1 and Figure 2-8 provide details as to the locations, dimensions, and grain sizes of LPRSA mudflats.



0 0.25 0.5 Miles

0 0.25 0.5 0.75 Kilometer

Shoreline classification
- Aquatic vegetation
- Mixed vegetation
- Bulkhead
- Riprap
- Exposed mudflat observed
- River mile
- Bridge
- Abutment
- Dock
- LPRSA

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Figure 2-7. LPRSA shoreline habitat classifications

Lower Passaic River Study Area Baseline Ecological Risk Assessment

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Table 2-1. Description of LPRSA mudflats

Mudflat No.	Bank Direction[a]	River Mile		Bank Type	Shoreline Vegetation	Mudflat Dimensions			Sediment Grain Size[b]					
		Start	End			Area (ac)	Length (ft)	Mean Width (ft)	Silt	Silt/ Sand	Sand	Gravel/ Sand	Rock/ Coarse Gravel	Unknown[c]
MF-01	left	0.0	0.8	riprap and mud and rock	none, mixed forest, and emergent vegetation	149	4,608	841	-	1.0	-	-	-	-
MF-02	right	0.5	0.9	bulkhead	none	2.1	2,340	36	0.65	-	-	0.04	0.22	0.1
MF-03	left	0.8	1.5	bulkhead and riprap	none	4.7	3,542	58	0.97	0.02	-	-	-	0.01
MF-04	right	0.9	1.5	bulkhead and riprap	none and emergent vegetation	4.3	3,383	54	0.68	-	-	-	0.23	0.08
MF-05	right	1.5	3.6	bulkhead, riprap, mud and rock	none, shrub-scrub, and emergent vegetation	31	11,117	134	0.90	-	-	0.02	0.04	0.04
MF-06	left	1.5	1.7	bulkhead	none	0.15	1,044	9	0.88	-	-	-	-	0.12
MF-07	left	1.7	2.0	bulkhead and mud and rock	none and shrub-scrub	1.3	1,364	40	0.30	0.45	-	-	-	0.24
MF-08	left	2.0	2.7	armored and mud and rock	shrub-scrub	3.4	3,830	35	0.48	0.45	-	-	-	0.08
MF-09	left	2.7	3.1	bulkhead	none	0.16	1,922	4	0.64	-	-	-	-	0.36
MF-10	left	3.1	4.3	mud and rock	mixed forest; emergent vegetation	14	6,437	102	0.72	0.17	-	0.03	-	0.08
MF-11	right	3.7	3.8	armored and mud and rock	mixed forest, shrub-scrub	0.27	556	22	-	-	-	0.72	-	0.28
MF-12	right	3.8	4.2	mud and rock	mixed forest, shrub-scrub	2.3	1,700	53	-	-	-	0.79	-	0.21
MF-13	right	4.2	4.3	mud and rock	mixed forest, shrub-scrub	0.06	724	4	-	-	-	0.96	-	0.04

Table 2-1. Description of LPRSA mudflats

Mudflat No.	Bank Direction[a]	River Mile		Bank Type	Shoreline Vegetation	Mudflat Dimensions			Sediment Grain Size[b]					
		Start	End			Area (ac)	Length (ft)	Mean Width (ft)	Silt	Silt/ Sand	Sand	Gravel/ Sand	Rock/ Coarse Gravel	Unknown[c]
MF-14	left	4.3	4.9	bulkhead	none	4.4	3,259	64	0.12	0.78	-	-	-	0.1
MF-15	right	4.6	4.8	bulkhead	none	0.03	1,131	3	-	-	-	0.42	-	0.58
MF-16	right	4.8	5.3	bulkhead	none and shrub-scrub	1.1	2,714	18	-	0.38	-	0.03	-	0.59
MF-17	left	4.9	5.3	bulkhead	none and shrub-scrub	0.37	2,068	9	0.01	0.41	-	-	0.12	0.47
MF-18	right	5.4	5.5	bulkhead	none	0.10	315	14	-	0.74	-	-	-	0.26
MF-19	left	5.4	5.5	bulkhead	none	0.17	696	13	-	0.40	-	-	-	0.60
MF-20	right	5.6	5.8	bulkhead	none and mixed forest, shrub-scrub	0.31	1,348	12	-	0.51	-	0.14	0.1	0.25
MF-21	left	5.6	5.7	bulkhead	none	0.04	603	6	-	0.05	-	-	0.48	0.47
MF-22	left	5.8	6.1	bulkhead	none	0.46	1,351	16	0.51	-	-	0.3		0.2
MF-23	right	6.0	6.1	bulkhead	none	0.06	770	5	0.01	-	-	0.14	0.03	0.82
MF-24	left	6.1	6.6	bulkhead, riprap, mud and rock	none and mixed forest, shrub-scrub	2.9	2,928	49	0.68	-	-	0.02	0.1	0.19
MF-25	right	6.3	7.1	bulkhead	mixed forest, shrub-scrub	4.6	4,520	46	0.73	-	-	0.23	-	0.03
MF-26	left	6.7	7.8	mud and rock	mixed forest, shrub-scrub	14	5,751	109	0.77	-	-	0.04	0.11	0.08
MF-27	right	7.2	7.8	bulkhead	mixed forest, shrub-scrub	3.8	3,121	52	0.01	0.88	-	-	-	0.11

Table 2-1. Description of LPRSA mudflats

Mudflat No.	Bank Direction[a]	River Mile		Bank Type	Shoreline Vegetation	Mudflat Dimensions			Sediment Grain Size[b]					
		Start	End			Area (ac)	Length (ft)	Mean Width (ft)	Silt	Silt/ Sand	Sand	Gravel/ Sand	Rock/ Coarse Gravel	Unknown[c]
MF-28	right	7.8	8.2	bulkhead	shrub-scrub	2.2	2,446	44	-	0.33	-	0.32	0.33	0.01
MF-29	left	7.8	8.2	mud and rock	mixed forest, shrub-scrub	0.44	2,160	10	-	-	-	-	0.1	0.9
MF-30	left	8.2	8.5	mud and rock	mixed forest, shrub-scrub	1.4	1,501	43	0.38	-	-	-	0.26	0.36
MF-31	right	8.3	8.5	bulkhead	none	0.28	1,065	12	-	-	-	-	0.75	0.25
MF-32	right	8.5	8.6	bulkhead and armored	mixed forest, shrub-scrub	0.10	640	7	-	-	-	-	0.49	0.51
MF-33	left	8.5	9.2	armored and mud and rock	mixed forest, shrub-scrub, emergent vegetation	3.1	3,538	38	0.47	-	0.01	0.28	0.21	0.03
MF-34	right	8.7	9.2	armored	mixed forest, shrub-scrub	1.3	2,619	23	-	-	-	0.98	-	0.02
MF-35	right	9.2	9.6	armored and mud and rock	mixed forest, shrub-scrub	1.4	2,285	29	0.49	-	-	0.36	-	0.15
MF-36	left	9.2	9.4	armored and mud and rock	mixed forest, shrub-scrub	0.46	1,124	18	-	-	-	0.96	-	0.04
MF-37	left	9.4	9.7	mud and rock	mixed forest, shrub-scrub	3.2	1,641	81	0.57	-	-	0.33	-	0.10
MF-38	right	9.6	9.8	armored	mixed forest, shrub-scrub	0.17	1,158	7	-	-	-	0.47	-	0.53
MF-39	left	9.7	10.1	mud and rock	mixed forest, shrub-scrub	4.2	2,366	80	0.99	-	-	-	-	0.01

Table 2-1. Description of LPRSA mudflats

Mudflat No.	Bank Direction[a]	River Mile		Bank Type	Shoreline Vegetation	Mudflat Dimensions			Sediment Grain Size[b]					
		Start	End			Area (ac)	Length (ft)	Mean Width (ft)	Silt	Silt/ Sand	Sand	Gravel/ Sand	Rock/ Coarse Gravel	Unknown[c]
MF-40	right	9.8	10.3	armored and mud and rock	mixed forest, shrub-scrub	1.0	2,373	19	0.44	-	-	0.25	0.27	0.03
MF-41	left	10.1	10.3	bulkhead	none	0.14	932	6	-	-	-	0.77	-	0.23
MF-42	right	10.3	10.8	armored and bulkhead	mixed forest, shrub-scrub	1.2	2,595	20	-	-	-	0.78	0.14	0.08
MF-43	left	10.3	10.5	armored	mixed forest, shrub-scrub	0.28	814	17	-	-	-	0.45	-	0.55
MF-44	left	10.5	11.1	mud and rock	mixed forest, shrub-scrub, emergent vegetation	6.9	3,225	97	0.59	-	0.37	0.02	-	0.01
MF-45	right	10.8	11.1	armored and mud and rock	mixed forest, shrub-scrub	1.5	1,407	42	-	-	-	0.98	-	0.02
MF-46	right	11.1	11.8	bulkhead and mud and rock	mixed forest, shrub-scrub	5.2	4,029	59	-	0.64	-	0.19	0.13	0.04
MF-47	left	11.1	11.4	armored and mud and rock	mixed forest, shrub-scrub	0.55	1,829	13	-	-	-	0.03	0.84	0.13
MF-48	left	11.4	11.6	armored and mud and rock	mixed forest, shrub-scrub	0.82	1,076	31	-	-	-	0.65	0.35	0.00
MF-49	left	11.6	12.1	armored and mud and rock	mixed forest, shrub-scrub	1.2	2,594	21	-	-	-	0.91	-	0.09
MF-50	right	11.8	12.1	bulkhead	none	0.15	1,505	6	-	-	-	-	0.60	0.40
MF-51	right	12.1	12.6	mud and rock	mixed forest, shrub-scrub	1.6	2,767	26	0.01	-	-	0.02	0.80	0.17

Table 2-1. Description of LPRSA mudflats

Mudflat No.	Bank Direction[a]	River Mile		Bank Type	Shoreline Vegetation	Mudflat Dimensions			Sediment Grain Size[b]					
		Start	End			Area (ac)	Length (ft)	Mean Width (ft)	Silt	Silt/ Sand	Sand	Gravel/ Sand	Rock/ Coarse Gravel	Unknown[c]
MF-52	left	12.2	12.5	armored and mud and rock	mixed forest	0.37	1,370	12	-	-	-	1.0	-	-
MF-53	left	12.5	12.9	armored and mud and rock	mixed forest, shrub-scrub	1.2	2,440	23	-	-	-	0.97	-	0.03
MF-54	right	12.6	13.0	armored and mud and rock	mixed forest, shrub-scrub	0.22	1,610	9	-	-	-	0.62	0.22	0.16
MF-55	right	13.0	13.1	mud and rock	mixed forest, shrub-scrub	0.19	751	12	-	-	-	0.99	-	0.01
MF-56	left	13.0	13.1	mud and rock	mixed forest, shrub-scrub	0.24	649	15	-	-	-	0.96	-	0.04
MF-57	left	13.2	13.2	mud and rock	mixed forest	0.05	368	7	-	0.02	-	0.98	-	-
MF-58	right	13.3	13.7	bulkhead and mud and rock	mixed forest, shrub-scrub	0.58	1,814	16	-	-	-	0.79	-	0.21
MF-59	left	13.3	13.3	mud and rock	mixed forest	0.08	190	20	-	0.23	-	0.76	-	0.01
MF-60	left	13.4	13.6	armored	mixed forest, shrub-scrub	0.39	1,055	16	-	-	-	0.84	-	0.16
MF-61	left	13.6	14.0	armored	mixed forest, shrub-scrub	0.25	1,839	7	-	0.01	-	0.87	-	0.11
MF-62	right	13.8	13.9	armored	mixed forest, shrub-scrub	0.04	644	4	-	-	-	-	-	1.0
MF-63	right	14.0	14.1	concrete embankment	mixed forest, shrub-scrub	0.01	875	1	-	-	-	0.89	-	0.11
MF-64	left	14.0	14.6	bulkhead and mud and rock	mixed forest, shrub-scrub	4.2	3,251	62	-	0.79	-	-	0.2	0.01
MF-65	right	14.2	14.2	concrete embankment	mixed forest, shrub-scrub	0.08	218	28	-	-	-	0.06	0.94	-

Table 2-1. Description of LPRSA mudflats

Mudflat No.	Bank Direction[a]	River Mile		Bank Type	Shoreline Vegetation	Mudflat Dimensions			Sediment Grain Size[b]					
		Start	End			Area (ac)	Length (ft)	Mean Width (ft)	Silt	Silt/ Sand	Sand	Gravel/ Sand	Rock/ Coarse Gravel	Unknown[c]
MF-66	right	14.2	14.4	armored	mixed forest, shrub-scrub	0.08	1,015	3	-	-	-	-	0.94	0.06
MF-67	right	14.5	14.6	armored	mixed forest, shrub-scrub	0.05	665	7	-	-	-	-	0.93	0.07
MF-68	right	14.6	14.9	bulkhead	shrub-scrub	0.40	1,295	15	-	-	0.24	-	0.76	-
MF-69	left	14.8	14.9	armored	mixed forest, shrub-scrub	0.04	477	30	-	-	-	-	0.87	0.13
MF-70	right	14.9	15.0	armored	mixed forest, shrub-scrub	0.55	762	45	-	-	-	0.89	0.06	0.05
MF-71	left	15.0	15.3	mud and rock	mixed forest, shrub-scrub	7.5	1,667	189	-	-	-	0.81	0.14	0.06
MF-72	right	15.1	15.1	armored	mixed forest, shrub-scrub	0.13	311	17	-	-	-	0.16	0.54	0.30
MF-73	right	15.3	15.7	armored and mud and rock	mixed forest, shrub-scrub	3.3	2,187	69	-	-	-	0.63	0.16	0.21
MF-74	left	15.3	15.5	armored and mud and rock	mixed forest, shrub-scrub	3.0	1,033	134	-	-	-	0.13	0.80	0.07
MF-75	left	15.5	15.7	armored	mixed forest, shrub-scrub	0.51	978	124	-	-	-	0.28	0.60	0.11
MF-76	right	15.7	15.8	mud and rock and armored	mixed forest, shrub-scrub	0.41	616	29	-	-	-	0.51	-	0.49
MF-77	left	15.7	15.8	mud and rock and armored	mixed forest, shrub-scrub	0.76	639	59	-	-	-	0.53	0.42	0.04
MF-78	river wide	15.8	16.0	mud and rock and armored	mixed forest, shrub-scrub	5.6	1,322	182	-	-	-	0.99	-	0.01

Table 2-1. Description of LPRSA mudflats

Mudflat No.	Bank Direction[a]	River Mile		Bank Type	Shoreline Vegetation	Mudflat Dimensions			Sediment Grain Size[b]					
		Start	End			Area (ac)	Length (ft)	Mean Width (ft)	Silt	Silt/ Sand	Sand	Gravel/ Sand	Rock/ Coarse Gravel	Unknown[c]
MF-79	river wide	16.0	17.1	mud and rock; some bulkheads	Sheltering forest; shrub-scrub	37	5,774	294	-	-	-	0.01	-	0.99
MF-80	left	16.5	16.7	mud and rock	Sheltering forest; shrub-scrub	2.6	1,159	100	-	-	-	-	-	1.0
MF-81	river wide	17.1	17.4	mud and rock; some bulkheads	Sheltering forest; shrub-scrub	11	1,461	335	-	-	-	-	-	1.0

Note: Mudflat areas for ecological receptors are those areas where the river bottom slope is ≤ 6° and the depth is ≥ -2 ft MLLW. In the event that mudflats were dredged after the collection of sediment for chemical analyses, those sediment chemistry data were omitted from the calculation of EPCs because the sediment chemistry data is no longer representative of site conditions.

a Bank direction assumes that the observer is facing downstream.

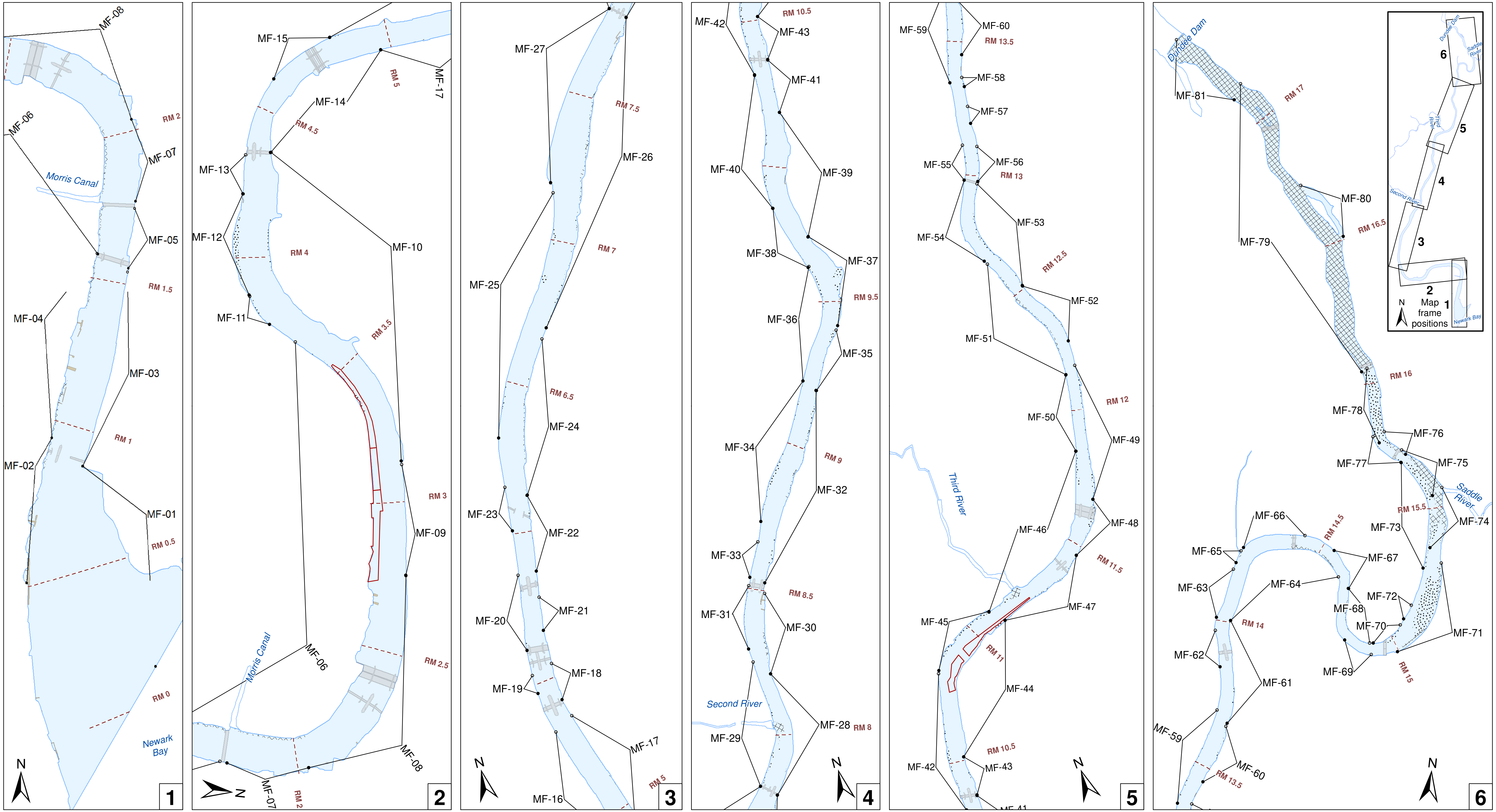
b Grain size values reported as a fraction of the total mudflat area. Grain size data based on Aqua Survey (2006) geophysical survey.

c Any portions of mudflats that were not characterized by the geophysical survey (Aqua Survey 2006) have been categorized as "unknown."

EPC – exposure point concentration

LPRSA – Lower Passaic River Study Area

MLLW – mean lower low water



0 0.25 0.5 0.75 Kilometers

0 0.25 0.5 Miles

Mudflat areas for ecological receptors are those areas where the river bottom slope is ≤ 6° and the depth is ≥ -2 ft MLLW, based on the 2007 CPG bathymetric survey. Grain size is based on the 2005 Geophysical Survey. Kearney Point and areas above RM 16.1 were not surveyed; grain size for these areas is based on field experience.
[a]Grain size values shown as a surface weighted average (SWA) of fractions of the total mudflat area.

Grain size SWA[a]
- Rock and coarse gravel
- Gravel and sand
- Sand
- Sand and silt
- Silt
- Unknown

Mudflat area with coarse sediment
- Rock and coarse gravel
- Gravel and sand

Mudflat extent
- Mudflat upper end
- Mudflat lower end

- Dredge zone
- River mile
- Bridge
- Abutment
- Dock
- LPRSA

Figure 2-8. LPSRA mudflats

Lower Passaic River Study Area Baseline Ecological Risk Assessment

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2.1.2.2 Riparian and aquatic vegetation

Riparian vegetation along the LPRSA is limited due to urbanization in the watershed. Development of the uplands downstream of RM 8 is extensive, and little riparian habitat was observed during a complete habitat survey of the LPRSA (Windward 2014b). Figure 2-9 shows the general plant communities found along the LPRSA shoreline.

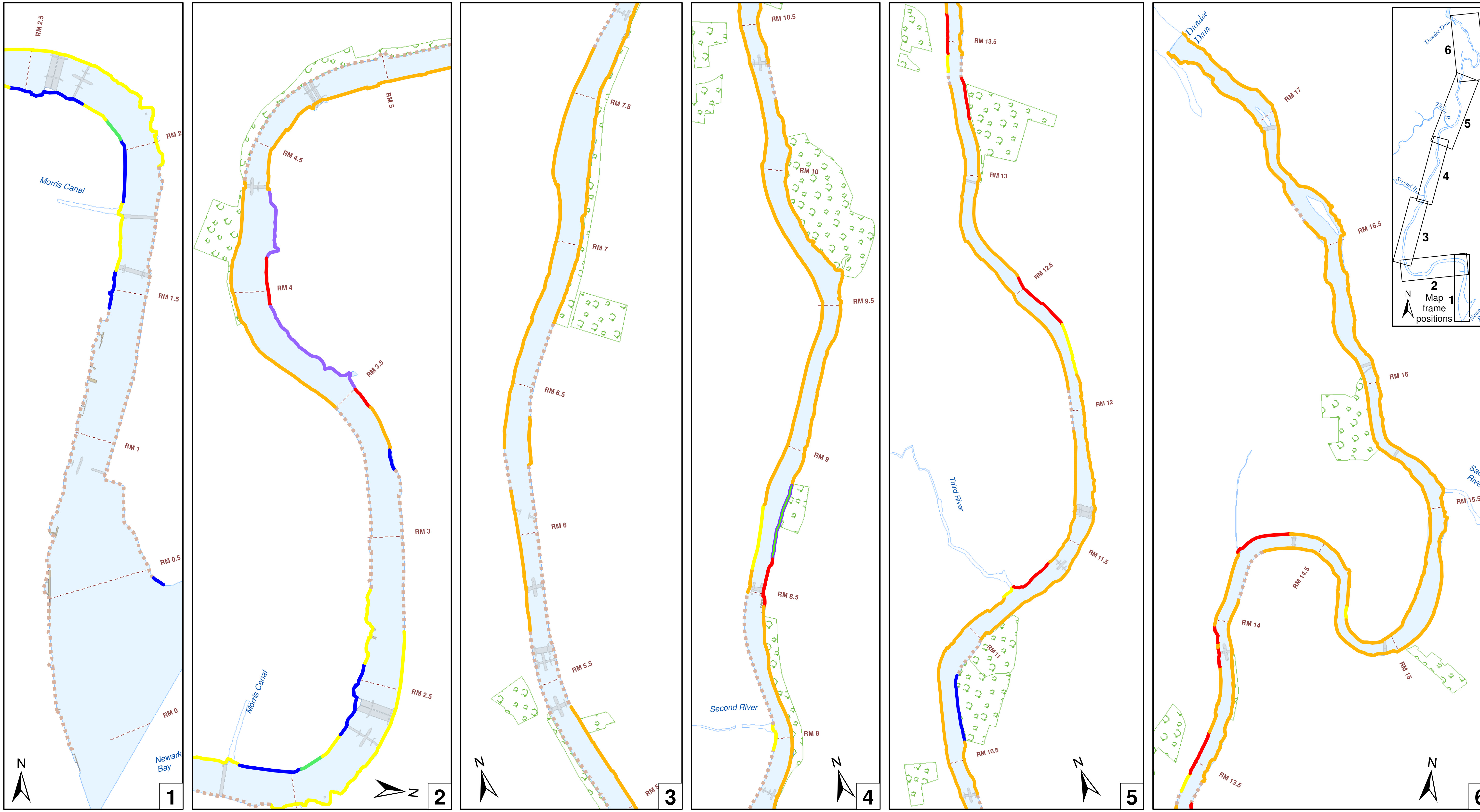


Figure 2-9. LPRSA shoreline plant communities

Lower Passaic River Study Area
Baseline Ecological Risk Assessment

FINAL

Prepared by mikey 5/30/2019; W:\Projects\06-58-01 Passaic RI\Data\GIS\Maps_and_Analysis\BERA\Revised BERA 2016\4384_Shoreline vegetation types_LSM_20161003.mxd

In the lower portions of the LPRSA, where most of the shoreline has been industrially developed, the plant community is less diverse, comprised mostly of scrub-shrub vegetation, such as groundsel tree (*Baccharis halimifolia)*, frequently intermixed with individual or small stands of trees (Windward 2014b). In general, tree species present are representative of disturbed conditions and consist primarily of tree-of-heaven (*Ailanthus altissima)*, eastern cottonwood (*Populus deltoides)*, Norway maple (*Acer platanoides*), locust (*Gleditsia* spp. or *Robinia* spp.), and catalpa (*Catalpa speciosa*). Sites dominated by emergent vegetation are primarily below RM 3.5 and are associated with intertidal mudflats. In general, these areas are dominated by smooth cordgrass (*Spartina alterniflora)* or common reed *(Phragmites australis)*. Japanese knotweed is also a dominant emergent species in the lower portion of the river (as well as throughout the remainder of the LPRSA).

Further upriver, where the shoreline is flanked by wider urban green spaces and parks, mixed forest is more prevalent and diverse with the addition of elm (*Ulmus* spp.), sycamore (*Platanus occidentalis*), ash (*Fraxinus* spp.*)*, and willow (*Salix* spp.) trees; the emergent plant community also includes *Amaranthus* spp., purple loosestrife, and goldenrod (*Solidago* spp.), among others. Although there are fewer sites with emergent vegetation upriver, a mature canopy with overhanging vegetation and large woody debris is more prevalent along the shoreline above RM 10. From RM 16.5 to Dundee Dam (at RM 17.4) exists a large floodplain, consisting mostly of silver maple (*Acer saccharinum*).

Table 2-2 identifies the commonly observed plants in the LPRSA (Windward 2014b). Plants observed along the LPRSA shoreline include a mix of native and non-native plants.

Table 2-2. Common plant species identified in the LPRSA

Common Name	Scientific Name	Status[a]
American elm	*Ulmus americana*	native
Aster (unidentified)	*Aster* spp.	
Black locust	*Robinia pseudoacacia*	native (naturalized from southeast United States)
Boneset (unidentified)	*Eupatorium* spp.	
Box elder	*Acer negundo*	native
Catalpa	*Catalpa speciosa*	native
Cattail (unidentified)	*Typha* spp.	
Common reed	*Phragmites australis*	native
Eastern cottonwood	*Populus deltoides*	native
Elm (unidentified)	*Ulmus* spp.	
Goldenrod (unidentified)	*Solidago* spp.	
Groundsel tree[b]	*Baccharis halimifolia*	native
Hickory (unidentified)	*Carya* spp.	

Table 2-2. Common plant species identified in the LPRSA

Common Name	Scientific Name	Status[a]
Honeysuckle (unidentified)	*Lonicera* spp.	
Horseweed (unidentified)	*Conyza* spp.	
Japanese knotweed	*Fallopia japonica*, syn. *Polygonum cuspidatum*	non-native
Jewelweed (unidentified)	*Impatiens* spp.	
Locust (unidentified)	*Gleditsia* spp. or *Robinia* spp.	
Maple (unidentified)	*Acer* spp.	
Mimosa (unidentified)	*Mimosa* spp.	
Mugwort (unidentified)	*Artemisia* spp.	
Mulberry[b]	*Morus* spp.	
Multiflora rose	*Rosa multiflora*	non-native
Northern red oak	*Quercus rubra*	native
Norway maple	*Acer platanoides*	non-native
Oak (unidentified)	*Quercus* spp.	
Pigweed/amaranth	*Amaranthus* spp.	
Poison ivy	*Toxicodendron radicans*	native
Pokeweed	*Phytolacca americana*	native
Princess tree	*Paulownia tomentosa*	non-native
Purple loosestrife	*Lythrum salicaria*	non-native
Ragweed (unidentified)	*Ambrosia* spp.	
Ragwort	*Jacobaea vulgaris*, syn. *Senecio jacobaea*	non-native
Red osier dogwood	*Cornus sericea*	native
Silk tree	*Albizia julibrissin*	non-native
Silver maple[b]	*Acer saccharinum*	native
Smooth cordgrass	*Spartina alterniflora*	native
Snakeroot (unidentified)	*Ageratina* spp.	
Sycamore	*Platanus occidentalis*	native
Tree-of-heaven	*Ailanthus altissima*	non-native
Virginia creeper	*Parthenocissus quinquefolia*	native
Weeping willow	*Salix babylonica*	non-native
White ash	*Fraxinus americana*	native
Willow (unidentified)	*Salix* spp.	

[a] Native/non-native status is provided only for plants identified to the species level, when available (NRCS 2010).

[b] Species identified by USEPA oversight personnel.

LPRSA – Lower Passaic River Study Area

USEPA – US Environmental Protection Agency

The LPRSA has limited aquatic vegetation. During the 2010 habitat survey of the LPRSA (Windward 2014b), only approximately 1% of the shoreline was classified as containing aquatic vegetation, most of which was emergent rather than SAV. Such vegetation was limited to protected fringes of intertidal mudflats. Mudflats were found along 35% of the total LPRSA shoreline and were more prevalent below RM 8 (Table 2-1); 90% of the left bank mudflats were below RM 8, and 62% of the right bank mudflats were below RM 8, although the areal extent of mudflats was predominately on the right bank (91% of the total area) due to the inclusion of the expansive Kearney Point mudflats (Table 2-1).

2.2 BENTHIC INVERTEBRATES

Benthic invertebrates represent a highly diverse group of taxa that plays a key role in estuarine and riverine food webs (Thorp and Covich 2010a). Benthic invertebrates are an integral member of a fully functioning aquatic system and have a marked impact on ecosystems, because they sort, rework, and oxygenate sediment (Bolam et al. 2002) and alter biogeochemical fluxes (e.g., nutrient cycling through processing of detritus) (Covich et al. 1999). Furthermore, they provide a source of sustenance to many fish and wildlife species, particularly large-bodied individuals such as decapods (i.e., crabs) or mollusks (i.e., bivalves).

The purpose of this section is to describe the benthic invertebrate community data collected in the LPRSA by CPG between 2009 and 2010. In addition to small, infaunal invertebrates, macroinvertebrates (decapods) and mollusks (bivalves and snails) collected in the LPRSA by CPG during fish and decapod sampling events are briefly described in this section.

2.2.1 Benthic invertebrate community

Salinity exerts a primary influence on the benthic community structure, a relationship observed by numerous ecologists since it was initially described by Carriker (1967). In addition to salinity, other (potentially interrelated) factors that may affect the structure of the LPRSA benthic invertebrate community include sediment grain size; DO; water temperature; and other physical, chemical, and biological factors that are expressed over a similar spatial gradient. The influence of sediment contamination on the structure of the benthic invertebrate community is evaluated in Section 6 (and related appendices). The LPRSA benthic community is discussed in the context of various salinity zones, per agreement with USEPA.

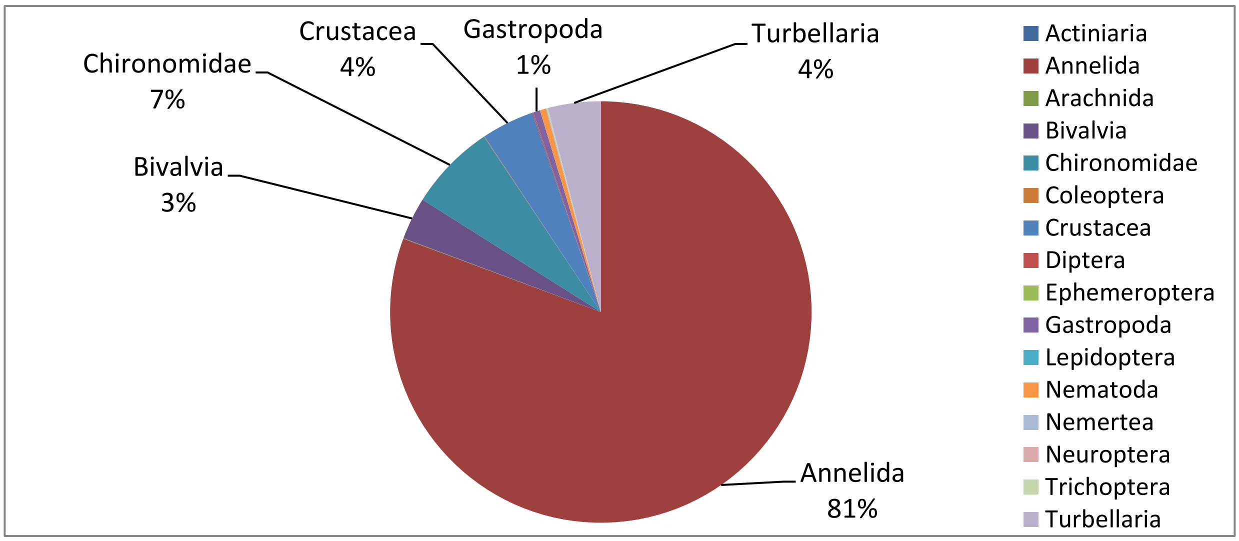
As discussed in Section 2.1.1.1, the location and extent of the salt wedge in the LPRSA, which affects the location of the fluvial estuarine zone, is dependent on seasonal and daily flow conditions. These flow conditions are influenced by both freshwater discharge and the tidal cycle. Based on the freshwater discharge during benthic community sampling in 2009, the salt wedge at this time extended upstream to approximately RM 7 to RM 8. The overall extent of the salt wedge, after considering the influence of the daily tidal cycle of 2.5 to 4.5 mi (Moffatt & Nichol 2013), varied from approximately RM 3 to RM 12. Thus, during benthic invertebrate community sampling in 2009, the fluvial estuarine salinity zone is estimated to have extended as far downstream as RM 3 and as far upstream as RM 12. The salinity zones described in Section 2.1.1.1 characterize these conditions.

The following subsections describe the major benthic taxa, seasonal trends in relative abundance, and an overview of the LPRSA benthic community.

2.2.1.1 Major taxa

Major taxonomic groups were identified for all individuals observed within the LPRSA in order to group species according to similar phylogenetic traits. Benthic data collected in fall 2009 were used to describe the benthic invertebrate community. Multiple invertebrate seasonal surveys were conducted (fall 2009, spring 2010, and summer 2010); however, all surveys indicated similar trends in terms of species counts and benthic community structure (Section 2.2.2).

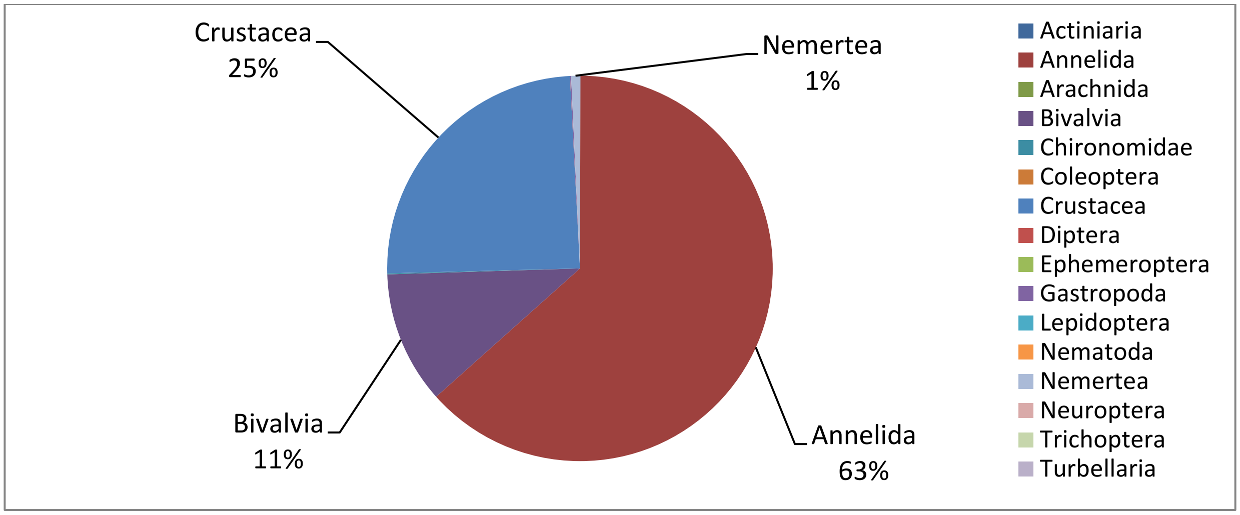
Figure 2-10 presents the benthic invertebrate community abundance of major taxa for the entire LPRSA, and Figures 2-11, 2-12, and 2-13 present the benthic invertebrate community abundance of major taxa within the three benthic salinity zones. The distribution of major taxa by river mile is shown in Figure 2-14.



Source: Windward (2014a)

Note: Major taxa not indicated contribute < 1% to the total abundance; Diptera category excludes Chironomidae.

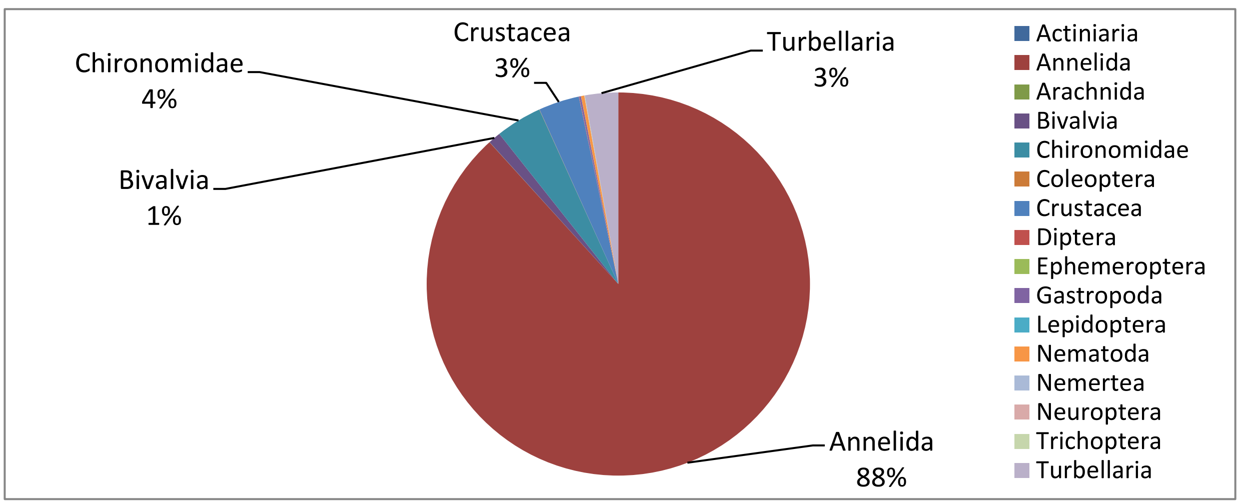
Figure 2-10. Distribution of benthic community relative abundance among major taxa throughout the LPRSA



Source: Windward (2014a)

Note: Major taxa not indicated contribute < 1% to the total abundance; Diptera category excludes Chironomidae.

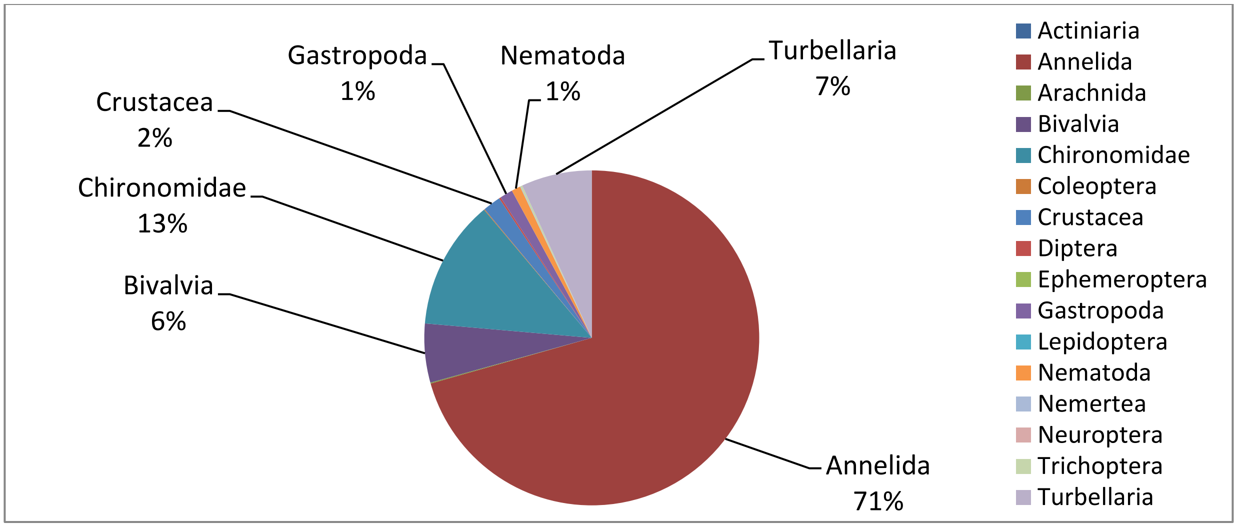
Figure 2-11. Distribution of benthic community relative abundance among major taxa in the benthic upper estuarine zone



Source: Windward (2014a)

Note: Major taxa not indicated contribute < 1% to the total abundance; Diptera category excludes Chironomidae.

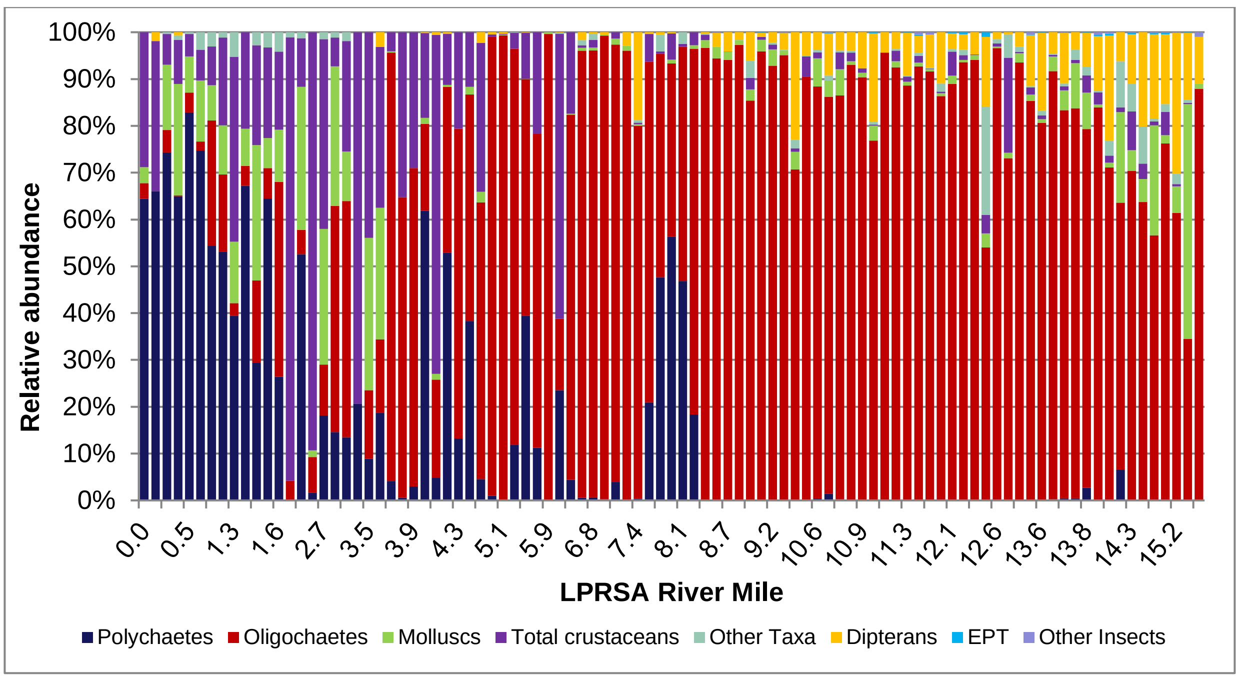
Figure 2-12. Distribution of benthic community relative abundance among major taxa in the benthic fluvial estuarine zone



Source: (Windward 2014a)

Note: Major taxa not indicated contribute < 1% to the total abundance; Diptera category excludes Chironomidae.

Figure 2-13. Distribution of benthic community relative abundance among major taxa in the benthic tidal freshwater zone



Source: Windward (2014a)

Note: Major taxa not indicated contribute < 1% to the total abundance; Chironomidae are included in Dipterans category; Other Insects category includes all non-Dipteran and non-EPT insects; Other Taxa category includes all other non-insect taxa (e.g., Turbellaria, Nematoda, Nemertea, Hirudinea [leeches], etc.).

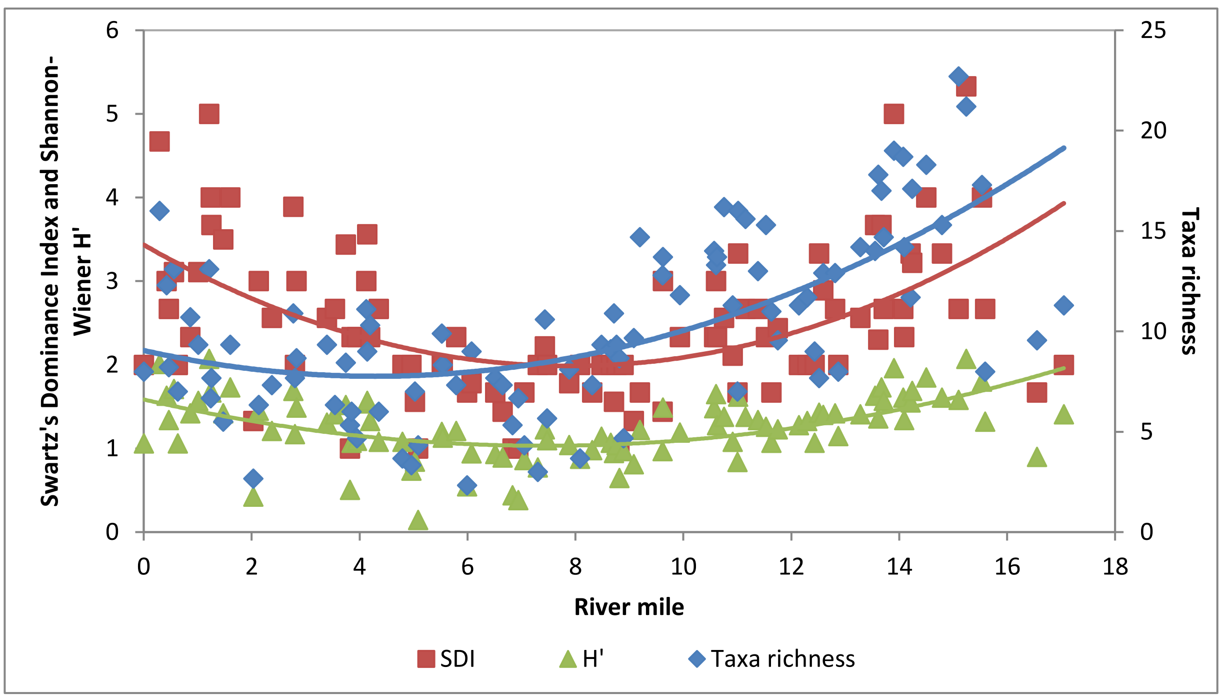
Figure 2-14. Relative abundance of major taxa in the LPRSA

Interpolating data collected in 2009 and 2010, the benthic invertebrate community in the LPRSA can be summarized as follows:

- **Entire LPRSA** – The majority of individuals within the entire LPRSA are annelid worms (approximately 81%). The vast majority of these worms are oligochaetes, particularly in the benthic tidal freshwater or fluvial estuarine zones; polychaetes are more prevalent in the benthic upper estuarine zone (Figure 2-10). Other major taxa that contribute to the total abundance throughout the LPRSA include bivalves, chironomids, crustaceans, gastropods, and turbellarians. The remaining taxa contribute < 1% to the overall abundance.
- **Upper Estuary** – Within the upper estuarine zone (RM 0 to RM 4), annelids account for approximately 63% of the total abundance (Figure 2-11). Where predominantly estuarine conditions are expected, crustacean and bivalve abundances account for 25 and 11% of the total abundance, respectively. The clam *M. balthica* is the most abundant bivalve in this zone, although various other clams have been observed. Crustaceans are predominately composed of *Gammarus* amphipods, *Balanus* barnacles, and *Cyathura polita*, an estuarine isopod.
- **Fluvial Estuary** – Within the benthic fluvial estuarine zone (i.e., RM 4 to RM 13), where salinities are most variable due to seasonal and daily excursions of the salt wedge, the structure of the benthic invertebrate community is dominated by annelids (88% of total abundance) (Figure 2-12). The majority of annelids in this zone are the oligochaete *Limnodrilus hoffmeisteri*, which is the most abundant species throughout all locations in the benthic fluvial estuarine zone (Windward 2014a). The relative abundances of crustaceans and bivalves decrease compared to communities in the benthic upper estuarine salinity zone, while the abundances of chironomids and turbellarians (which are more prevalent in freshwater) increase to 4 and 3% of the total abundance, respectively. Chironomids are composed primarily of *Chironomus* and *Procladius* genera. The relative abundances among bivalve taxa shift notably away from estuarine species to freshwater species, and in particular to the freshwater clams *Pisidium* spp. and *Sphaeriidae* spp. and *Corbicula* spp., which are brackish-tolerant clams.
- **Tidal Freshwater** – Between RM 13 and RM 17.4, above the expected upper extent of seasonal and daily salt wedge excursions, annelids contribute a lesser percentage of the total abundance compared to communities in the benthic fluvial estuarine zone (Figure 2-13). Gastropods, chironomids, turbellarians, and bivalves are present in greater total abundances between RM 13 and RM 17.4, relative to the benthic estuarine and fluvial estuarine zones. Gastropods are predominately composed of freshwater Hydrobiidae (mud snails), *Micromenetus dilatatus* (freshwater planorbid snail), and *Ferrissia* spp. (freshwater limpets). Chironomids are composed of *Chironomus* and *Procladius*

genera, as well as various other highly abundant species or genera. *Corbicula* spp. are the dominant bivalve taxa.

Review of the spatial distribution of major taxa in the LPRSA suggests that salinity (and its daily variation) plays a major role in structuring the benthic communities throughout the LPRSA. Crustaceans appear to be most abundant in higher-salinity estuarine waters. Conversely, gastropods, insects (e.g., chironomids), and non-annelid worms such as nematodes and turbellarians are more abundant in freshwater. Numerical dominance of the community by annelid worms is lower in sections of the river where the salinity regime is relatively stable (e.g., benthic upper estuarine and tidal freshwater zones) and significant in areas of variable salinity (i.e., fluvial estuarine zone). The dominance of salinity-tolerant species generally decreases and the diversity and richness of species generally increases from RM 8.5 to RM 17.4, peaking at approximately RM 15, above the influence of the salt wedge (Figure 2-15). The structure by river mile shows that salinity variations within the benthic fluvial estuarine zone have a marked influence on diversity and richness. As noted in Section 2.2.1, other (potentially interrelated) factors that may affect the structure of the LPRSA benthic invertebrate community include sediment grain size; DO; water temperature; and other physical, chemical, and biological factors that are expressed over a similar spatial gradient.



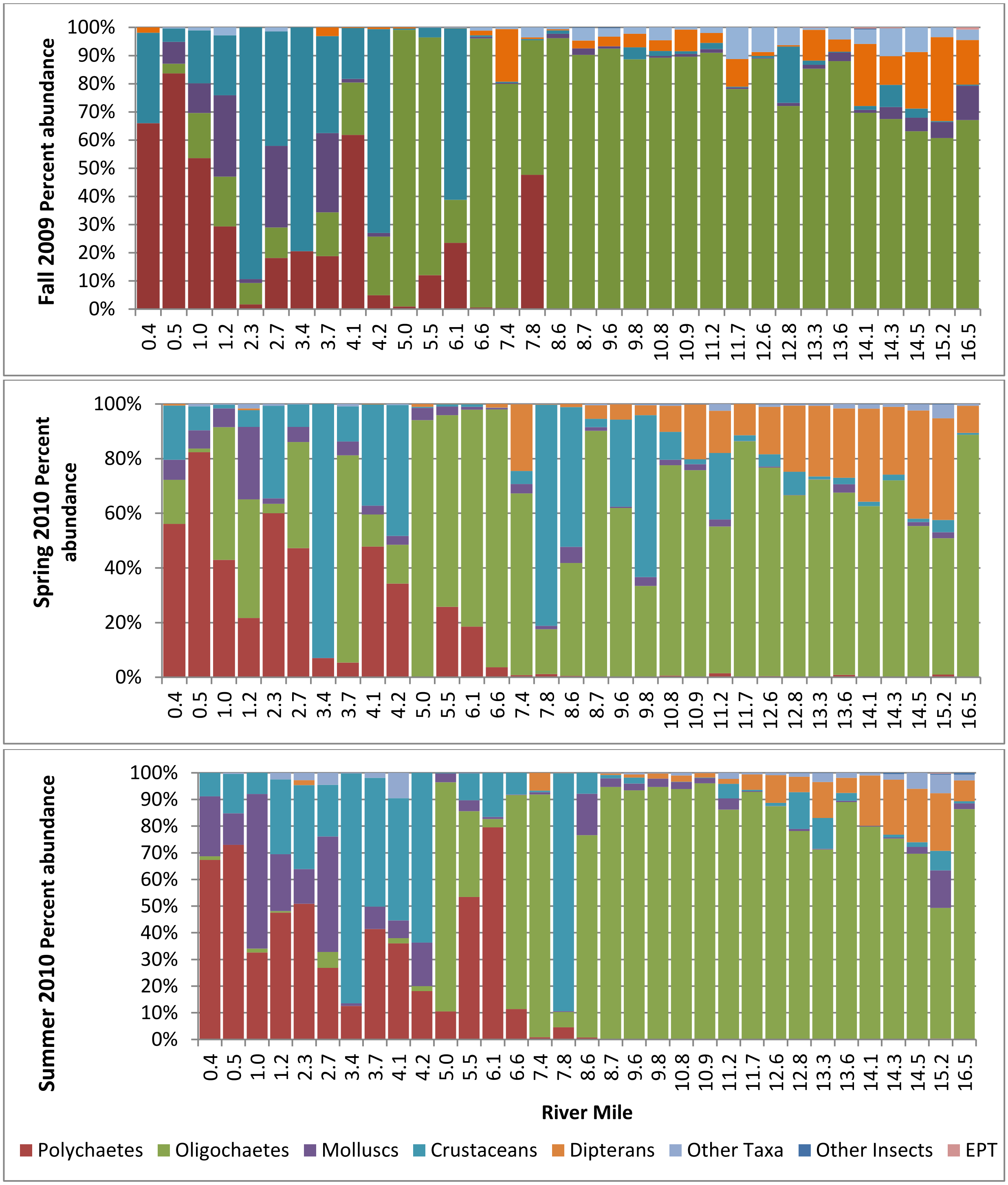
Source: Windward (2014a)

Note: Polynomial curves are provided for each metric, and the color of the curve matches that of the points for the respective metric; Swartz's Dominance Index (SDI) and Shannon-Wiener (H') are shown on the same scale, although the units of the two metrics are not the same.

Figure 2-15. Spatial trends in mean dominance, diversity, and taxa richness in locations from 2009 sampling in the LPRSA

2.2.1.2 Seasonal trends in relative abundance

Seasonal trends across the three surveys (fall 2009, spring 2010, and summer 2010) were evaluated. Figure 2-16 presents a comparison of major benthic invertebrate taxonomic groups across the three surveys. The distributions of polychaetes and oligochaetes (the dominant taxa) tracked the seasonal trends in interstitial salinity. The distribution of freshwater oligochaetes extended downriver seasonally, with the input of freshwater from storm events, generally beginning in the fall and lasting through the spring; the oligochaete distribution mostly remained upriver during the summer (at approximately RM 5, rather than to approximately RM 0.5), when there was less rainfall and, thus, less freshwater input (Windward 2014c).



Source: Windward (2014c, 2014a)

Note: Only resampled locations are shown. Chironomidae are included in Dipterans category; Other Insects category includes all non-Dipteran and non-EPT insects; Other Taxa category includes all other non-insect taxa, including Turbellaria, Nematoda, Nemertea, Hirudinea (leeches), and others.

Figure 2-16. Comparison of major benthic invertebrate taxonomic groups present in the LPRSA for the 2009 and 2010 surveys

Most of the species found in the brackish waters of the LPRSA are common to other estuaries of the northeastern United States (USEPA REMAP 2002). When the dominance of oligochaetes below RM 5 decreases during the summer, mollusk species (e.g., bivalves such as *M. balthica* or *Mulinia lateralis*) increase in dominance (Windward 2014a). In spring and summer, bivalves are particularly abundant between RM 0 and RM 3 (within the benthic upper estuarine zone) (Windward 2014c). In 2010, gastropod mollusks were also among the numerically dominant groups in the lower portion of the river, particularly at locations between RM 3 and RM 8.5; bivalves decreased in abundance in that same area.

Above RM 5, oligochaetes are numerically dominant at most locations, although chironomids, turbellarians, bivalves (e.g., *Corbicula* spp.), and amphipods are also dominant or among the most abundant taxa at some locations (Windward 2014a).

In fall 2009, diversity and richness in the zones with more stable salinity (between RM 0 and RM 1.5 and between RM 13.5 and RM 15.5) tended to be much greater than in other parts of the river (Figure 2-15), particularly compared to locations within the benthic fluvial estuarine zone (Windward 2018f). In summer 2010, diversity tended to be greatest near the mouth of the LPRSA, between RM 0 and RM 4, and above RM 14 (Windward 2014c). The seasonal shift in diversity is likely related to changes in seasonal freshwater flow and confirms the impact of saltwater migration on benthic community metrics. Greater but stable salinities at the mouth of the LPRSA during summer may allow for colonization up to RM 4 (Windward 2014c) by species adapted to higher salinities. Conversely, greater freshwater flows during fall exclude those same species between RM 1.5 and RM 4 (Windward 2018f). Taxa richness decreases within estuaries as salinity approaches approximately 5 to 8 ppth, or the “critical salinity” above which freshwater species and below which estuarine or marine species cannot effectively osmoregulate (Levinton 1982). Water temperatures may also play a role in invertebrate density and community composition (Haidekker 2004; Carolli et al. 2012).

2.2.1.3 Overview of LPRSA benthic community

The benthic invertebrate community of the LPRSA is dominated by deposit feeders, filter feeders, and detritivores (e.g., annelid worms, chironomids, and bivalves) (Germano & Associates 2005; Iannuzzi et al. 2008; Windward 2014c, a).[37] Polychaetes numerically dominate the upper estuarine zone, and oligochaetes numerically dominate the tidal freshwater zone. The distributions of polychaetes and oligochaetes are consistent with seasonal trends in interstitial salinity, which vary with the input of

[37] The SPI survey (Germano & Associates 2005) found that the average redox potential discontinuity layer depths for the LPRSA were 1.6 cm for the upper estuary (RM 0 to RM 4), 1.7 cm for the fluvial estuary (RM 4 to RM 13), and 2.1 cm for the tidal freshwater zone (RM 13 to RM 17.4 [i.e., Dundee Dam]). SPI was not feasible in the coarse sediment nearer to Dundee Dam.

freshwater due to high-flow events, generally beginning in the fall and lasting through the spring.

The structure of the benthic community appears to be driven primarily by salinity and the salinity gradient, in that the distributions of various taxa vary spatially along the tidal gradient (and shift upstream and downstream seasonally). Benthic community metrics of taxa richness, diversity, and evenness/dominance indicate that communities are disturbed at locations where salinity is most variable (i.e., fluvial estuarine salinity zone) relative to areas where salinity is more stable (i.e., tidal freshwater and upper estuarine salinity zones).

SPI data from the LPRSA suggests that locations in the LPRSA above RM 9 contain communities at a mature successional stage (Germano & Associates 2005). More locations below RM 6.5 were categorized as recently disturbed, and imagery indicated that such disturbances were primarily physical in nature (e.g., due to significant erosion or deposition events). Germano & Associates (2005) also observed excessive OC at more SPI survey locations in the upper estuarine zone, as evidenced by methane bubbles in profile images. Therefore, evidence provided from imaging suggests that dynamic hydrogeology (e.g., erosion and deposition), salinity, and organic enrichment are drivers of benthic community succession in the LPRSA.

The benthic community is typical of an urban estuarine system in the lower reaches of the LPRSA and of a freshwater community in the upper reaches of the LPRSA. Benthic diversity and richness in the fluvial estuarine zone are less than in downstream and upstream zones of more stable and tolerable salinities (Section 2.2.1). The fluvial estuarine zone is prone to changing conditions, to which few species are adapted. Seasonal shifts in the salt wedge due to changes in freshwater discharge determine where diverse communities can become established in the LPRSA.

2.2.2 Macroinvertebrates and mollusks

Several epibenthic decapod species were identified in the LPRSA, and a limited number of bivalves and gastropods were also observed (Table 2-3) during the 2009 and 2010 fish/decapod field collection efforts (Windward 2010c, 2011c). All decapods identified were classified as epibenthic omnivores; a very small number of other invertebrates encountered were either bivalves (n = 2 organisms) or gastropods (n = 1). The 2009 and 2010 fish/decapod sampling events were not intended to capture small epibenthic invertebrates such as mollusks, so the small numbers in Table 2-3 are not indicative of a depauperate or uniform epifaunal community.

Table 2-3. Summary of macroinvertebrates and mollusks collected during 2009 and 2010 LPRSA sampling

Common Name	Scientific Name	Habitat Preference	Reaches Where Collected	Count[a]
Invertebrate (decapods)– Epibenthic Omnivore				
Blue crab	*Callinectes sapidus*	freshwater/estuary	1, 2, 3, 4, 5, 6, 7, 8	1,148
Chinese mitten crab	*Eriocheir sinensis*	migratory/freshwater	8	2
Crayfish (unspecified)	na	freshwater/estuary	8	7
Grass shrimp	*Palaemonetes pugio*	freshwater/estuary	1, 2, 3, 4, 8	465
Mud crab (unspecified)	na	freshwater/estuary	1, 2, 3, 4, 5, 6, 7	280
Spinycheek crayfish	*Orconectes limosus*	estuary	7, 8	5
Guild total				**1,907**
Invertebrate (bivalves)				
Blue mussel	*Mytilus edulis*	fresh water/estuary	1	1
Clam (unspecified)	na	fresh water/estuary	8	1
Guild total				**2**
Invertebrate (gastropods)				
Snail (unspecified)	na	fresh water/estuary	1	139
Guild total				**139**
Invertebrate Total				**2,048**

[a] Count refers to the total number of each species caught and has not been normalized to area.
LPRSA – Lower Passaic River Study Area
na – not applicable (e.g., species not identified)

Of all epibenthic invertebrates caught during the recent fish surveys, 93% were decapods from the omnivore guild (Table 2-3). The most common decapod was blue crab (60% of total decapods), a target ecological receptor identified in the PFD (Windward and AECOM 2009) found in all reaches of the LPRSA. Blue crab were found in greater numbers in the lower portions of the LPRSA (80% of the blue crab collected in the LPRSA were collected below RM 10) than in the upper portions of the LPRSA.

Blue crabs are detritivores and scavengers throughout their range. Immature larvae are phytoplanktivorous (Darnell 1959 as cited in Hill et al. 1989) and consume dinoflagellates and copepod nauplii (Tagatz 1968 as cited in Hill et al. 1989). The omnivorous adults eat fish larvae, small shellfish, and aquatic plants (Van Engel 1958, Darnell 1959, and Tagatz 1968, all as cited in Hill et al. 1989); cannibalism is common among all blue crab life stages (Hay 1905, Churchill 1919, Darnell 1959, and Tagatz 1968, all as cited in Hill et al. 1989). Post-larval crabs are considered scavengers, bottom carnivores, detritivores, and omnivores (Hay 1905, Darnel1 1959, and Adkins 1972, all as cited in Hill et al. 1989). Diet studies have shown that the predominant prey consumed by blue crab vary greatly. Some common items are dead and live fish, crabs (including juvenile or molting blue crabs), organic debris, shrimp, mollusks (including mussels, clams, oysters, and snails), and aquatic plants (Newcombe 1945, Darnell 1959, Williams 1965, Tagatz 1968, Arnold 1984, and Warren 1985, all as cited in Iannuzzi et al. 1996). Truitt (1939, as cited in Hill et al. 1989) found that the roots, shoots, and leaves of eel grass (*Zostera* spp.), ditch grass (*Ruppia* spp.), sea lettuce, and salt marsh grass (*Spartina* spp.) were commonly eaten by crabs in saltwater marshes, tidal creeks, and other shallow estuarine areas. Darnell (1958 as cited in Hill et al. 1989) concluded that mollusks were the dominant prey of crabs wider than 120 mm.

2.3 FISH

Forty-five estuarine or freshwater fish species were identified throughout the LPRSA (Windward 2010c, 2011c) (Table 2-4). Earlier fish community surveys conducted in 1999 and 2000 that were limited to approximately the lower 7 mi of the LPRSA encountered many of the same species (Iannuzzi and Ludwig 2004). Results of the recent fish surveys taken over 1 year (Windward 2010c, e, j, 2011c) indicate many estuarine fish move throughout the river, as far upstream as Dundee Dam. Freshwater species generally follow the salt wedge, thus their location in the LPRSA changes accordingly.

Table 2-4. Summary of fish collected during 2009 and 2010 LPRSA sampling

Common Name	Scientific Name	Habitat Preference	Reaches Where Collected	Count[a]
Benthic Omnivore				
American eel, small (< 50 cm in length)[b]	*Anguilla rostrata*	freshwater/estuary	1, 2, 3, 4, 5, 6, 7, 8	743
Banded killifish	*Fundulus diaphanus*	freshwater	1, 2, 3, 4, 5, 6	359
Bluegill	*Lepomis macrochirus*	freshwater	3, 4, 5, 6, 7, 8	146
Common carp	*Cyprinus carpio*	freshwater/estuary	3, 4, 5, 6, 7, 8	215
Goby (unspecified)	na	estuary	1, 2	12
Green sunfish	*Lepomis cyanellus*	freshwater	8	2
Mummichog	*Fundulus heteroclitus*	estuary	1, 2, 3, 4, 5, 6, 8	1,696
Pumpkinseed	*Lepomis gibbosus*	freshwater	4, 5, 6, 7, 8	132
Redbreast sunfish	*Lepomis auritus*	freshwater	4, 5, 7, 8	113
Striped killifish	*Fundulus majalis*	freshwater	1, 2, 3, 4, 5, 6, 7	412
Tessellated darter	*Etheostoma olmstedi*	freshwater	4, 5, 6, 7, 8	52
Guild total				**3,882**
Invertivore/Omnivore				
Atlantic croaker	*Micropogonias undulatus*	estuary	3	1
Atlantic silverside	*Menidia menidia*	estuary	1, 2, 3, 4, 5, 6	242
Atlantic tomcod	*Microgadus tomcod*	migratory/estuary	1, 2, 3	8
Bay anchovy	*Anchoa mitchilli*	freshwater/estuary	2	3

Table 2-4. Summary of fish collected during 2009 and 2010 LPRSA sampling

Common Name	Scientific Name	Habitat Preference	Reaches Where Collected	Count[a]
Brown bullhead	*Ameiurus nebulosus*	estuary	3, 4, 6, 7, 8	11
Catfish (unspecified)	na	freshwater/estuary	8	1
Channel catfish	*Ictalurus punctatus*	freshwater/estuary	5, 6, 7, 8	17
Hogchoker	*Trinectes maculatus*	freshwater	1, 4	3
Inland silverside	*Menidia beryllina*	freshwater	1, 2, 3, 4, 5, 8	193
Mottled sculpin	*Cottus bairdii*	freshwater	8	3
Northern pipefish	*Syngnathus fuscus*	estuary	1, 4	6
Satinfin shiner	*Cyprinella analostana*	freshwater	8	3
Shiner (unspecified)	na	freshwater	7, 8	34
Silver perch	*Bairdiella chrysoura*	freshwater/estuary	1	1
Silver shiner	*Notropis photogenis*	freshwater	8	62
Spottail shiner	*Notropis hudsonius*	freshwater	3, 4, 5, 6, 7, 8	194
Striped mullet	*Mugil cephalus*	migratory/freshwater	1, 2, 3, 4, 5, 6	78
Sucker (unspecified)	na	freshwater	8	15
Weakfish	*Cynoscion regalis*	estuary	1, 3	4
White perch, small (< 20 cm in length)[c]	*Morone americana*	freshwater/estuary	1, 2, 3, 4, 5, 6, 7, 8	1,273
White sucker	*Catastomus commersoni*	freshwater/estuary	4, 5, 6, 7, 8	41
Winter flounder	*Pseudopleuronectes americanus*	freshwater/estuary	1, 2	3
Guild total				**2,196**
Planktivore				
Atlantic menhaden	*Brevoortia tyrannus*	migratory/estuary	1, 2, 3, 4, 5	284
Gizzard shad	*Dorosoma cepedianum*	freshwater	1, 2, 3, 4, 5, 6, 7, 8	251
Alewife	*Alosa pseudoharengus*	migratory/estuary	1, 2	5
Guild total				**540**
Piscivore/Invertivore				
American eel, large (length ≥ 50 cm)[b]	*Anguilla rostrata*	freshwater/estuary	1, 2, 3, 4, 5, 6, 7	47
Black crappie	*Pomoxis nigromaculatus*	freshwater	4, 6	3
Crevalle jack	*Caranx hippos*	estuary	1, 4	2
Northern searobin	*Prionotus carolinus*	migratory/estuary	1	1
Redfin pickerel	*Esox americanus*	freshwater	7	1
Rock bass	*Ambloplites rupestris*	freshwater/estuary	6, 8	13
Striped bass	*Morone saxatilis*	freshwater/estuary	1, 2, 3, 4, 5, 6, 7, 8	87
Summer flounder	*Paralichthys dentatus*	estuary	1, 2	2
White catfish	*Ameiurus catus*	freshwater/estuary	2, 3, 4, 5, 6, 7, 8	38

Table 2-4. Summary of fish collected during 2009 and 2010 LPRSA sampling

Common Name	Scientific Name	Habitat Preference	Reaches Where Collected	Count[a]
White perch, large (length ≥ 20 cm)[c]	*Morone americana*	freshwater/estuary	1, 2, 3, 4, 5, 6, 7, 8	53
Guild total				**247**
Piscivore				
Bluefish	*Pomatomus saltatrix*	estuary	1, 2	37
Largemouth bass	*Micropterus salmoides*	freshwater/estuary	4, 5, 8	21
Longnose gar	*Lepisosteus osseus*	freshwater	5	1
Northern pike	*Esox lucius*	freshwater/estuary	5, 6	2
Smallmouth bass	*Micropterus dolomieu*	freshwater/estuary	4, 5, 6, 7, 8	40
Guild total				**101**
Fish Total				**6,966**

[a] Count refers to the total number of each species caught and has not been normalized to area.

[b] American eel were divided into two size classes (small eel < 50 cm in length and large eel ≥ 50 cm in length) based on the different feeding characteristics of juvenile and adult eel. Of the eel for which length information was available (some eel were weighted in groups or were not whole when collected, and thus could not be measured), 6% were ≥ 50 cm in length.

[c] White perch were divided into two size classes (small perch < 20 cm in length and large perch ≥ 20 cm in length) based on the different feeding characteristics of juvenile and adult white perch. Of the perch for which length information was available (some perch were weighted in groups or were not whole when collected, and thus could not be measured), 4% were ≥ 20 cm in length.

LPRSA – Lower Passaic River Study Area

na – not applicable (e.g., species not identified)

The fish identified in the 2009 and 2010 surveys were classified into five general feeding guilds based on a review of their feeding habits (FishBase 2013). These general feeding guilds, along with a brief description of the assumed feeding habits of each group, are as follows.

- **Benthic omnivore** – feed near the river bottom and consume primarily benthic invertebrates and detrital material
- **Invertivore/omnivore** – consume a varied diet of invertebrates (aquatic and terrestrial), plant material, small crustaceans, small fish, and other small organisms
- **Planktivore** – filter-feeding fish that consume primarily plankton and other suspended materials
- **Piscivore/invertivore** – consume primarily small fish and various aquatic/terrestrial invertebrates
- **Piscivore** – consume primarily small fish

Table 2-4 presents the counts for species by guild and indicates where these species were caught in the LPRSA. The most abundant species within each of these feeding guilds are discussed in Sections 2.3.1 through 2.3.5, and an overview of the LPRSA fish community is presented in Section 2.3.6. The feeding guilds assigned in Table 2-4 reflect the feeding strategies of larger individuals (e.g., adults) rather than those of juveniles or earlier life stages. Feeding strategies in fish often change throughout their life cycles, typically driven by "gape limitation," or the size of prey (or other dietary items) that can fit into a fish's mouth; gape limitation increases as the fish grows over time.

2.3.1 Benthic omnivore

Fish from the benthic omnivore guild made up 56% of the total fish population caught during the 2009/2010 fish surveys. The most common benthic omnivores included the following:

- **Killifish species** – These included mummichog (*Fundulus heteroclitus*) (24% of the total catch), striped killifish (*Fundulus majalis*) (6% of the total catch), and banded killifish (*Fundulus diaphanous*) (5% of the total catch), which was targeted as a representative ecological receptor identified in the PFD (Windward and AECOM 2009).[38] Mummichog, striped killifish, or banded killifish were found in all reaches, but were predominantly found in the lower portions of the LPRSA (below RM 10).
- **Small American eel** – Small eel (< 50 cm in length) made up 11% of the total catch, and were found in all reaches of the LPRSA, although the vast majority were collected from Reach 8 (RM 14 and above).
- **Sunfish** – Sunfish (including bluegill [*Lepomis macrochirus*], pumpkinseed [*Lepomis gibbosus*], and redbreast sunfish [*Lepomis auritus*]) made up 6% of the total catch. These species were collected in all reaches above RM 5 (i.e., from Reaches 3 through 8), although the majority were collected above RM 10.
- **Common carp** – Carp made up 5% of the total catch and were found between RM 4 and RM 17.4.

Mummichog are opportunistic and will feed on almost any subtidal or intertidal benthic or water column organism (Weisberg and Lotrich 1982). Several studies conducted in varied habitats have demonstrated that the mummichog diet consists of detritus, algae, small crustaceans (i.e., amphipods, tanaids, copepods, and ostracods), insects (adult and larvae), and polychaetes (Abraham 1985; Allen et al. 1994; James-Pirri et al. 2001; Kneib 1986; Currin et al. 2003). Mummichog often consume detritus incidentally while feeding on the water's surface or bottom substrate (Kneib 1986). The size of mummichog prey is limited by the size of the fish's mouth (Vince et al. 1976, as cited in Abraham 1985). Therefore, larger mummichog typically consume larger prey

[38] Banded killifish and darter species were grouped as one species evaluated in this BERA, referred to as "other forage fish." Mummichog was evaluated independently of that group.

that are found at the water's surface or within the water column (Kneib and Stiven 1982, as cited in Abraham 1985), whereas larval and juvenile mummichog, which are smaller and restricted to intertidal marsh or mudflat areas, have a diet that consists primarily of small benthic invertebrates (Kneib 1986).

Mummichog and other killifish exhibit fairly small home ranges in euryhaline habitats, where they live throughout their life cycle (Sweeney et al. 1998; Smith and Able 1994). Killifish species are expected to be present in the LPRSA during all seasons and life stages. Striped killifish, banded killifish, and tessellated darters (*Etheostoma olmstedi*) have diets, distributions, and habitat preferences similar to those of mummichog (Abraham 1985; Phillips et al. 2007; USGS 2006; Pennsylvania Sea Grant 2006; Environment Canada 2006).

American eel have a diverse diet that includes annelids, polychaetes, insect larvae and nymphs, crustaceans, bivalves, gastropods, fish, frogs, and mice (Facey and Van Den Avyle 1987; Morrison 2001; Gray 1992; ASMFC 2000). Juvenile eel feed primarily on the lower trophic level prey items listed herein, while adult American eels feed higher on the food chain (these larger eel are included in the invertivore/piscivore feeding guild; Section 2.3.4). The most common fish consumed by adult American eel are American eel elvers, other eel species, and slow-moving, bottom-dwelling fish (Gray 1992). American eel tend to feed near the sediment-water interface, and they scavenge from dead organisms (Facey and Van Den Avyle 1987). Eel larvae likely feed on plankton when living in a marine environment (Gray 1992). Fish and invertebrates at both juvenile and adult life stages are consumed by American eel (NJDEP 2001a); prey size tends to increase as eel size increases (Ogden 1970).

American eel may be present in the LPRSA during several life stages, including the glass, elver, yellow, and silver eel stages. Spawning eel return to the Sargasso Sea; eggs are fertilized and hatch in the marine environment, only drifting into nearshore Northeast US estuaries as juveniles (i.e., glass eel) (ASMFC 2000). Eel tend to have restricted home ranges (e.g., 100 m) (Ford and Mercer 1986) in areas where eel can seek shelter in soft substrate (ASMFC 2000) or under piers (Able et al. 1998).

Bluegill are opportunistic feeders and alter their diet according to food availability (Keast and Webb 1966). Fry feed primarily on zooplankton and small insects (Werner 1969). Juveniles feed on zooplankton, crustaceans, aquatic and terrestrial insects and worms, and some plant materials (Page and Burr 1991; Scott and Crossman 1973; Emig 1966; Scidmore and Woods 1960). Adults are known to feed on snails and small minnows (Page and Burr 1991). Pumpkinseed feed throughout the water column, and their diet consists of small but diverse food items including zooplankton, insects, insect larvae, mollusks, snails, other crustaceans, fish eggs, and small fish and vertebrates (Holtan 1998b; Scott and Crossman 1973; McCairns and Fox 2004). Redbreast sunfish are considered generalists, feeding on aquatic insects (e.g., dipterans, ephemeropterans, trichopterans, and terrestrial insects) (Bass and Hitt 1974; Sandow et al. 1974; Coomer et al. 1977; Benke et al. 1979; Henry 1979). As opportunistic feeders, their diet varies according to prey size and availability. Redbreast sunfish have also been found to ingest significant amounts of decapod crustaceans and fish (Aho et al. 1986).

Bluegill, pumpkinseed, and other sunfish live out their life cycles in low-velocity freshwater, preferring shallow waters with sufficient vegetative cover (Aho et al. 1986; Holtan 1998b). These sunfish species are expected to be present in the LPRSA during all seasons and life stages. Nesting occurs preferentially in sand and gravel, but may occur over any substrate (Stuber et al. 1982a).

Common carp eat a wide variety of aquatic plants, algae, insect larvae, other invertebrates, and small fish. Carp are mainly bottom dwellers but sometimes search for food in the middle and upper layers of the water column, and also consume aquatic plants and insects from the surface (FishBase 2007; FAO 2011). They usually feed by rooting in the bottom substrate with their snouts, eating the food they dislodge, along with fine sediment and detritus (Pennsylvania FBC 2011). It should also be noted that catfish, through their behavior and use of bedded sediment as habitat, can also disturb sediment, as can other benthic-feeding native species such as suckers. Adult common carp are opportunistic feeders that eat plant and animal material. Both adults and juveniles feed on aquatic insects, crustaceans, annelids, mollusks, aquatic plants and algae, benthic organisms (e.g., chironomids, gastropods, and other larval insects), detritus, insect/fish larvae, small fish, and plankton (e.g., cladocerans, copepods, amphipods, and mysids) (Maryland DNR 2007a; Garcia-Berthou 2001; USGS 2010). Algae, detritus, pebbles, and sediment are commonly found in the stomachs of common carp (Campos 2005).

Common carp tolerate a broad range of environmental conditions including high levels of turbidity and reduced DO (NJDEP 2001b). They are found primarily in low-velocity freshwater but will also tolerate brackish conditions to some extent (Edwards and Twomey 1982). Carp are resident, overwintering species (Edwards and Twomey 1982) and so are expected to be present in the LPRSA during all seasons and life stages.

2.3.2 Invertivore/omnivore

Fish from the invertivore/omnivore guild made up 32% of all fish caught during the 2009/2010 fish surveys. The most common invertivore/omnivores included the following:

- **Small white perch** – White perch less than 20 cm in length made up 18% of the total catch. White perch, a target ecological receptor identified in the PFD (Windward and AECOM 2009), were found in all reaches of the LPRSA, but predominately below RM 10.
- **Silversides** – Atlantic and inland silversides (*Menidia menidia* and *Menidia beryllina*, respectively) made up 6% of the total catch.

Channel catfish and brown bullhead (*Ameiurus nebulosus*) were also targeted as ecological receptors within this guild identified in the PFD (Windward and AECOM 2009) but were not frequently caught. Together, these species accounted for less than 1% of the total catch.

The white perch's common dietary components include amphipods, shrimp, and copepods, based on regional studies for the Hudson and Hackensack Rivers (Bath and O'Connor 1985; Weis 2005). White perch diets vary depending on the time of year and the maturity of the individual fish; the greater proportion of the white perch's late summer and fall diet consists of fish, while the greater proportion of their winter and spring diet is invertebrates (Bath and O'Connor 1985; Weis 2005). White perch tend to be benthic feeders, feeding near the sediment-water interface (Bath and O'Connor 1985; Weisberg and Janicki 1990).

White perch are euryhaline, residing in estuaries and rivers throughout their life cycle (Klauda et al. 1988). When spawning, adults migrate upstream into cooler freshwater, whereas post-yolk-sac larvae migrate downstream into estuaries (Klauda et al. 1988). Overwintering tends to occur in brackish or estuarine waters with soft substrates (Setzler-Hamilton 1991). Therefore, white perch may be present throughout the LPRSA during any season and during their entire life cycle.

Catfish are opportunistic bottom feeders, consuming a variety of plants, animals, and detritus. Channel catfish have a variable diet that includes SFF, terrestrial and aquatic insects, detritus, plant material, crayfish, and mollusks (Fewlass 1980; Holtan 1998a; McMahon and Terrell 1982). Adult catfish feed predominantly on fish, whereas juvenile catfish feed primarily on insects, insect larvae, and zooplankton (Wellborn 1988; Holtan 1998a; McMahon and Terrell 1982). As channel catfish grow, they begin to feed on snails, crayfish, and small fish, but still eat aquatic insects and occasionally plant matter (Holtan 1998a).

Channel catfish are present in warm, moderately flowing freshwater habitats throughout their life cycle, associated primarily with sandy and gravelly sediments (Wellborn 1988). They are resident, overwintering species (McMahon and Terrell 1982) and so are expected to be present in the LPRSA during all seasons and life stages.

Brown bullhead eat a wide variety of plant and animal material, including aquatic insects and larvae, worms, minnows and other small fish, crayfish, snails, mollusks, fish eggs, frogs, and algae (Wisconsin DNR 2008a). Studies have shown the brown bullhead's preference for midge larvae (i.e., chironomids), amphipods (*Hyalella* spp.), and oligochaetes (TAMS and Menzie-Cura 2000; USEPA 2002e). Brown bullhead can also consume filamentous algae, which can make up as much as 60% of their diet (Gunn et al. 1977 as cited in USEPA 2002e). Juveniles feed mostly on cladocerans, ostracods, amphipods, insects, and fish eggs and larvae (FishBase 2007).

Brown bullhead are a freshwater species that prefers muddy bottoms (USEPA 2002e). They can tolerate a range of environmental conditions including low DO and high turbidity (USEPA 2002e). Brown bullhead hibernate in their resident streams when overwintering (Wisconsin DNR 2008a) and so are expected to be present in the LPRSA during all seasons and life stages.

2.3.3 Planktivore

Fish from the planktivore guild made up 8% of all fish caught during the 2009/2010 fish surveys. The most common planktivores included the following:

- **Atlantic menhaden (*Brevoortia tyrannus*)** – Menhaden made up 4% of the total catch, and were found only below RM 10.
- **Gizzard shad (*Dorosoma cepedianum*)** – Shad also made up 4% of the total catch, and were found primarily between RM 6 and RM 14 (Reaches 4 through 7). Some gizzard shad were also found in Reaches 2 and 8.

Atlantic menhaden occupy two distinct feeding niches during their lifetime. Menhaden are size-selective zooplankton feeders as larvae and filter feeders as juveniles and adults (EBFM 2011; Lewis and Peters 1994). From the first-feeding larval stage into the pre-juvenile stage, Atlantic menhaden selectively sight-feed on individual planktonic organisms (Chipman 1959; June and Carlson 1971). Govoni et al. (1983) noted that small menhaden prey heavily on larger phytoplankton (predominantly dinoflagellates) and some zooplankton, benthos, and benthic detritus. As the menhaden larvae grow, phytoplankton become less important and (larger) zooplankton, especially copepods of all life stages, become more important.

Atlantic menhaden are predominately marine or estuarine fish, living out their life cycle in such habitats (Rogers and van den Avyle 1989). Adult menhaden have been known to migrate into tidal estuaries, moving into waters at the limits of their salinity tolerance (Rogers and van den Avyle 1989). Atlantic menhaden tend to migrate inshore during the spring and then to offshore waters during the fall to spawn (Rogers and van den Avyle 1989). Therefore, Atlantic menhaden found in the LPRSA will spend no more than half of their life cycle there.

Adult gizzard shad are almost entirely herbivorous filter feeders, primarily feeding on algae and organic matter filtered out of the water column and sediment; juveniles feed predominately on zooplankton (Mundahl 1988; Werner 1980, 2004). The diversity of diet items varies widely with season and local availability. Bodola (1965) found mostly free-floating phytoplankton in adult shad captured in open waters, whereas those captured in littoral vegetation contained cladocerans, copepods, rotifers, and small aquatic insect larvae.

Gizzard shad are found in fresh to brackish waters in the water column or along the sediment surface, and they exhibit only minor migrations during their life cycle (Williamson and Nelson 1985). Specifically, gizzard shad that are present in brackish waters migrate into freshwater to spawn; therefore, they are a resident species, expected to be present in the LPRSA during all seasons and life stages (Williamson and Nelson 1985).

2.3.4 Piscivore/invertivore

Fish from the piscivore/invertivore guild made up 3% of all fish caught during the recent fish surveys. The most common piscivore/invertivores included the following:

- **Striped bass (*Morone saxatilis*)** – Striped bass made up 1.2% of the total catch, and were found in all reaches of the LPRSA.
- **Large white perch** – White perch > 20 cm in length made up 0.8% of the total catch, and were found in all reaches of the LPRSA.
- **Large American eel** – American eel ≥ 50 cm in length made up 0.7% of the total catch, and were found in Reaches 1 through 7 (RM 0 through RM 14) of the LPRSA.

Striped bass are opportunistic and carnivorous and have a diverse diet (Westin and Rogers 1978). Young-of-the-year striped bass are known to prefer copepods, cladocerans and chironomid larvae, mysids, and other insects (Markle and Grant 1970; Beaven and Mihursky 1980). Juvenile bass feed primarily on small fish, decapods, amphipods, and mysids (Bason et al. 1975; Bason 1971; Markle and Grant 1970). Adult striped bass are primarily piscivorous and prey items vary with seasonal availability (Hollis 1952). In Chesapeake Bay, for example, bay anchovy (*Anchoa mitchilli*) and Atlantic menhaden are principal prey during summer and fall, while in winter, spot (*Leiostomus xanthurus*), and croaker dominate the bass's diet. In the spring, Manooch (1973) found that blue crabs were a major prey item. Other stomach contents recorded from adult striped bass include alewife (*Alosa pseudoharengus*), blueback herring, American eel, American lobster, squid, crabs, clams, and mussels (Smith and Wells 1977).

Striped bass are found under many conditions including marine, estuarine, and riverine conditions (Crance 1984). Adult bass live primarily in nearshore estuarine embayments, but migrate seasonally into streams to spawn in areas characterized by coarse substrate and high-velocity waters (Crance 1984). Spawning occurs from just above the salt wedge to (typically) within 40 km from the mouth of a stream (Crance 1984); therefore, striped bass could be present throughout the LPRSA, most likely seasonally (during spawning) and in freshwater areas.

White catfish (*Ameiurus catus*) were initially targeted as an alternate ecological receptor for channel catfish and brown bullhead in the PFD (Windward and AECOM 2009), although like these species, white catfish were not found frequently in the LPRSA. The diets of channel catfish and brown bullhead have been evaluated in several studies, but little information is available on the diet of white catfish. White catfish eat some plant material, but mostly consume midge larvae and other aquatic insects, crustaceans, and fish (Pennsylvania FBC 2011). The available information indicates that white catfish are carnivorous bottom feeders, with juveniles consuming mostly smaller invertebrates, and adult shifting their diets towards larger invertebrates and fish (California Fish Website 2013).

The habitat preferences of white catfish have not been discussed in great detail in the literature. It is assumed that white catfish and channel catfish exhibit similar preferences, the latter of which is described briefly in Section 2.3.2.

American eel and white perch were discussed in Sections 2.3.1 and 2.3.2, respectively.

2.3.5 Piscivore

Fish from the piscivore guild made up just 1% of all fish caught during the recent fish surveys. The most common piscivores included the following:

- **Smallmouth and largemouth bass (*Micropterus dolomieu* and *Micropterus salmoides*, respectively)** – These bass species made up 0.9% of the total catch, and were found above RM 6 (Reaches 4 through 8).
- **Bluefish (*Pomatomus saltatrix*)** – Bluefish made up 0.5% of the total catch, and were found only in the lower portion of the LPRSA (below RM 4).

Northern pike (*Esox lucius*) were not frequently caught (only two individuals were taken) and accounted for less than 0.1% of the total catch.

Both bass species have similar diets, which are limited by the size of the individual's mouth and the seasonal availability of prey (Edwards et al. 1983; Pflug and Pauley 1984). Adult bass are piscivorous, predominately eating fish such as bluegill, minnows, perch, shiners, smelt, sculpin, suckers, and smaller centrarchids (Scott and Crossman 1973). Largemouth bass longer than 5 cm feed almost exclusively on other fish (Scott and Crossman 1973; TAMS and Menzie-Cura 2000). They are opportunistic and will also eat crayfish, frogs, insects, snakes, and small mammals and birds that enter the water (Scott and Crossman 1973; FishBase 2007); adults also cannibalize young fish from other parents (Scott and Crossman 1973). Fry and juvenile bass feed on plankton, amphipods and copepods, insects, insect larvae, and small fish; they also cannibalize one another (Stuber et al. 1982b).

Smallmouth bass are non-migratory freshwater fish (Edwards et al. 1983). Smallmouth bass habitat is characterized by cool, clear waters with abundant shade and cover and coarse substrate (Edwards et al. 1983). Such habitat provides both safety from predators and a means to ambush prey. Conversely, largemouth bass prefer deeper, slow-moving waters with soft substrates, which allow for successful overwintering (Stuber et al. 1982b). Bass do not tolerate low DO but will tolerate periodically increased turbidity (Edwards et al. 1983; Stuber et al. 1982b). Bass show strong site-fidelity, moving very little from season to season (Edwards et al. 1983). Largemouth and smallmouth bass are present in the LPRSA throughout the year and during all life stages.

Bluefish are voracious predators throughout their life cycle, relying primarily on vision to detect prey (Olla et al. 1970; Wilk 1977). The diets of bluefish larvae and early juveniles have not been well studied, but they presumably select various zooplankton, including the larvae of other pelagic-spawning fish (Kendall and Walford 1979; Norcross et al. 1974). Young-of-the-year arriving in coastal nursery areas feed on small shrimp, anchovies, killifish, silversides, and other available prey; those at sea likely forage on small pelagic fish and crustaceans. The list of potential prey increases as bluefish grow in size. A wide variety of fish and invertebrates have been recovered from adult bluefish stomachs, including common squid, various species of shrimp and crabs, alewives, shad, herring, Atlantic menhaden, silver hake (*Merluccius bilinearis*), pinfish (*Lagodon rhomboids*), spot, butterfish (*Lepidocybium flavobrunneum*), smaller bluefish, and many other species.

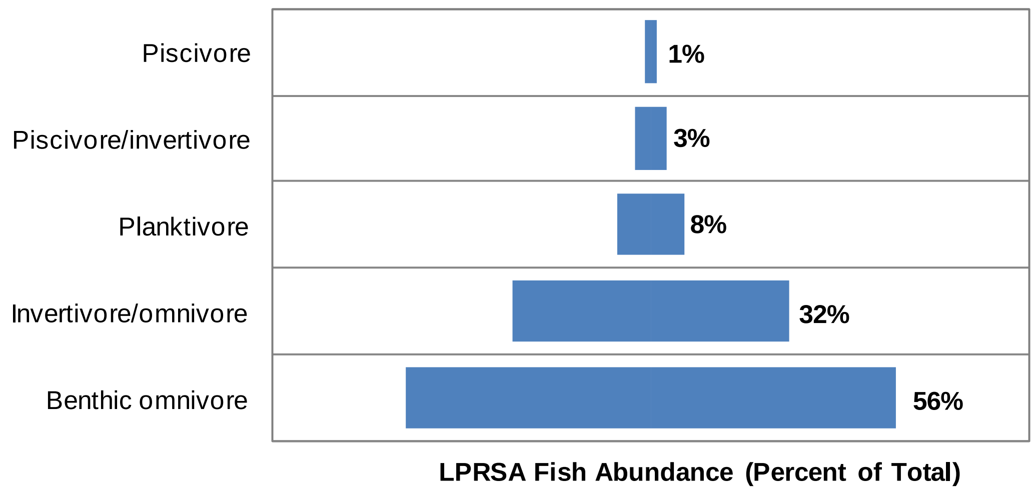
Bluefish are marine and estuarine species that spend little time within tidal rivers, preferring higher-energy waters along coastal rocky headlands or surf zones (Heavner 2001). Small bluefish reside within estuaries year-round, typically migrating further offshore as they grow larger (Heavner 2001). Bluefish tolerate salinities as low as 7 ppth (Heavner 2001), so this species is typically limited to the first river mile of the LPRSA. During high freshwater discharges (e.g., during spring), it is unlikely that bluefish will tolerate salinities in the LPRSA at all, and will migrate to marine waters.

Northern pike are primarily ambush piscivores, preying on fish species such as shiners, minnows, perch, bluegill and other sunfish, and suckers (Wisconsin DNR 2008b; MDNR 2013b). Young pike feed on zooplankton and aquatic invertebrates, but soon switch to a fish diet; large pike have been known to feed on frogs, ducklings, small waterfowl, rodents and other small mammals, or any living vertebrate that can fit down its gullet, including smaller pike (Wisconsin DNR 2008b; MDNR 2013a, b, c). Pike prefer food that is approximately one-third to one-half their own size (MDNR 2013c).

Northern pike prefer sluggish, shallow, cool water with extensive vegetation (Wisconsin DNR 2008b). They are able to tolerate fairly low DO and winter conditions in freshwaters (Wisconsin DNR 2008b; Inskip 1982). Northern pike are restricted to freshwaters in North America (Inskip 1982), so they have a limited distribution in the LPRSA. However, Northern pike are present in the LPRSA during all life stages and seasons.

2.3.6 Overall LPRSA fish community

Of the total fish caught during the recent surveys,[39] the majority of fish collected (87%) were classified as benthic omnivores or invertivores/omnivores.[40] The remaining fish caught were classified as planktivores (8%), invertivores/piscivores (3%), or piscivores (1%) (Figure 2-17). The general numbers of fish caught within each feeding guild during these surveys are assumed to generally represent the relative numbers of fish present in the LPRSA. A variety of sampling methods were used during the 2009 and 2010 fish sampling efforts to target different types of fish, including minnow traps, eel traps, box traps, trotlines, cast nets, dip nets, gillnets, beach seine nets, backpack electrofishers, and boat electrofishers. Additionally, the sampling events were conducted throughout the year to cover the range of seasons in the LPRSA, further adding to the representativeness of this fish community data.



Note: Data are based on fish species counts presented in Table 2-4. Fish guilds are based on information in the literature about feeding strategies of large individuals (e.g., adults) within a species. Percentages are rounded to the nearest percent.

Figure 2-17. LPRSA general fish feeding guild abundance from the 2009 and 2010 fish community surveys

As discussed in Sections 2.3.1 through 2.3.5, the feeding guilds in Figure 2-17 were primarily composed of the following species:

- **Benthic omnivore (56% of total catch)** – composed primarily of SFF (mummichog and killifish) and small American eel; also includes common carp

[39] Total fish collected is based on fish caught during the three fish community seasonal surveys conducted from August to September 2009 (late spring/early summer), January to February 2010 (winter), and June to July 2010 (late spring/early summer), as well as the fish collected as part of the SFF sampling efforts from June to August 2010.

[40] The sum of percentages reported in Figure 2-17 for benthic omnivores and invertivores/omnivores is 88% due to rounding of values in the figure.

- **Invertivore/omnivore (32% of total catch)** – composed primarily of small white perch and silverside
- **Planktivore (8% of total catch)** – composed primarily of Atlantic menhaden and gizzard shad
- **Piscivore/invertivore (3% of total catch)** – composed primarily of striped bass, large white perch, and large American eel
- **Piscivore (1% of total catch)** – composed primarily of smallmouth bass, largemouth bass, and bluefish

These data indicate that the LPRSA fish community is primarily a benthic-dominated food chain, as a large percentage of the fish species found are predominately benthic feeders, consistent with an urban river system. Similarly, the available data for the fish community above Dundee Dam indicate that it is benthic dominated (77% benthic omnivores; Figure 2-1 of Appendix J).

2.4 Birds

The LPRSA provides important but limited and fragmented habitat for avian species. Based on a 2010 survey of LPRSA shoreline habitats (Windward 2014b), it was determined that there are limited mudflats for sediment-probing birds and some riparian habitat for species inhabiting the shoreline. Specifically, mudflats were present along 35% of the total LPRSA shoreline, mostly downstream of RM 8; the large mudflat at Kearny Point (RM 0) accounted for the majority of all mudflats in the LPRSA.[41] Significant marsh habitat is largely absent from the LPRSA shoreline, although it is present at Kearny Point near the mouth of the LPRSA. CPG conducted four seasonal bird surveys throughout the 17.4 mi of the LPRSA in 2010 and 2011 (summer [August 2010], fall [October 2010], winter [January 2011], and spring [May 2011]) (Table 2-5) (Windward 2011a, 2019e) to characterize the avian community. A total of 49 aquatic- or semi-aquatic-feeding bird species were observed during the four seasonal surveys. Many of the same species were observed during the 2010/2011 surveys as during earlier avian survey conducted in 1999/2000 on the lower portion of the LPRSA (up to RM 6) (Ludwig et al. 2010). Three additional species that were observed infrequently in the 1999/2000 surveys were not seen in the 2010/2011 surveys: white-winged scoter, black scoter, and little blue heron.

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[41] Although spatially limited, the existing mudflats in the LPRSA provide important foraging habitat for aquatic-feeding birds.

Table 2-5. Aquatic- and semi-aquatic-feeding bird species observed during 2010 and 2011 LPRSA field surveys

Species		Season Observed			
Common Name	Scientific Name	Summer 2010	Fall 2010	Winter 2011	Spring 2011
Gulls and terns					
Common tern[a]	*Sterna hirundo*	X			X
Glaucous gull	*Larus hyperboreus*			X	
Great black-backed gull	*Larus marinus*	X	X	X	X
Herring gull	*Larus argentatus*	X	X	X	X
Iceland gull	*Larus glaucoides*			X	
Laughing gull	*Larus atricilla*	X	X		X
Lesser black-backed gull	*Larus fuscus*	X	X	X	
Ring-billed gull	*Larus delawarensis*	X	X	X	X
Swans, geese, and ducks					
American black duck	*Anas rubripes*	X	X	X	X
Brant	*Branta bernicla*		X	X	X
Bufflehead	*Bucephala albeola*			X	
Cackling goose	*Branta hutchinsii*		X		
Canada goose	*Branta canadensis*	X	X	X	X
Canvasback	*Aythya valisineria*			X	
Common merganser	*Mergus merganser*			X	
Gadwall	*Anas strepera*		X	X	X
Hooded merganser	*Lophodytes cucullatus*			X	
Mallard	*Anas platyrhynchos*	X	X	X	X
Mute swan	*Cygnus olor*		X		
Northern pintail	*Anas acuta*	X	X		
Red-breasted merganser	*Mergus serrator*			X	
Ruddy duck	*Oxyura jamaicensis*			X	
Snow goose	*Chen caerulescens*		X		
Wood duck	*Aix sponsa*	X	X		X
Shorebirds					
Black-bellied plover	*Pluvialis squatarola*				X
Greater yellowlegs	*Tringa melanoleuca*		X		
Killdeer	*Charadrius vociferus*	X	X		X
Least sandpiper	*Calidris minutilla*	X			X
Lesser yellowlegs	*Tringa flavipes*	X			
Pectoral sandpiper	*Calidris melanotos*	X			

Table 2-5. Aquatic- and semi-aquatic-feeding bird species observed during 2010 and 2011 LPRSA field surveys

Species		Season Observed			
Common Name	**Scientific Name**	**Summer 2010**	**Fall 2010**	**Winter 2011**	**Spring 2011**
Sanderling[a]	*Calidris alba*	X			
Semipalmated plover	*Charadrius semipalmatus*	X			X
Semipalmated sandpiper[a]	*Calidris pusilla*	X			X
Solitary sandpiper	*Tringa solitaria*	X			
Spotted sandpiper[a]	*Actitis macularia*	X	X		X
Western sandpiper	*Calidris mauri*	X			
White-rumped sandpiper	*Calidris fuscicollis*	X			
Wading birds					
Black-crowned night heron[a]	*Nycticorax*	X	X		X
Great blue heron[a]	*Ardea herodias*	X	X	X	X
Great egret	*Ardea alba*	X	X		X
Green heron	*Butorides virescens*	X			X
Snowy egret[a]	*Egretta thula*	X			
Hawks and allies					
Bald eagle[b]	*Haliaeetus leucocephalus*		X		
Osprey[c]	*Pandion haliaetus*	X	X		X
Cormorants					
Double-crested cormorant	*Phalacrocorax auritus*	X	X	X	X
Kingfishers					
Belted kingfisher	*Ceryle alcyon*	X	X		X
Icterids					
Boat-tailed grackle	*Quiscalus major*	X	X		
Red-winged blackbird	*Agelaius phoeniceus*	X	X	X	X
Rusty blackbird	*Euphagus carolinus*		X		

[a] Listed by New Jersey as a species of special concern.

[b] Listed by New Jersey as endangered for the breeding population and threatened for the nonbreeding population.

[c] Listed by the State of New Jersey as threatened for the breeding population.

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The most frequently observed birds from 2010/2011 were separated into five major groups for discussion, as follows:

- Gulls and terns
- Swans, geese, and ducks
- Shorebirds
- Wading birds
- Other aquatic-feeding birds, including osprey (*Pandion haliaetus*), bald eagle (*Haliaeetus leucocephalus*), cormorants (*Phalacrocoracidae* spp.), and kingfishers (*Alcedinidae* spp.)

Birds in the first two groups (gulls and terns; swans, geese, and ducks) were the most common, with numbers and relative abundances of species varying by season (Figure 2-18). Shorebirds, wading birds, and other bird species were less frequently observed. Each of these groups of birds is described in more detail in the subsections that follow.

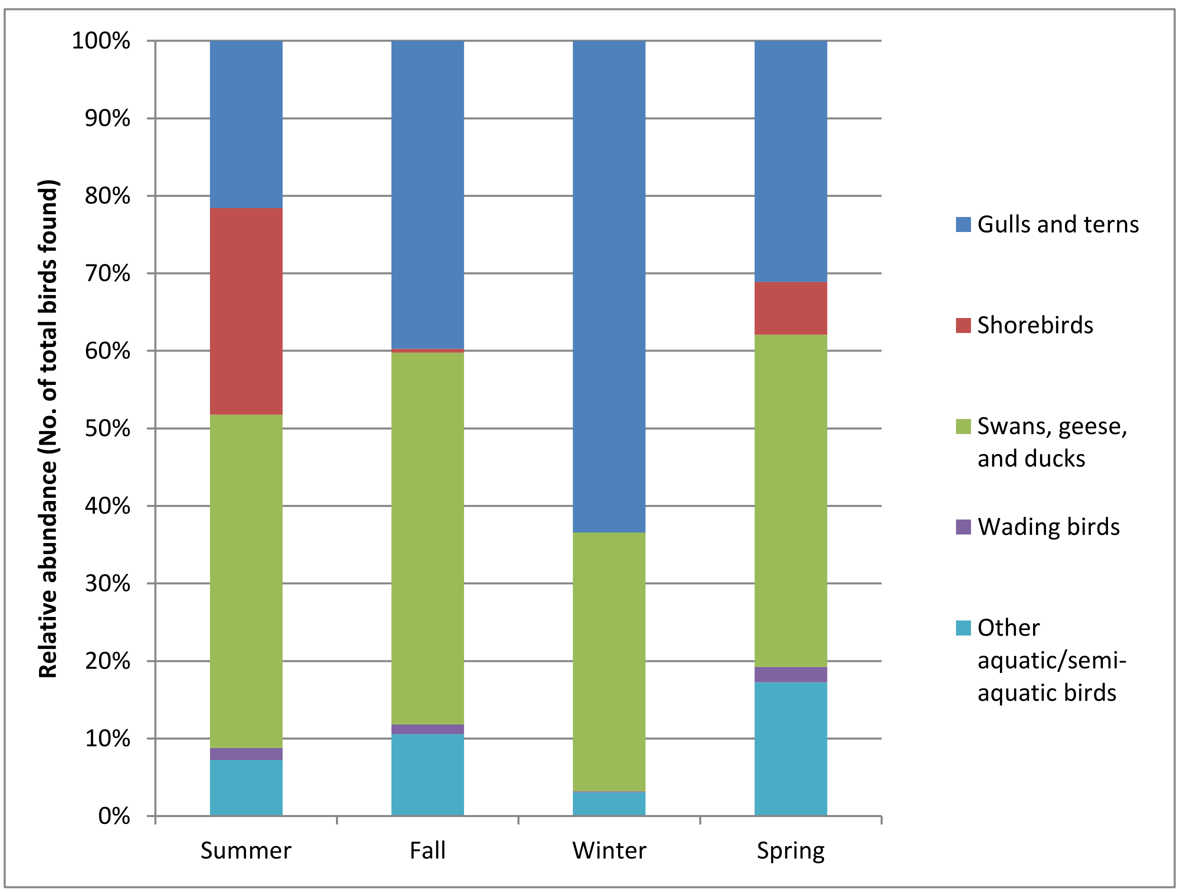

Figure 2-18. Relative abundance of aquatic- and semi-aquatic-feeding bird species observed in the LPRSA during the four 2010/2011 avian community surveys

2.4.1 Gulls and terns

Seven species of gulls were observed in the LPRSA during the 2010 and 2011 surveys (Table 2-5). Herring gull (*Larus argentatus*) and ring-billed gull (*Larus delawarensis*), along with most of the other gull species observed in the LPRSA, breed primarily in parts of the northern United States, Canada, or Alaska, although they may spend the winter or live year-round in the LPRSA. Numbers of gulls were highest in the fall and winter and lowest in the spring (Figure 2-18). Gulls were observed most frequently on water or on built structures. The ring-billed gull was the most frequently observed gull species, followed by the herring gull. Ring-billed gulls are opportunistic feeders and eat mostly insects, earthworms, fish, rodents, and grain (Pollet et al. 2012). Herring gulls are also opportunistic, but feed primarily on prey such as marine fishes and invertebrates (Pierotti and Good 1992).

The common tern (*Sterna hirundo*) was the only tern species seen during the 2010/2011 surveys, and it was infrequently observed (i.e., a total of five times) only in the spring and summer. The common tern feeds on small fish, crustaceans, and insects (Nisbet 2002). The species is currently designated as a species of special concern in New Jersey (NJDEP 2013).

The New Jersey Audubon Society conducted extensive mapping of the ranges of birds in the state from 1994 to 1997, including a survey to identify evidence of breeding (Walsh et al. 1999; as cited in Ludwig et al. 2010). Evidence of breeding gulls or terns was not observed in any of the four blocks surveyed in the vicinity of the LPRSA.

2.4.2 Swans, geese, and ducks

Of the 10 species of swans, geese, and ducks in the LPRSA, mallard (*Anas platyrhynchos*) and Canada goose (*Branta Canadensis*) were the most commonly observed during the 2010/2011 surveys. The mallard is the most abundant duck species in North America, and the Canada goose is the most widely distributed goose species in North America (Drilling et al. 2002; Mowbray et al. 2002). While both species may breed and overwinter in New Jersey (Drilling et al. 2002; Mowbray et al. 2002), the breeding bird survey conducted by the New Jersey Audubon Society (Walsh et al. 1999; as cited in Ludwig et al. 2010) found evidence confirming only the breeding of mallard in the vicinity of the LPRSA. However, USEPA Region 2 recently confirmed that breeding of Canada goose occurs on the Passaic River (USEPA 2015b).

Mallards are dabbling ducks, feeding opportunistically on insect larvae, invertebrates, plants, and aquatic vegetation (Drilling et al. 2002), whereas Canada goose are almost exclusively herbivorous (Mowbray et al. 2002). Mallard and Canada goose were most abundant in the summer and fall of the 2010/2011 surveys (Figure 2-18), and were present in greater numbers in the upper reaches of the LPRSA than in the lower reaches. Swans, geese, and ducks were observed in the LPRSA most frequently on water, but were also observed frequently on the shoreline.

2.4.3 Shorebirds

During the 2010/2011 surveys (Table 2-5), 13 shorebird species were observed. Shorebirds were most abundant in the summer and, to a lesser extent, in the spring (Figure 2-18); spotted sandpiper (*Actitis macularia*), least sandpiper (*Calidris minutilla*), and semipalmated sandpiper (*Calidris pusilla*) were the three shorebird species most frequently observed in these seasons. In the fall, few shorebirds were present, and none were observed in the winter.

The spotted sandpiper breeds throughout much of North America, including northern New Jersey, typically from May through August (Oring et al. 1997; Reed et al. 2013). During the winter, spotted sandpipers migrate to Central America, traveling as far south as Bermuda, Peru, and central Argentina (Oring et al. 1997; Reed et al. 2013). Least sandpiper and semipalmated sandpiper breed in subarctic tundra and far northern boreal forest, and winter in southern North America (least sandpiper) and South America (both species) (Hicklin and Gratto-Trevor 2010; Nebel and Cooper 2008). Walsh et al. (1999; as cited in Ludwig et al. 2010) found evidence of spotted sandpiper and killdeer (*Charadrius vociferus*) breeding in the survey block including Kearny Marsh (near the southern end of the Hackensack Meadowlands, approximately 1 mi from the LPRSA; Figure 2-2) during a breeding bird survey.

The spotted sandpiper diet consists of primarily aquatic invertebrates and insects, (including flying insects [e.g., midges and mayflies]), worms, fish, crustaceans, mollusks, and carrion (Oring et al. 1983; Rubbelke 1976), although they will eat almost any animal that is small enough (Rubbelke 1976). The diets of the least sandpiper and semipalmated sandpiper consist of benthic and terrestrial invertebrates (Hicklin and Gratto-Trevor 2010; Nebel and Cooper 2008). Sandpipers typically feed in intertidal areas and have been observed most frequently on mudflats or along the shoreline. Shorebirds were more abundant in the lower portion of the LPRSA than in the upper reaches during the 2010/2011 surveys.

Three shorebirds are listed by the State of New Jersey as species of special concern: sanderling (*Calidris alba*) (nonbreeding population), semipalmated sandpiper (nonbreeding population), and spotted sandpiper (breeding population) (NJDEP 2013).

2.4.4 Wading birds

Five species of wading birds were observed throughout the LPRSA in 2010/2011 (Table 2-5). Great blue heron was the most common species, followed by great egret (*Ardea alba*) and black-crowned night heron (*Nycticorax nycticorax*). Wading birds were observed relatively infrequently in summer, fall, and spring (< 2% of observations), and were observed very infrequently in winter (0.1% of observations) (Figure 2-18). Great blue heron was the only wading bird observed in the winter (n = 6). Wading birds were observed most often on the shoreline and vegetation, but were also frequently seen on water, mudflats, and built structures.

Great blue heron are abundant throughout most of North America; there are both migratory and non-migratory populations. In general, in the winter, great blue heron move south from their breeding areas in North America. However, some New Jersey populations are year-round, non-migratory residents that both overwinter and nest in the state (Butler 1992; Antonucci et al. 2008). No evidence of great blue heron breeding was found in the vicinity of the LPRSA in a breeding survey conducted by the New Jersey Audubon Society from 1994 to 1997 (Walsh et al. 1999; as cited in Ludwig et al. 2010), although it has been reported that in the summer, Kearny Marsh becomes a roost for large numbers of herons and egrets, including great blue herons (Boyle 2002). Great blue herons feed opportunistically on a variety of organisms, including fish, small mammals, reptiles, amphibians, insects, and crustaceans (Kushlan 1978; Butler 1993).

Great egrets are less widely distributed in North America than great blue heron; great egrets are present along the southeast, southern, and western edges of the United States, and throughout Central America (McCrimmon et al. 2011). The great egret is a migratory species, and may breed along the northeastern coast of North America (McCrimmon et al. 2011). The breeding survey conducted by the New Jersey Audubon Society from 1994 to 1997 (Walsh et al. 1999; as cited in Ludwig et al. 2010) did not find any evidence of great egret breeding in the vicinity of the LPRSA. However, great egret nesting sites have been reported in the New York Harbor area (Kerlinger 1997). In addition, great egrets from elsewhere have been reported to roost in the Kearny Marsh in the summer (Boyle 2002). Great egrets feed on small fish, other small vertebrates, and invertebrates (especially crustaceans) (McCrimmon et al. 2011).

Black-crowned night heron are widespread and breed throughout most of North America (Vennesland and Butler 2011; Hothem et al. 2010). During the winter, black-crowned night herons tend to migrate to the southern Atlantic coast and Caribbean shores, but some have been found to winter as far north as New England (Hothem et al. 2010). Kearny Marsh supports a black-crowned night heron colony and provides roosting and feeding habitat (USFWS 1997). Black-crowned night herons are opportunistic foragers with a varied diet that includes fish, aquatic and terrestrial insects, small birds, and crayfish (Hothem et al. 2010).

In addition to the great blue heron and the great egret, the snowy egret (*Egretta thula*) is known to roost in Kearny Marsh in the summer (Boyle 2002). The breeding bird survey conducted by the New Jersey Audubon Society from 1994 to 1997 (Walsh et al. 1999; as cited in Ludwig et al. 2010) found evidence indicating possible green heron (*Butorides virescens*) breeding in the vicinity of the LPRSA. Nesting sites for the snowy egret and green heron have been reported in the New York Harbor area as well (Kerlinger 1997). The diet of snowy egrets consists of approximately 75% fish and 25% crustaceans (Parsons and Master 2000). Green herons are opportunistic foragers with a varied diet that includes invertebrates, fish, crustaceans, insects, amphibians, reptiles, and rodents (Davis and Kushlan 1994).

The breeding population of black-crowned night heron is currently listed as threatened by the State of New Jersey because of a 90% population loss from the late 1970s to the late 1990s; the non-breeding population has special concern status (NJDEP 2013). The breeding population of great blue heron has special concern status in the State of New Jersey (NJDEP 2013).

2.4.5 Other aquatic-feeding birds

Other aquatic-feeding birds observed during the LPRSA 2010/2011 surveys were the fish-eating osprey, bald eagle, double-crested cormorant (*Phalacrocorax auritus*), and belted kingfisher (*Ceryle alcyon*). Information on each of these species is presented in the following section.

Osprey were observed in the summer (n = 13), fall (n = 12), and spring (n = 1), but not in the winter. The osprey is widely distributed throughout the world (NJDEP 2013). The species is restricted to areas near bodies of water that support adequate fish populations, because osprey feed almost exclusively on fish (Poole et al. 2002). Osprey are present in New Jersey during the breeding season; the majority of New Jersey osprey winter in northern South America (NJDEP 2013). No osprey nests have been observed along the LPRSA. The breeding population of osprey is currently listed as a threatened species by the State of New Jersey (NJDEP 2013).

Bald eagle were observed only in the fall (n = 2). The bald eagle is restricted to North America and resides year-round in New Jersey (NJDEP 2013). Bald eagles are usually found in close proximity to open water and require a nesting location that is safe from the threat of human disturbance. In 2012, the two bald eagle nests closest to the LPRSA were at least 10 mi away; one was located in Linden, approximately 10 mi southwest of the mouth of the Passaic River, and the other in Parsippany, approximately 16 mi west of RM 14 of the LPRSA (NJDEP 2012b). Some bald eagles that move south from the extreme northern part of their range may overwinter in New Jersey. The breeding population of bald eagles is currently listed as endangered by the State of New Jersey, and the non-breeding population is listed as threatened (NJDEP 2013).

Double-crested cormorants were observed relatively frequently during all four seasons, and were more common in the lower reaches of the LPRSA than in the upper reaches. Double-crested cormorants are common inhabitants of coastal areas and inland waters (Hatch and Weseloh 1999). Both breeding and overwintering habitats are present in New Jersey. In the New Jersey Audubon Society breeding survey conducted from 1994 to 1997 (Walsh et al. 1999; as cited in Ludwig et al. 2010), no evidence of double-crested cormorant nesting was found in the vicinity of the LPRSA. However, USEPA Region 2 recently confirmed that double-crested cormorants nest in Newark Bay (USEPA 2015b).

Belted kingfisher were observed more frequently in the summer and fall, and rarely in the winter and spring. Belted kingfisher were more often found upstream of RM 6 than in areas downstream of RM 6. The belted kingfisher is one of the most widespread land birds in North America (Kelly et al. 2009) and is present year-round in New Jersey, although uncommon in the winter (Boyle 2011). Their nesting distribution is limited by the availability of vertical banks for nest sites and a nearby food supply (Boyle 2002). In spring 2006, the USACE, New Jersey Department of Transportation (NJDOT), and National Oceanic and Atmospheric Administration (NOAA) conducted a survey to identify belted kingfisher burrows along the banks and riparian zones of the lower 16 mi of the LPRSA and the lower portions of several LPR tributaries (i.e., Second, Third, and Saddle Rivers), and to characterize the suitability of available habitat for breeding kingfishers based on US Fish and Wildlife Service (USFWS) habitat suitability index (HSI) models (Malcolm Pirnie et al. 2006). A total of nine kingfisher burrows were found along the LPRSA: two near RM 4, one at RM 7.5, one at RM 8.5, four between RM 11.1 and RM 11.4, and one at RM 13.1; however, none of the burrows were active, and most showed evidence of mammal use (Baron 2011). In general, belted kingfisher breeding habitat was found to be limited in the lower 6 mi of the Passaic River (Baron 2011).

2.4.6 Overall bird community

Gulls, geese, and ducks were the most commonly observed birds along the LPRSA in the 2009/2010 surveys. Shorebirds, wading birds (including herons/egrets), and other bird species (including piscivorous birds such as osprey, belted kingfisher, and double-crested cormorants) were less frequently observed. Avian habitat is limited within and along the LPRSA, and potential avian foraging areas include mudflats and patches of shoreline vegetation. Sediment-probing shorebirds use mudflat habitats along the LPRSA, and piscivorous birds (e.g., heron/egret and belted kingfisher) have been observed seasonally along the LPRSA and its tributaries, primarily on manmade structures (Windward 2014a, 2011a). There is some evidence that kingfishers may breed along the LPRSA (Baron 2011), and great blue herons are known to roost in nearby Kearny Marsh (approximately 1 mi from the LPRSA) (Boyle 2002). In addition, evidence of spotted sandpiper breeding was found by (Walsh et al. 1999; as cited in Ludwig et al. 2010) in the survey block including Kearny Marsh, although it has not been observed in the LPRSA.

2.5 MAMMALS

No surveys of water-associated mammals have been conducted to date in the LPRSA; however, few mammalian species have been noted as present in or near the LPRSA. Combined, approximately 4,500 hrs (including observations during sunrise and sunset) of habitat, avian, and aquatic species surveys were conducted, during which there was little evidence of aquatic mammalian species, likely due to limited suitable shoreline habitat. Potential foraging areas in the LPRSA include mudflats and patches of shoreline vegetation identified in habitat surveys (Iannuzzi and Ludwig 2004; Windward 2014b). Shelter is provided for some species by the forested banks above RM 9.5. Examples of mammals sighted in-water or on the banks of the LPRSA during previous surveys[42] include Eastern gray squirrel (*Sciurus carolinensis*), Eastern chipmunk (*Tamias striatus*), groundhog (*Marmota monax*), Norway rat (*Rattus norvegicus)*, domestic dog (*Canis lupus familiaris*), and feral cat (*Felis catus*). No current reports, either anecdotal or from surveys, were found of river otter (*Lutra canadensis*) in the LPRSA. The only recent report of river otter in the LPRSA was an individual animal observed during the late 1990s in the Hackensack Meadowlands that was believed to have escaped from captivity in a local zoo (Kiviat and MacDonald 2002). Muskrats (*Ondatra zibethicus*) are aquatic mammals that dig burrows in banks and feed primarily on vegetation. Raccoons (*Procyon lotor*) are likely to be present as well. Mink (*Neovison vison*) tracks were photographed near Dundee Dam, but a GIS analysis by Windward Environmental LLC (Windward) (Appendix I) indicated that there may be insufficient riparian tree and shrub cover in the LPRSA to provide the habitat necessary for a breeding population. Several species of seals (e.g., harbor [*Phoca vitulina*], gray [*Halichoerus grypus*], harp [*Pagophilus groenlandicus*], and hooded [*Cystophora cristata*]) winter in the NY/NJ Harbor, which is near the LPRSA. While it is unlikely that seals would spend significant time in the LPRSA, they may be infrequent visitors (USEPA 2015b).

2.6 AMPHIBIANS AND REPTILES

Surveys were not conducted specifically for amphibians and reptiles in the LPRSA. No amphibian species have been directly observed during sampling events in the LPRSA. Few reptiles have been directly observed in the LPRSA; eastern spiny softshell turtle (*Apalone spinifera spinifera*), red-eared slider (*Trachemys scripta elegans*), and common snapping turtle (*Chelydra serpentina*) were sighted during fish tissue sampling (Windward 2010c, e, j). Also, garter snakes (*Thamnophis* spp.) have been observed along the banks of the LPRSA during sampling by Windward. Wood turtle (*Glyptemys insculpta*) were sighted above Dundee Dam during fish tissue sampling (Windward 2012b). Several species of sea turtles could be found in the NY/NJ Harbor estuary, which is near the LPRSA. While it is unlikely that sea turtles would spend significant time in the LPRSA, they may be infrequent visitors, although the LPRSA would not provide adequate habitat or conditions to support this ecological group (USEPA 2015b).

Conditions in the LPRSA provide limited suitable habitat for some amphibian and reptile species, specifically in the small patches of marsh in the estuarine portion of the river and the wooded shorelines and riverfront parks in the freshwater section above RM 9.5. Suitable habitat for amphibians that require undisturbed vernal pools for breeding is limited in the LPRSA. In addition, roads and highways create barriers for species that migrate to and from breeding grounds separate from their adult habitat.

[42] Surveys did not target mammals, although mammalian species were sometimes noted incidentally. Examples of surveys wherein mammals were observed included the 2010 habitat and avian surveys (Windward 2014b, 2011a).

These factors were included in assessing the likelihood of a species' presence. Table 2-6 lists the amphibians and reptiles that could be present in the LPRSA and have aquatic diets or life stages. Additional information on reptile species observed in the LPSRA is presented in the remainder of this section.

Table 2-6. Amphibians and reptiles that potentially use the LPR

Species	Scientific Name
Eastern garter snake[a,b]	*Thamnophis sirtalis sirtalis*
Northern ringneck snake [b]	*Diadophis punctatus edwardsii*
Northern water snake[b]	*Nerodia sipedon*
Common snapping turtle[a,b]	*Chelydra serpentina*
Diamondback terrapin[b]	*Malaclemys terrapin*
Eastern painted turtle	*Chrysemys picta*
Eastern spiny softshell turtle[a,c]	*Apalone spinifera spinifera*
Redbelly turtle[d]	*Pseudemys rubriventris*
Red-eared slider[a,c]	*Trachemys scripta elegans*
Wood turtle[d]	*Glyptemys insculpta*
Bullfrog	*Rana catesbeiana*
Green frog[b]	*Rana clamitans melanota*
Northern cricket frog	*Acris crepitans*
Northern gray treefrog	*Hyla versicolor*
Northern two-lined salamander	*Eurycea bislineata*
Red-spotted newt	*Notophthalmus viridescens viridescens*

[a] Species observed during Windward fish sampling events.
[b] Observed in the Hackensack Meadowlands (Kiviat and MacDonald 2004).
[c] Introduced species (NJDFW 2013).
[d] State-listed threatened species (NJDFW 2013).
LPR – Lower Passaic River
NJDFW – New Jersey Division of Fish and Wildlife
Windward – Windward Environmental LLC

Eastern garter snakes (*Thamnophis sirtalis sirtalis*) prefer damp habitats near water; inhabit suburban landscapes; and eat frogs and toads, salamanders, fish, and earthworms. Garter snakes are found in various habitats; northern ringneck snakes (*Diadophis punctatus edwardsii*) prefer woodlands, are sometimes found in damp basements, and consume insects, slugs, and snails.

Common snapping turtles are common in Hackensack Meadows. Snapping turtles live in any permanent freshwater body, including rivers, but prefer muddy and vegetated bottoms. They are omnivorous, eating any live animal they can swallow, carrion, and aquatic plants (EOL 2013). The red-eared slider is an introduced species found throughout New Jersey. They prefer slow water with a muddy bottom but will use other habitats. The eastern spiny softshell, which is expanding from the south, is another introduced species of turtle.

2.7 THREATENED, ENDANGERED, AND SPECIAL STATUS SPECIES

State of New Jersey and USFWS conservation statuses were determined for those fish, birds, mammals, and reptiles or amphibians reported to use the LPRSA (for those groups that were surveyed), or that have the potential to use the study area (for those groups that were not surveyed). Nine bird species and one reptile species were listed by the State of New Jersey as either endangered, threatened, or of special concern (Table 2-7). One fish species is under evaluation by USFWS for listing as threatened.

Table 2-7. Conservation status for species reported or possibly present in the LPRSA

Common Name	Scientific Name	Status	Notes
Fish			
American eel[a]	*Anguilla rostrata*	under evaluation (federal)	In 2011, USFWS determined that the listing of American eel throughout its entire range may be warranted.
Birds			
Bald eagle[b]	*Haliaeetus leucocephalus*	endangered (New Jersey)	breeding population only
		threatened (New Jersey)	non-breeding population only
Black-crowned night heron	*Nycticorax nycticorax*	threatened (New Jersey)	breeding population only
		special concern (New Jersey)	non-breeding population only
Common tern	*Sterna hirundo*	special concern (New Jersey)	breeding population only
Great blue heron	*Ardea herodias*	special concern (New Jersey)	breeding population only
Osprey[b]	*Pandion haliaetus*	threatened (New Jersey)	breeding population only
Sanderling	*Calidris alba*	special concern (New Jersey)	non-breeding population only
Semipalmated sandpiper	*Calidris pusilla*	special concern (New Jersey)	non-breeding population only
Spotted sandpiper	*Actitis macularius*	special concern (New Jersey)	breeding population only
Snowy egret	*Egretta thula*	special concern (New Jersey)	breeding population only
Reptiles			
Wood turtle[c]	*Glyptemys insculpta*	threatened (New Jersey)	any population

a Frequently caught during 2009/2010 fish surveys (Windward 2010c, 2011c).

b Observed in LPRSA during 2010/2011 bird surveys (Windward 2019e, 2011a)

c Observed near the LPRSA (i.e., above Dundee Dam) during fish survey (Windward 2012b)

LPRSA – Lower Passaic River Study Area

USFWS – US Fish and Wildlife Service

Of the nine bird species, only the breeding population of the bald eagle was listed as endangered by the State of New Jersey, while the non-breeding population was listed as threatened (NJDEP 2013). Between 1970 and 1982, there was only one active bald eagle nest in New Jersey (Conserve Wildlife 2013a). As a result, a re-introduction project was conducted beginning in 1983, during which 60 young eagles were released over an 8-year period. This project was successful; in 2012, there were 119 nests with 165 young fledged (Conserve Wildlife 2013a) in the State of New Jersey. Bald eagle was removed from the USFWS endangered species list in 2007.

In 1999, the breeding population of black-crowned night heron was listed as threatened by the State of New Jersey; the non-breeding population was listed as a species of special concern (NJDEP 2013). The population declined from about 1,500 individuals in the late 1970s to only 200 in the late 1990s (Conserve Wildlife 2013b). This reduction was attributed to habitat destruction.

The breeding population of osprey was listed as threatened by the State of New Jersey in 1974 (NJDEP 2013). Prior to 1950, more than 500 nests had been found along the New Jersey coastline; in 1974, only 50 nests remained (Conserve Wildlife 2013c). Recovery began when the use of DDT was banned and management efforts were implemented to provide nest structures. Ongoing efforts have been conducted to monitor and manage the osprey population (Conserve Wildlife 2013c). In 2009, 486 nesting pairs were documented, and in 2012 it was estimated that the population was significantly more than 500 pairs (Clark and Wurst 2012).

Of the bird species reported or possibly present in the LPRSA, seven species have been listed by the State of New Jersey as species of special concern. This status applies to species that warrant special attention because of inherent vulnerability to environmental deterioration or habitat modification that would result in a threat to that species if conditions began to (or continued to) deteriorate (NJDEP 2013).

The American eel is currently under evaluation for listing as a federally threatened species under the Endangered Species Act. In September 2011, USFWS announced a 90-day finding on a petition to list the species as threatened in the *Federal Register* (76 FR 60431). USFWS stated that the status of threatened may be warranted based on a causal link between oceanic changes and global warming (i.e., increasing sea surface temperature with a corresponding shift in spawning location, decreasing food availability, or a shift in the transport of the larval stage by currents) and decreasing American eel recruitment. USFWS initiated a status review at the time of the 90-day finding to determine whether listing the American eel as threatened is warranted.

The wood turtle was listed as a threatened species by the State of New Jersey in 1979 because of population declines due to habitat loss and stream degradation (Conserve Wildlife 2013d). The wood turtle was observed upstream of Dundee Dam during the 2012 fish community survey (Windward 2012b), but its presence in the LPRSA itself has not been confirmed.

3 Summary of Problem Formulation

As described in Section 1, the LPR has been industrialized and urbanized for more than two centuries, having served as the receiving environment for industrial and municipal waste discharges since the 19th century. As a result, the LPR has been subjected to a broad range of contaminant loadings from multiple sources (e.g., untreated industrial and municipal wastewater, CSOs/ SWOs, direct runoff, and atmospheric deposition) for a long time. Its distinguishing factor is elevated levels of 2,3,7,8-TCDD, which is atypical among urban sites. The Lister Avenue site, which was a significant source of 2,3,7,8-TCDD, was identified as OU-1 of the Diamond Alkali Superfund site. The objective of this BERA is to identify unacceptable risks posed by site-related chemicals to ecological receptors in the LPRSA.

This section presents a summary of the ecological problem formulation, including a description of the ecological CSM. This section summarizes and updates the USEPA-approved PFD (Windward and AECOM 2009), based on the data collected by CPG as part of the RI, including the biological surveys conducted by CPG under USEPA oversight. Recognizing the unique characteristics of the LPRSA is critical to developing an accurate understanding of ecological and human receptors and their potential interactions with environmental media. Site-specific factors—including urbanization, mixed land uses, non-chemical stressors, hardened/altered shorelines, and the estuarine environment—influence receptors in the LPRSA and the pathways of exposure to site-related contamination.

To determine which organisms to assess for potential ecological risk, it is critical to understand the setting and habitat types within and adjacent to the river. As described in Section 2, the ecological environment of the LPRSA is typical of urban systems, with reduced habitat quality and increased urban inputs (Baron 2011; Germano & Associates 2005; Iannuzzi and Ludwig 2004; Iannuzzi et al. 2008; Ludwig et al. 2010; Windward and AECOM 2009, 2015). The quality of LPRSA ecological habitat is severely impaired. The historical and current industrial use and residential development of the shoreline have limited the shoreline habitats. The lower portion of the LPRSA (RM 0 to approximately RM 8) is largely characterized by a developed shoreline with structures abutting industrial properties; above RM 8, the LPRSA is characterized predominately by mixed vegetation abutting roads, parks, and residential properties.

3.1 Ecological Receptors

As described in the USEPA-approved PFD (Windward and AECOM 2009), preliminary representative receptor species were selected and approved by USEPA based on the biological surveys and other information (e.g., habitat data) on the

LPRSA and the surrounding area. Factors considered in the selection of receptor species include the following:

- Potential for exposure to contaminated site sediments
- Relative ability to bioaccumulate/biomagnify site-related chemicals
- Societal and cultural significance (including species highly valued by society)
- Ecological significance (including species serving a unique ecological function)
- Sensitivity to site-related chemicals

The USEPA-approved ecological receptor groups selected for evaluation in this BERAinclude the following:

- Benthic invertebrate community
- Macroinvertebrate populations (i.e., blue crabs[43])
- Mollusk populations (i.e., ribbed and freshwater mussels)
- Fish populations (i.e., mummichog, other forage fish [banded killifish/darter], white perch, channel catfish, brown bullhead, American eel, and largemouth bass)[44]
- Bird populations (i.e., mallard duck,[45] spotted sandpiper, heron/egret, and belted kingfisher)
- Mammal populations (i.e., river otter and mink)
- Amphibian/reptile populations
- Aquatic plant community
- Zooplankton community

The species that were evaluated for these receptor groups are summarized in Table 3-1.

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[43]Crayfish were identified in the PFD (Windward and AECOM 2009) as representing freshwater macroinvertebrate populations. However, per agreement between CPG and USEPA (Windward 2010g), blue crab were the only species used to represent the macroinvertebrate population, for both the estuarine and freshwater portions of the LPRSA, because of the limited number of crayfish collected in the freshwater portion.

[44] Common carp, white catfish, white sucker, smallmouth bass, and northern pike were also collected during the late summer/early fall 2009 sampling effort, and were evaluated in this BERA as additional fish species, consistent with 2017 communications between CPG and USEPA.

[45] The mallard duck was not proposed to be quantitatively evaluated because the potential exposure to chemicals was expected to be greater for other higher-trophic-level avian species (i.e., invertivores and piscivores).

Table 3-1. Selected ecological receptors

Receptor Group	Feeding Guild	Receptor Species
Benthic invertebrate community	na	multiple infaunal species, including *Hyalella azteca*, *Chironomus dilutus*, *Ampelisca abdita,* polychaetes (i.e., *Nereis virens*), and oligochaetes (i.e., *Lumbriculus variegatus*)
Macroinvertebrate populations	na	blue crab[a]
Mollusk populations	na	ribbed mussel and freshwater mussel
Fish populations	benthic omnivore (SFF)	mummichog, other SFF (e.g., banded killifish/darter), and common carp
	invertivore	white perch, channel catfish, brown bullhead, white catfish, and white sucker
	piscivore	American eel, largemouth bass, smallmouth bass, and northern pike
Bird populations	aquatic herbivore	mallard duck[b]
	sediment-probing invertivore	spotted sandpiper
	migratory piscivore	heron/egret species[c]
	resident piscivore	belted kingfisher
Mammal populations	piscivore[d]	river otter and mink
Zooplankton community	na	multiple species
Amphibian/reptile populations	na	multiple species[e]
Aquatic plant community	na	multiple species, including submerged macrophytes[f]

a Crayfish were identified in the PFD (Windward and AECOM 2009) as representing freshwater macroinvertebrate populations. However, per agreement between CPG and USEPA (Windward 2010g), blue crab were the only species used to represent the macroinvertebrate population, for both the estuarine and freshwater portions of the LPRSA, because of the limited number of crayfish collected in the freshwater portion.

b In the PFD (Windward and AECOM 2009), the mallard duck was not proposed to be a quantitatively evaluated receptor because the potential exposure to chemicals was expected to be greater for other higher-trophic-level avian receptors (i.e., invertivores and piscivores).

c Herons/egrets were evaluated as both migratory and resident species using two different SUFs.

d The selected semi-piscivorous mammal (i.e., river otter) is expected to be protective of herbivorous and omnivorous mammals (e.g., muskrat) because piscivorous and omnivorous mammals feed on organisms that are higher on the food chain. The mink was also assessed since possible mink tracks were observed near Dundee Dam during the CPG LPRSA biological surveys.

e Amphibians and reptiles have a limited presence in the estuarine portion of the LPRSA.

f The aquatic plant community in the LPRSA is limited by the physical development of the shoreline and poor light penetration of the water.

CPG – Cooperating Parties Group
LPRSA – Lower Passaic River Study Area
na – not applicable
PFD – problem formulation document
SFF – small forage fish
SUF – site use factor
USEPA – US Environmental Protection Agency

Exposure of ecological receptors to chemicals can be through contact (e.g., direct contact of benthic organisms to sediment), ingestion of water or sediments, or ingestion of contaminated prey. Several of the ecological receptors in the LPRSA (e.g., spotted sandpiper) utilize mudflats. In tidal rivers such as the LPR, intertidal and shallow subtidal areas are important and productive habitats. Many ecological receptors, including spotted sandpiper and wading birds, feed primarily along mudflats and other shallow areas. Forage fish, which serve as a food source for larger fish, mammals and birds, also utilize shallow water areas for feeding and refuge. Potential chemical exposure pathways were evaluated for all receptors (Table 3-1) to determine which pathways would be evaluated.

3.2 Assessment and Measurement Endpoints

Assessment endpoints, risk questions, and measurement endpoints were used to define the evaluation of ecological risks, consistent with the USEPA-approved PFD (Windward and AECOM 2009). USEPA (1998) defines assessment endpoints as "explicit expressions of the actual environmental value that is to be protected, operationally defined by an ecological entity and its attributes." This BERA is based on community- or population-level assessments and evaluates the assessment endpoints and selected receptor groups presented in Table 3-2. Table 3-2 also presents the risk questions, measurement endpoints (modified with additional details from the assessment endpoint table presented in the PFD), data use objectives for each measurement endpoint, and types of abiotic and biotic data that were used for the risk evaluation.

Table 3-2. Ecological assessment endpoints for the LPRSA

Testable Risk Question	Description of Measurement Endpoint	Data Use Objective	LPRSA Data to be Used to Derive Exposure Concentrations
Assessment Endpoint No. 1—Maintenance of the zooplankton community that serves as a food base for juvenile fish **Selected Receptor Group—**Zooplankton community (multiple species represented)			
Are COPEC concentrations in surface water in the LPRSA at levels that might affect the maintenance of the zooplankton community as a food resource for fish?	chemical concentrations in surface water collected from relevant exposure areas as compared with toxicity-based values (i.e., aquatic thresholds)	estimating the exposure of zooplankton to chemicals in surface water via various exposure pathways	surface water chemistry and conventional (i.e., physical) parameters from relevant exposure areas based on the 2011-2012 CPG sampling efforts and any additional data that meet DQOs[a]
Assessment Endpoint No. 2—Protection and maintenance (i.e., survival, growth, and reproduction) of the benthic invertebrate community, both as an environmental resource in itself and as one that serves as a forage base for fish and wildlife populations **Selected Receptor Group—**Benthic invertebrate community (multiple infaunal species represented)			
Are benthic communities different from those found in similar nearby water bodies, where chemical concentrations are at background levels?	community structure data (e.g., total invertebrate abundance, species richness, and abundance of species or specific taxonomic groups) as compared with appropriate reference information[b] datasets using diversity indices and multivariate and spatial statistical techniques; to be used as part of the benthic invertebrate SQT approach	assessing adverse effects of LPRSA chemicals on the benthic invertebrate community via various exposure pathways; evaluating reference information[b] and physical/biological stressors	benthic invertebrate community data based on taxonomy data collected during fall 2009 and spring and summer 2010 and any additional data that meet DQOs[a]
Are COPEC residues in benthic invertebrate tissues from the LPRSA at levels that might cause an adverse effect on survival, growth, and/or reproduction of infaunal invertebrates?	chemical concentrations in laboratory-exposed benthic infaunal invertebrate tissues (*Nereis virens* in the estuarine portion and *Lumbriculus variegatus* in the freshwater portion) exposed to LPRSA sediment in 28-day bioaccumulation tests as compared with CTR	assessing adverse effects of LPRSA chemicals on benthic infaunal invertebrates; developing a FWM for higher organisms	whole-body infaunal benthic invertebrate tissue from laboratory bioaccumulation tests based on LPRSA surface sediment collected during fall 2009 and any additional data that meet DQOs[a]
Are COPEC concentrations in LPRSA sediments from the biologically active zone at levels that might cause an adverse effect on survival, growth, and/or reproduction of the benthic invertebrate community?	chemical concentrations in sediment as compared with toxicity-based sediment quality values from the literature that are specific to benthic invertebrates; to be used as part of the benthic invertebrate SQT approach	estimating the exposure of benthic invertebrates to chemicals in sediment via various exposure pathways	surface (0 to 15 cm) sediment chemistry and conventional parameters based on 2008-2012 LPRSA surface sediment data, and any additional data that meet DQOs[a]
	laboratory bioassay tests (28-day survival and growth of *Hyalella azteca* throughout the LPRSA, 10-day survival and growth of *Chironomus dilutus* in the freshwater portion, and 10-day survival of *Ampelisca abdita* in the estuarine portion) using LPRSA sediment compared with control and reference information;[b] to be used as part of the benthic invertebrate SQT approach	assessing adverse effects of LPRSA chemicals in sediment on benthic invertebrates via various exposure pathways; evaluating reference information[b] and physical/biological stressors	toxicity tests based on surface (0 to 15 cm) sediment collected during fall 2009 and any additional data that meet DQOs[a]

Table 3-2. Ecological assessment endpoints for the LPRSA

Testable Risk Question	Description of Measurement Endpoint	Data Use Objective	LPRSA Data to be Used to Derive Exposure Concentrations
Are COPEC concentrations in surface water from the LPRSA at levels that might cause an adverse effect on survival, growth, and/or reproduction of the benthic invertebrate community?	chemical concentrations in surface water collected from relevant benthic invertebrate exposure areas as compared with toxicity-based values (i.e., aquatic thresholds)	estimating the exposure of benthic invertebrates to chemicals in surface water via various exposure pathways	surface water chemistry and conventional parameters from relevant exposure areas (e.g., near-bottom) based on the 2011-2012 sampling efforts and any additional data that meet DQOs[a]
Assessment Endpoint No. 3—Protection and maintenance (i.e., survival, growth, and reproduction) of healthy populations of blue crab and crayfish that serve as a forage base for fish and wildlife populations and as a base for sports fisheries **Selected Receptor Group**—Decapods (blue crab)			
Are COPEC residues in benthic macroinvertebrate tissues from the LPRSA at levels that might cause an adverse effect on survival, growth, and/or reproduction of macroinvertebrate (blue crab and crayfish) populations in the LPRSA?	chemical concentrations in site-collected benthic macroinvertebrate whole-body tissue (i.e., crab) as compared with literature-based CTR	estimating the exposure of benthic macroinvertebrates to chemicals via various exposure pathways; developing a FWM	whole-body benthic macroinvertebrate tissue of blue crab collected from the late summer/early fall 2009 sampling effort and any additional data that meet DQOs[a]
Are COPEC concentrations in LPRSA sediments from the biologically active zone at levels that might cause an adverse effect on survival, growth, and/or reproduction of macroinvertebrate populations?	chemical concentrations in sediment as compared with toxicity-based sediment quality values from the literature that are specific to benthic macroinvertebrates	estimating the exposure of benthic invertebrates to chemicals in sediment via various exposure pathways	surface (0 to 15 cm) sediment chemistry and conventional parameters based on 2008-2012 LPRSA surface sediment data, and any additional data that meet DQOs[a]
Are COPEC concentrations in surface water from the LPRSA at levels that might cause an adverse effect on survival, growth, and/or reproduction of macroinvertebrate populations?	chemical concentrations in surface water collected from relevant benthic macroinvertebrate exposure areas as compared with toxicity-based values (i.e., aquatic thresholds)	estimating the exposure of benthic macroinvertebrates to chemicals in surface water via various exposure pathways	surface water chemistry and conventional parameters from relevant exposure areas (e.g., near-bottom) based on the 2011-2012 sampling efforts and any additional data that meet DQOs[a]
Assessment Endpoint No. 4—Protection and maintenance (i.e., survival, growth, and reproduction) of healthy mollusk populations **Selected Receptor Group**—Bivalves (multiple species represented)			
Are COPEC residues in bivalve mollusk tissues from the LPRSA at levels that might cause an adverse effect on survival, growth, and/or reproduction of mollusk populations in the LPRSA?	chemical concentrations in tissue from *in situ* caged bivalves (ribbed mussel [*Geukensia demissa*] and freshwater mussel (*Elliptio complanata*])	assessing adverse effects of LPRSA chemicals on bivalves; developing a FWM	whole-body bivalve mollusk tissue of selected test bivalve species

Table 3-2. Ecological assessment endpoints for the LPRSA

Testable Risk Question	Description of Measurement Endpoint	Data Use Objective	LPRSA Data to be Used to Derive Exposure Concentrations
Are COPEC concentrations in LPRSA sediments from the biologically active zone at levels that might cause an adverse effect on survival, growth, and/or reproduction of mollusk populations?	chemical concentrations in sediment as compared with toxicity-based sediment quality values from the literature that are specific to bivalve mollusks	estimating the exposure of bivalve mollusks to chemicals in sediment via various exposure pathways	surface (0 to 15 cm) sediment chemistry and conventional parameters based on 2008-2012 LPRSA surface sediment data, and any additional data that meet DQOs[a]
Are COPEC concentrations in surface water from the LPRSA at levels that might cause an adverse effect on survival, growth, and/or reproduction of mollusk populations?	chemical concentrations in surface water collected from relevant bivalve mollusk exposure areas as compared with toxicity-based values (i.e., aquatic thresholds)	estimating the exposure of bivalve mollusks to chemicals in surface water via various exposure pathways	surface water chemistry and conventional (e.g., near-bottom) parameters from relevant exposure areas based on the 2011-2012 sampling efforts and any additional data that meet DQOs[a]
Assessment Endpoint No. 5—Protection and maintenance (i.e., survival, growth, and reproduction) of omnivorous, invertivorous, and piscivorous fish populations that serve as a forage base for fish and wildlife populations and as a base for sports fisheries **Selected Receptor Groups—**Benthic omnivore: mummichog, banded killifish/darter, common carp (a non-native species). Invertivore: white perch, channel catfish, brown bullhead, white catfish, white sucker. Piscivore: American eel, largemouth bass, northern pike, smallmouth bass			
Are COPEC concentrations in fish tissue from the LPRSA at levels that might cause an adverse effect on survival, growth, and/or reproduction of populations of fish that use the LPRSA?	chemical concentrations or toxic equivalencies measured in site-collected fish whole-body tissue (and estimated egg tissue based on egg lipid data) as compared with literature-based CTR; exposure areas and SUFs based on potential LPRSA habitat and where fish are present in LPRSA per fish community surveys	estimating the exposure of selected fish species, and other fish species that prey upon those organisms, to chemicals via various exposure pathways; evaluating background levels and physical/biological stressors as part of risk characterization to help make informed risk management decisions	whole-body fish tissue based on: fish collected in late summer/early fall 2009 and summer 2010, and any additional data that meet DQOs;[a] LPRSA mummichog egg lipid content collected in 2010; whole-body tissue concentrations for several selected fish species using the methods presented in the Data Usability Plan (Windward and AECOM 2015)
	prey taxonomy identified in selected LPRSA fish species	defining the exposure parameters (e.g., diet, trophic level) and prey composition of fish species within the LPRSA	fish stomach prey taxonomy based on regional literature; LPRSA-specific data are not available because of the limited number of fish collected in the late summer/early fall 2009 (Windward 2010a).
	physical and biological information based on gross internal/external fish health observations; histopathology of selected fish species may also be evaluated per USEPA direction	assisting in the interpretation of the results in terms of fish population health	gross internal/external health observations based on LPRSA fish community data collected in 2009 and 2010
	literature-based information on fish trophic feeding level and habitat use of selected LPRSA fish species	defining the exposure parameters (e.g., diet, trophic level) and exposure areas (e.g., habitat identification and stratification) for selected fish species within the LPRSA	LPRSA fish community data collected in 2009 and 2010; literature search[c]

Table 3-2. Ecological assessment endpoints for the LPRSA

Testable Risk Question	Description of Measurement Endpoint	Data Use Objective	LPRSA Data to be Used to Derive Exposure Concentrations
Are modeled dietary exposures to COPECs from LPRSA prey at levels that might cause an adverse effect on survival, growth, and/or reproduction of fish populations that use the LPRSA?	species-specific modeled daily doses of COPECs (estimated from surface sediment and prey [invertebrate and fish] tissue chemistry[d]) as compared with literature-based dietary effect thresholds; exposure areas and SUFs will be based on potential LPRSA habitat and where fish are present in LPRSA per fish community surveys; LPRSA water temperature data will be used to determine fish ingestion rates	estimating the exposure of selected fish species to chemicals via the dietary exposure pathway	surface (0 to 15 cm) sediment chemistry from relevant exposure areas and benthic invertebrate and fish prey (or representative prey) tissue; sediment data based on LPRSA surface sediment collected from 2008 to 2012, and any additional data that meet DQOs;[a] tissue data based on invertebrate and fish tissue collected from the late summer/early fall 2009 sampling effort and any additional data that meet DQOs[a]
Are COPEC concentrations in surface water from the LPRSA at levels that might cause an adverse effect on survival, growth, and/or reproduction of fish populations that use the LPRSA?	chemical concentrations in surface water collected from relevant fish exposure areas as compared with literature-based toxicity values (i.e., aquatic thresholds); exposure areas and SUFs will be based on potential LPRSA habitat	estimating the exposure of selected fish species to chemicals in surface water via various exposure pathways	surface water chemistry from relevant exposure areas based on the 2011-2012 sampling efforts and any additional data that meet DQOs[a]
What are the egg numbers (or mass) from estuarine benthic omnivores (i.e., mummichog) from the LPRSA?	egg counts (or mass) in selected gravid mummichog	assisting in the interpretation of the results in terms of fish population health	LPRSA mummichog eggs from selected gravid females collected in 2010
Assessment Endpoint No. 6—Protection and maintenance (i.e., survival, growth, and reproduction[f]) of herbivorous, omnivorous,[g] sediment-probing, and piscivorous bird populations; use of LPRSA habitat for breeding used to determine the relative weight for the bird egg measurement endpoint **Selected Receptor Groups—**Aquatic herbivore: mallard duck; sediment-probing invertivore: spotted sandpiper; migratory piscivore:[h] heron/egret; resident piscivore: belted kingfisher			
Are modeled dietary doses of COPECs based on LPRSA biota, sediment, and surface water and/or modeled piscivorous bird egg tissues based on LPRSA fish at levels that might cause an adverse effect on survival, growth, and/or reproduction of bird populations that use the LPRSA?	species-specific modeled daily doses (estimated from surface water, surface sediment, and prey [invertebrate and fish] tissue chemistry) as compared with literature-based dietary dose effect thresholds; modeled piscivorous bird egg tissue-residue concentrations (estimated from fish prey tissue chemistry using dietary dose/maternal transfer model) as compared with literature-based bird egg tissue-residue effect thresholds; exposure areas and SUFs will be based on potential LPRSA habitat areas and presence of species per avian community surveys	estimating the exposure of selected bird species to chemicals in surface water, sediment, and prey tissue[i] via various exposure pathways; developing a FWM	surface (0 to 15 cm) sediment and surface water chemistry from relevant exposure areas and benthic invertebrate and fish prey (or representative prey) tissue; based on surface sediment data from 2008 to 2012, surface water data from 2011 to 2012, and tissue data from 2009 to 2010

Table 3-2. Ecological assessment endpoints for the LPRSA

Testable Risk Question	Description of Measurement Endpoint	Data Use Objective	LPRSA Data to be Used to Derive Exposure Concentrations
Assessment Endpoint No. 7—Protection and maintenance (i.e., survival, growth, and reproduction) of aquatic mammal populations **Selected Receptor Group—**Piscivore: river otter			
Are modeled dietary doses of COPECs based on LPRSA biota, sediment, and surface water at levels that might cause an adverse effect on survival, growth, and/or reproduction of aquatic mammal populations that use the LPRSA?	Focal species-specific modeled daily doses (estimated from surface water, surface sediment, and prey [invertebrate and fish] tissue chemistry) as compared with literature-based dietary dose effect thresholds; exposure areas and SUFs will be based on potential LPRSA habitat areas	estimating the exposure of selected mammal species to chemicals in surface water, sediment, and prey tissue via various exposure pathways; developing a FWM	surface (0 to 15 cm) sediment and surface water chemistry from relevant exposure areas and benthic invertebrate and fish prey (or representative prey) tissue; based on surface sediment data from 2008 to 2012, surface water data from 2011 to 2012, and tissue data from 2009 to 2010
Assessment Endpoint No. 8—Maintenance of healthy aquatic plant populations as a food resource and habitat for fish and wildlife populations **Selected Receptor Group—**Aquatic plant populations (multiple species represented)			
Are COPEC concentrations in surface sediment and/or surface water in the LPRSA at levels that might affect the maintenance of healthy aquatic plant populations as a food resource and habitat to fish and wildlife?	chemical concentrations in surface water and/or sediment collected from relevant aquatic plant exposure areas as compared with toxicity-based values (i.e., aquatic thresholds); exposure areas will be based on potential LPRSA habitat	estimating the exposure of aquatic plants to chemicals in surface sediment and/or surface water via direct contact with chemicals in sediment and water	surface (0 to 15 cm) sediment and surface water chemistry and conventional parameters from relevant exposure areas; surface water data based on 2011-2012 sampling effort(s) and any additional data that meet DQOs[a]
Assessment Endpoint No. 9—Protection and maintenance (i.e., survival, growth, and reproduction) of healthy amphibian and reptile populations **Selected Receptor Group—**Amphibian (early-life stage) and reptile populations (multiple species represented)			
Are COPEC concentrations in surface water and/or surface sediment from LPRSA at levels that might cause an adverse effect on the survival, growth, and/or reproduction of amphibian and reptile populations that use the LPRSA?	chemical concentrations in surface water and/or sediment collected from relevant amphibian and/or reptile exposure areas as compared with available toxicity-based values (i.e., aquatic thresholds); exposure areas will be based on potential LPRSA habitat	estimating the exposure of amphibian and reptiles to chemicals in surface sediment and/or surface water via direct contact	surface (0 to 15 cm) sediment and surface water chemistry and conventional parameters from relevant exposure areas; surface water data based on 2011-2012 sampling efforts and any additional data that meet DQOs[a]

Notes: Assessment endpoints as presented in the PFD (Windward and AECOM 2009). Although each endpoint focuses on chemical exposure, additional data will be collected on conventional parameters (e.g., grain size) to help in ecosystem characterization as part of the risk characterization for risk management decisions.

[a] Any additional current LPRSA data that meet the risk assessment-specific DQOs described in the data usability plan (Windward and AECOM 2015) could also be used to estimate exposure.

[b] The terminology presented in the PFD (Windward and AECOM 2009) was changed from “regional background levels” to “background and reference information” for consistency with the terminology and definition provided by USEPA (USEPA 2013b).

[c] Additional physical and biological information collected during the fish community surveys (e.g., gross internal/external health observations) will also be used in the risk assessment to assist in the interpretation of the results in terms of fish population health.

[d] For chemicals that are metabolized or otherwise regulated by fish, a tissue residue approach is not appropriate; therefore, a dietary model will be used as a LOE for evaluating risks to fish from metabolized or otherwise regulated chemicals.

[e] Surface water will not be incorporated into the fish dietary assessment, as WIRs for fish are largely unavailable, and fish toxicity studies that measure both food and water ingestion of chemicals are very limited.

[f] Given that few aquatic-feeding birds currently use the LPRSA for breeding because of habitat constraints, the reproduction assessment endpoint for birds will evaluate whether the existing chemical concentrations would impact reproduction if suitable habitat were present.

[g] Consistent with the PFD (Windward and AECOM 2009), omnivorous birds were not identified in the CSM as a feeding guild to be quantitatively evaluated. A representative species was not selected because the evaluation of other avian feeding guilds (i.e., sediment-probing and piscivorous birds) will be protective of omnivorous birds.

[h] Herons/egrets were evaluated as both migratory and resident species.

[i] Additional biological information collected during the bird community surveys will also be used in the risk assessment to assist in the interpretation of the results in terms of avian population health.

BERA – baseline ecological risk assessment
COPEC – chemical of potential ecological concern
CPG – Cooperating Parties Group
CSM – conceptual site model
CTR – critical tissue residue
DQO – data quality objective
FWM – food web model
LOE – line of evidence
LPRSA – Lower Passaic River Study Area
PFD – problem formulation document
SQT – sediment quality triad
SUF – site use factor
USEPA – US Environmental Protection Agency
WIR – water ingestion rate

The assessment endpoints were evaluated in the following order in this BERA:

- **Benthic invertebrate community –** Protection and maintenance (i.e., survival, growth, and reproduction) of the benthic invertebrate community, both as an environmental resource in itself and as one that serves as a forage base for fish and wildlife populations
- **Blue crab –** Protection and maintenance (i.e., survival, growth, and reproduction) of healthy populations of blue crab and crayfish[46] that serve as a forage base for fish and wildlife populations and as a base for sports fisheries
- **Mollusks –** Protection and maintenance (i.e., survival, growth, and reproduction) of healthy mollusk populations
- **Fish –** Protection and maintenance (i.e., survival, growth, and reproduction) of omnivorous, invertivorous, and piscivorous fish populations that serve as a forage base for fish and wildlife populations and as a base for sports fisheries
- **Birds –** Protection and maintenance (i.e., survival, growth, and reproduction[47]) of herbivorous, omnivorous,[48] sediment-probing, and piscivorous bird populations; use of LPR habitat for breeding used to determine the relative weight for the bird egg measurement endpoint
- **Mammals –** Protection and maintenance (i.e., survival, growth, and reproduction) of aquatic mammal populations
- **Zooplankton –** Maintenance of the zooplankton community that serves as a food base for juvenile fish
- **Amphibians/Reptiles –** Protection and maintenance (i.e., survival, growth, and reproduction) of healthy amphibian and reptile populations
- **Aquatic plants –** Maintenance of healthy aquatic plant populations as a food resource and habitat for fish and wildlife populations

[46] Crayfish were identified in the PFD (Windward and AECOM 2009) as representing freshwater macroinvertebrate populations. However, per agreement between CPG and USEPA (Windward 2010g) blue crab were the only species used to represent the macroinvertebrate population, for both the estuarine and freshwater portions of the LPRSA, because of the limited number of crayfish collected in the freshwater portion.

[47] Few aquatic-feeding birds currently use the LPRSA for breeding because of habitat constraints. The reproduction assessment endpoint for birds evaluates whether existing chemical concentrations could impact reproduction if suitable habitat were present.

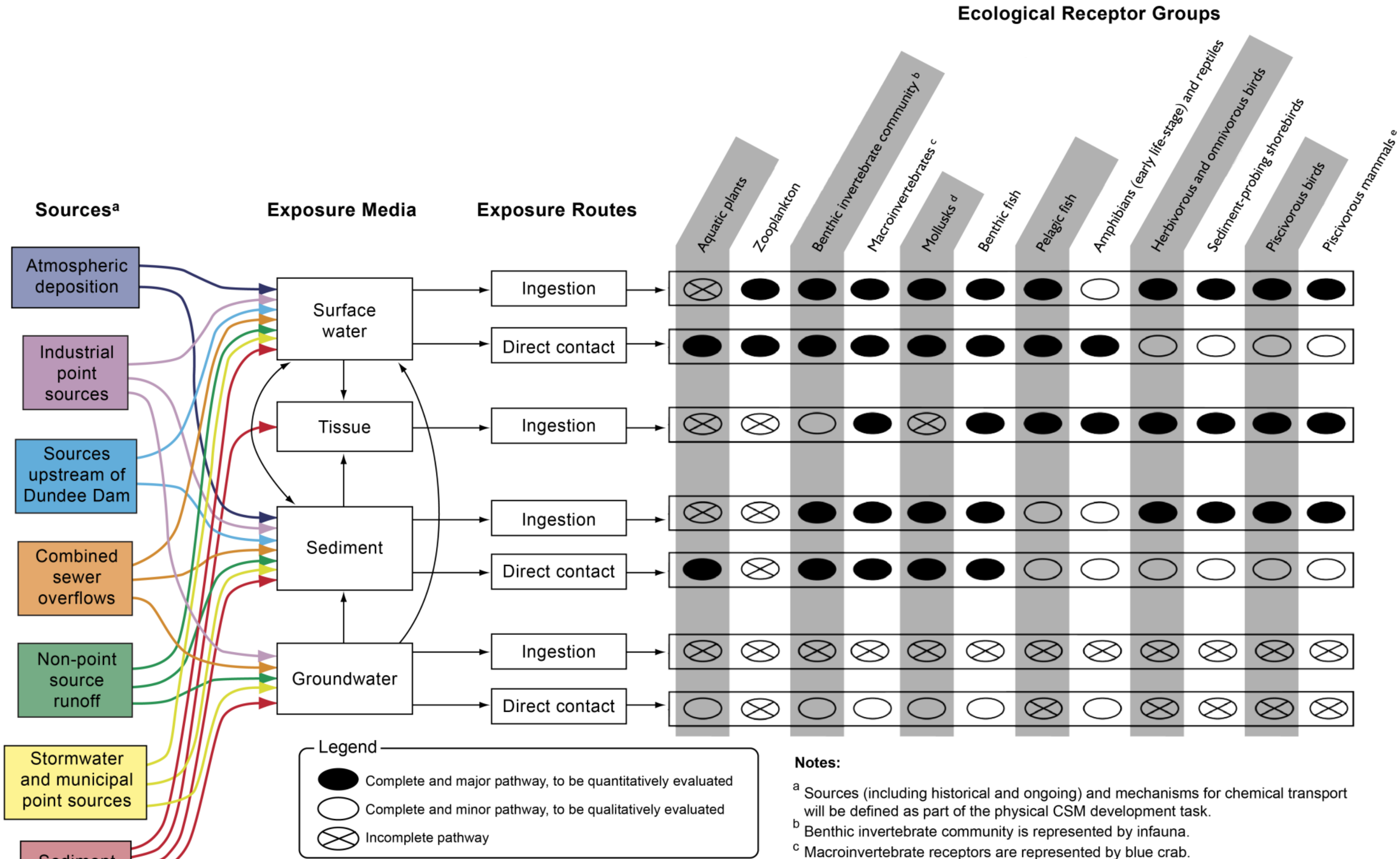
[48] Consistent with the PFD (Windward and AECOM 2009), omnivorous birds were not identified in the CSM as a feeding guild to be quantitatively evaluated. A representative species was not selected because the evaluation of other avian feeding guilds (i.e., sediment-probing and piscivorous birds) is protective of omnivorous birds.

3.3 Ecological Conceptual Site Model

An ecological CSM is used to describe the pathways by which contaminants move from sources, including those resulting from human activities, to ecological receptors at a site. The USEPA-approved ecological CSM (Windward and AECOM 2009) for the LPRSA is based on site-specific information about species typically present at the site or similar urbanized river systems and potential exposure pathways. This BERA reflects an updated CSM based on the current understanding (using data collected to date) of the connection among the pathways, exposure areas, and the overall ecological system in the LPRSA.

3.3.1 Ecological exposure pathways

The general ecological CSM is presented on Figure 3-1. Receptors were evaluated according to the area(s) where they were found or expected to be found (in some cases, the entire LPRSA).



Notes:

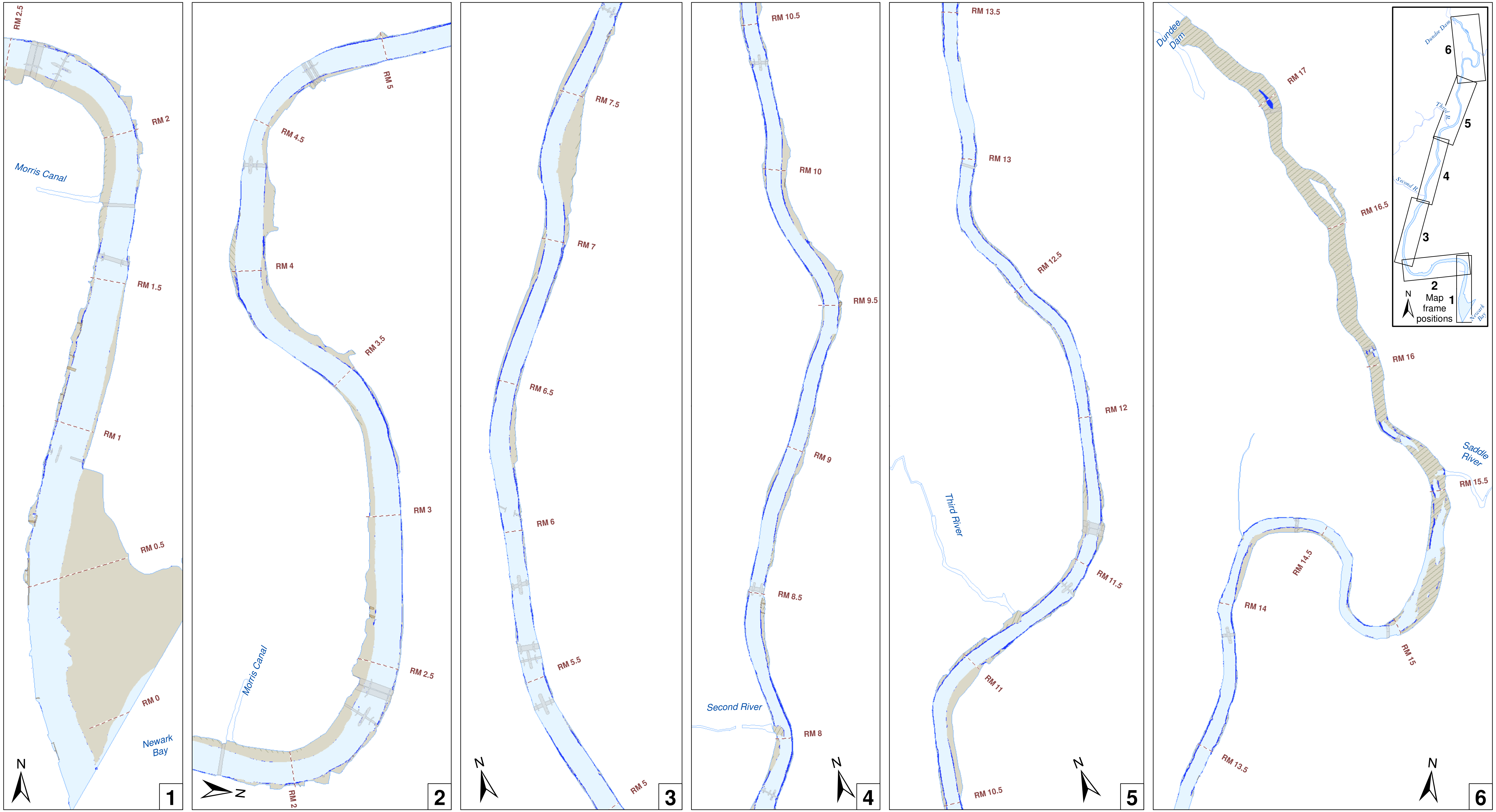
[a] Sources (including historical and ongoing) and mechanisms for chemical transport will be defined as part of the physical CSM development task.
[b] Benthic invertebrate community is represented by infauna.
[c] Macroinvertebrate receptors are represented by blue crab.
[d] Mollusk receptors are represented by bivalves.
[e] Piscivorous mammals are expected to be protective of omnivorous and herbivorous mammals. Therefore, no receptors were selected for those feeding guilds.

Figure 3-1.General ecological CSM for the LPRSA

3.3.2 Conceptual site model

Figure 3-1 presents exposure pathways inclusive of both estuarine and freshwater organisms. As discussed in Section 2.1.1.1, the LPRSA is typically evaluated based on three general salinity zones: upper estuarine (RM 0 to RM 4), fluvial estuarine (RM 4 to RM 13), and tidal freshwater (RM 13 to RM 17.4) (Figure 2-2). The boundaries of these general salinity zones are qualitative because the location of the salt wedge is influenced by freshwater input and tidal flows, as well as the system geometry. Daily and seasonal variations in salinity in the fluvial estuarine zone of the LPRSA have a significant impact upon the benthic invertebrate community.

Specific salinity zones were not developed for assessing fish. Unlike the benthic invertebrate community, fish and crab communities generally use the river regardless of prescribed salinity zones; estuarine fish, including American eel and white perch, were found throughout the LPRSA, and freshwater fish such as common carp and catfish (i.e., brown bullhead, channel catfish, and white catfish) were found down to RM 2. SFF were generally found in shallow, nearshore habitat (e.g., mudflats); thus, these areas also provide preferential feeding habitat for piscivorous fish, birds, and mammals (Figure 3-2). Fish species-specific exposure areas were determined based on where the organisms were found (see Section 7.1.2 for a discussion of fish exposure areas). Bird and mammal exposure areas (see Sections 8.1.2 and 9.1.2, respectively) were determined independent of salinity.



Mudflat areas for ecological receptors are those areas where the river bottom slope is ≤ 6° and the depth is ≥ -2 ft MLLW, based on the 2007 CPG bathymetric survey.

Figure 3-2. Shallows and shallow flats in the LPRSA

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4 Data Evaluation and Reduction

This section provides a summary of the criteria used for establishing acceptable chemistry and toxicity datasets for use in this BERA (i.e., DQOs); a description of the sediment, tissue, surface water, and biological survey data considered acceptable for use in this BERA (collected by CPG under USEPA oversight); and the methods used for reducing chemistry data for risk calculations. Additional details on the data evaluation criteria and data reduction methods are presented in the *Data Usability and Data Evaluation Plan for the Lower Passaic River Study Area Risk Assessments* (Windward and AECOM 2015), hereafter referred to as the Data Usability Plan.

4.1 Data Quality Objectives

Data used to define potential exposure and/or estimate potential risks (i.e., exposure point concentrations [EPCs] based on chemistry data or metrics based on toxicity or community data) underwent an evaluation to determine if the data quality was appropriate for the intended data use and therefore met the DQOs (Windward and AECOM 2015). Data that did not meet the DQOs for use in the derivation of EPCs in the risk assessment may still be evaluated for other aspects of the LPRSA RI/FS, such as site characterization, nature and extent, trend analysis of chemical concentrations over time, background evaluation, and modeling (Windward and AECOM 2015). The DQO review process was consistent with USEPA risk guidance (USEPA 1992). Five general levels for defining/applying DQOs were identified: event, location, sample, result, and validation. These DQOs are outlined in Table 4-1 and detailed in the *Data Usability and Data Evaluation Plan for the LPRSA Risk Assessments* (Windward and AECOM 2015). Only those data that met the specified DQOs were used in this BERA.

Table 4-1. DQOs for the BERA dataset

Event Level
DQO No. 1 – Original hard copies or electronic copies of data report(s) must be available.
DQO No. 2 – Data must represent current conditions.
Location Level
DQO No. 1 – Sediment cannot be collected from dredged[a] or capped areas.
DQO No. 2 – Field coordinates must be available to verify where data were collected.
Sample Level
DQO No. 1 – Sample depth interval must be identified.
DQO No. 2 – Sample and/or analysis type must be clearly identified.
Result Level
DQO No. 1 – DLs must be appropriately reported.
DQO No. 2 – Constituent parameters for summations must be available.
DQO No. 3 – Chemical analytical methods must be acceptable.
DQO No. 4 – Toxicity and bioaccumulation test methods must be acceptable.

Table 4-1. DQOs for the BERA dataset

DQO No. 5 – Invertebrate community data must be reported to the lowest practical taxonomic level.
DQO No. 6 – Benthic invertebrate community metric calculations must be documented.
Validation Level
DQO No. 1 – Chemistry data must be validated and include validation qualifiers, or sufficient information must be available to validate data.
DQO No. 2 – Sufficient information must be available to confirm the quality of the biological test data.
DQO No. 3 – Sufficient information must be available to confirm the quality and comparability of the taxonomic data.
DQO No. 4 – Chemistry data reports must contain laboratory-generated forms that include results for each sample.
DQO No. 5 – Existence and location of documentation that supports the dataset must be known.

[a] Includes dredged areas that have been backfilled with clean material.

BERA – baseline ecological risk assessment

DL – detection limit

DQO – data quality objective

All chemistry and toxicity data collected during sampling events implemented by CPG since the beginning of the CPG-led LPRSA RI (initiated in 2007) were considered for use in this BERA for the calculation of risk estimates. CPG-led QAPPs specified DQOs that were consistent with USEPA guidance to ensure that the data collected were of sufficient quality to support the RI, including the risk assessments. During the December 14 and 16, 2010, meetings between USEPA and CPG representatives, it was agreed that the EPCs in the risk assessments would be calculated using only current (i.e., CPG) data that met the DQOs specified in Table 4-1.[49] All data collected by CPG with the intention of being used in the risk assessments (detailed in Section 4.2) were considered appropriate for the calculation of risk estimates and met the DQOs outlined in Table 4-1, with the exception of the following:

- Surface sediment data collected from all areas that have since been dredged (i.e., RM 10.9 and Lister Avenue site dredge areas)[50]
- Subsurface sediment data (i.e., data from sediment below the sediment depth associated with ecological exposure)[51]

[49] Older data (collected prior to the initiation of the CPG-led LPRSA RI) may be considered when evaluating nature and extent and time-related trends.

[50] These areas were excluded because sediment has been dredged; therefore, samples do not represent current conditions, as required by location-level DQO No. 1 (Table 4-1).

[51] These data were excluded because only sediment data collected from the depth interval of 0 to 15 cm (0 to 6 in.) below the sediment surface were considered acceptable for inclusion in both risk assessments, as required by sample-level DQO No. 1 (Windward and AECOM 2015).

4.2 DATA USED IN THE BERA

This section describes the surface sediment chemistry data, surface sediment toxicity data, tissue chemistry data, water chemistry data, and data from various biological surveys used in this BERA. Per the agreement between USEPA and CPG, only data collected by CPG (under USEPA oversight) since 2007 were considered to be representative of current conditions within the LPRSA, and only these data were used in deriving exposure concentrations in the risk assessments.

Data used to define background conditions were based on data collected by CPG (above Dundee Dam since 2007) per the USEPA-approved benthic QAPP (Windward 2012a) and regional data collected by other parties from Jamaica Bay and the Mullica River as directed by USEPA (USEPA 2013b). Comparable sediment chemistry, toxicity, biological survey, and tissue data were not available from Jamaica Bay and the Mullica River after 2006, except for data from one location in Mullica River/Great Bay that was sampled in 2010 as part of the National Coast Condition Assessment (NCCA) program (Table 4.2). The most recent data from Jamaica Bay (1993 to 2005) and the Mullica River (1995 to 2006 and 2010) were used to define regional background conditions. Appendix K provides the LPRSA BERA dataset, and Appendix L provides the background and reference dataset. While the chemistry data used to describe background conditions are the most recent available, the use of dated Jamaica Bay and Mullica River data adds some level of uncertainty and may impact the background comparison evaluation. That uncertainty, however, does not impact the conditions in the LPRSA or the potential COCs/risk drivers at the site.

4.2.1 Sediment chemistry data

Only sediment data collected from the depth interval of 0 to 15 cm (0 to 6 in.) below the sediment surface were included for the derivation of risk estimates and in the evaluation of background. Surface sediment chemistry data used in this BERA were from samples collected during eight sampling events from 2008 to 2013, as follows (Table 4-2, Figure 4-1):

- 2008 low-resolution coring (LRC) sampling[52]
- 2009 surface sediment sampling[53]
- 2010 surface sediment sampling (co-located with tissue samples)
- 2011 RM 10.9 sediment characterization sampling[54]

[52] One location from the RM 10.9 dredge area was excluded.

[53] Two locations from the RM 10.9 dredge area and one location from the Lister Avenue site dredge area were excluded.

[54] Twenty-two locations from the RM 10.9 dredge area were excluded.

- 2012 RM 10.9 sediment characterization sampling[55]
- 2012 LRC supplemental sampling program
- 2012 RM 10.9 sediment investigation
- 2013 LRC supplemental investigation

Background datasets were developed for both freshwater and estuarine sediment chemistry:

- Freshwater background concentrations (i.e., urban habitat) were derived from sediment samples collected by CPG from above Dundee Dam in 2008 and 2012 (Figure 4-2).
- Two datasets were developed for estuarine background using regional data collected by non-CPG parties. Sediment samples collected from Jamaica Bay (New York) from 1993 to 2005 were used to derive background estuarine sediment concentrations (Figure 4-3) representative of a similarly urban environment. Sediment samples collected from the Mullica River and Great Bay from 1999 to 2006 were used to derive estuarine background sediment concentrations (rural habitat) (Figure 4-4). Only sediment chemistry data with co-located toxicity data were used. The regional background datasets were identified by USEPA for use in this BERA as background and reference datasets.

4.2.2 Sediment toxicity data

LPRSA sediment toxicity test data used in this BERA were from 98 of the 107 surface (0 to 15 cm) sediment locations sampled in 2009 for sediment chemistry (Table 4-2, Figure 4-5).[56] Of these 98 LPRSA sediment chemistry and toxicity samples, 97 were also analyzed for benthic invertebrate community indices to support the SQT evaluation. As discussed in Section 2.2.1, the LPRSA benthic invertebrate community is dominated by deposit feeders and detritivores (e.g., annelid worms, chironomids, and bivalves).

The following toxicity tests were conducted using LPRSA sediment:

- 10-day amphipod (*Ampelisca abdita*) survival (27 estuarine locations)
- 10-day midge (*Chironomus dilutus*) survival and growth (71 freshwater locations)

[55] Five locations from the RM 10.9 dredge area were excluded.

[56] Two sediment quality triad (SQT) sampling locations were in the RM 10.9 dredge area, and one SQT sampling location was in the Lister Avenue site dredge area. These three SQT sampling locations are not in the chemistry dataset but have been retained for SQT analysis to understand relationships among synoptic data collected in 2009 (before dredging).

- 28-day amphipod (*Hyalella azteca*) survival and growth test (98 locations; 27 estuarine and 71 freshwater)

For the evaluation of reference information, toxicity test data included 24 surface (0 to 15 cm) sediment samples with co-located sediment chemistry and benthic invertebrate community data (i.e., SQT locations) collected from above Dundee Dam in 2012 (Figure 4-6). Reference information also included toxicity test data for surface (0 to 15 cm) sediment samples with co-located chemistry samples collected from Jamaica Bay from 1999 to 2005, and the Mullica River and Great Bay from 1999 to 2006 (Figures 4-3 and 4-4, respectively). Toxicity test reference information was available for the following:

- 10-day amphipod (*A. abdita*) survival (66 Jamaica Bay locations and 20 Mullica River and Great Bay locations)
- 10-day midge (*C. dilutus*) survival and growth (24 locations above Dundee Dam)
- 28-day amphipod (*H. azteca*) survival and growth (24 locations above Dundee Dam)

Table 4-2. Sediment chemistry and toxicity data included in the BERA dataset

Sampling Event	Sampling Period	Description	Number of Locations	Chemical Group/Toxicity Test	Source
LPRSA					
2008 LRC program	July to December 2008	LRC/sediment sampling throughout 17.4-mi LPRSA and tributaries; only surface (i.e., 0 to 15 cm) sediment data used	98	metals, SEM metals, butyltins, PAHs, alkylated PAHs, SVOCs, PCB Aroclors, PCB congeners, PCDDs/PCDFs, organochlorine pesticides, herbicides, VOCs, TPH, general chemistry (i.e., AVS, ammonia, cyanide, Kjeldahl nitrogen, phosphorus, and total sulfide), TOC, and grain size	AECOM (2014a)
2009 benthic sediment sampling	October to November 2009	surface (0 to 15 cm) sediment grab samples collected throughout 17.4-mi LPRSA	107	metals, SEM metals, butyltins, PAHs, alkylated PAHs, SVOCs, PCB Aroclors, PCB congeners, PCDDs/PCDFs, organochlorine pesticides, herbicides, VOCs, TPH, general chemistry (i.e., AVS, ammonia, cyanide, Kjeldahl nitrogen, phosphorus, and total sulfide), TOC, and grain size	Windward (2015a)
			98[a, b]	whole-sediment toxicity tests using the following methods: 10-day *A. abdita* survival (estuarine sediment), 10-day *C. dilutus* survival and growth (freshwater sediment), 28-day *H. azteca* survival and growth (estuarine and freshwater sediment)	Windward (2018f)
2010 benthic sediment sampling	August 2010	surface (0 to 15 cm) sediment grab samples collected at locations where SFF were collected	21	metals, SEM metals, butyltins, PAHs, alkylated PAHs, SVOCs, PCB Aroclors, PCB congeners, PCDDs/PCDFs, organochlorine pesticides, herbicides, VOCs, TPH, general chemistry (i.e., AVS, ammonia, cyanide, Kjeldahl nitrogen, phosphorus, and total sulfide), TOC, and grain size	Windward (2015a)
2011 RM 10.9 sediment investigation	August to November 2011	coring/sediment sampling in the vicinity of RM 10.9, only surface (i.e., 0 to 15 cm) sediment data used	32	metals, SEM metals, butyltins, PAHs, alkylated PAHs, SVOCs, PCB Aroclors, PCB congeners, PCDDs/PCDFs, organochlorine pesticides, herbicides, VOCs, TPH, general chemistry (i.e., AVS, ammonia, cyanide, Kjeldahl nitrogen, phosphorus, and total sulfide), TOC, and grain size	CH2M HILL (2013)
2012 LRC supplemental investigation	January to February 2012	LRC/sediment sampling throughout 17.4-mi LPRSA; only surface (i.e., 0 to 15 cm) sediment data used	85	metals, SEM metals, butyltins, PAHs, alkylated PAHs, SVOCs, PCBs Aroclors, PCB congeners, PCDDs/PCDFs, organochlorine pesticides, VOCs, TPH, general chemistry (i.e., AVS, ammonia, cyanide, Kjeldahl nitrogen, phosphorus, and total sulfide), TOC, and grain size	AECOM (2013b)
2012 RM 10.9 sediment investigation (Addendum A)	May 2012	sediment cores collected in the vicinity of RM 10.9; only surface (i.e., 0 to 15 cm) sediment data used	10	metals, butyltins, PAHs, alkylated PAHs, SVOCs, PCB Aroclors, PCB congeners, PCDDs/PCDFs, organochlorine pesticides, VOCs, TPH, general chemistry (i.e., cyanide), TOC, and grain size	ddms (2013f); CH2M HILL (2013)

Table 4-2. Sediment chemistry and toxicity data included in the BERA dataset

Sampling Event	Sampling Period	Description	Number of Locations	Chemical Group/Toxicity Test	Source
2013 LRC supplemental investigation 2	September to October 2013	LRC/sediment sampling from RM 7 to RM 15 in the LPRSA; only surface (i.e., 0 to 15 cm) sediment data used	75	metals, SEM metals, butyltins, PAHs, alkylated PAHs, SVOCs, PCB congeners, PCDDs/PCDFs, organochlorine pesticides, VOCs, TPH, general chemistry (i.e., AVS, ammonia, cyanide, Kjeldahl nitrogen, phosphorus, and total sulfide), TOC, and grain size	AECOM (Draft)
Passaic River Above Dundee Dam					
2008 LRC program	September to October 2008	LRC/sediment sampling above Dundee Dam	6	metals, SEM metals, butyltins, PAHs, alkylated PAHs, SVOCs, PCBs Aroclors, PCB congeners, PCDDs/PCDFs, organochlorine pesticides, herbicides, VOCs, TPH, general chemistry (i.e., AVS, ammonia, cyanide, Kjeldahl nitrogen, phosphorus, and total sulfide), TOC, and grain size DFs, SVOCs, TPH, VOCs, and wet chemistry	AECOM (2014a)
2012 upstream sediment sampling	November 2012	surface (0 to 15 cm) sediment grab samples collected above Dundee Dam	40	metals, SEM metals, butyltins, PAHs, alkylated PAHs, SVOCs, PCBs Aroclors, PCB congeners, PCDDs/PCDFs, organochlorine pesticides, VOCs, TPH, general chemistry (i.e., AVS, ammonia, cyanide, Kjeldahl nitrogen, total phosphorus, and sulfide), TOC, and grain size	Windward (2019d)
			24	toxicity tests using the following methods: 10-day *C dilutus* survival and growth, 28-day *H. azteca* survival and growth	Windward (2018d)
Jamaica Bay					
1999 CARP sediment ambient study	August 1999	surface sediment grab samples (0 to 10 cm) collected from Jamaica Bay	1	metals, PAHs, alkylated PAHs, SVOCs, PCB congeners, PCDDs/PCDFs, organochlorine pesticides, TOC, grain size	NOAA (2013) and (USEPA 2016h)[c]
				toxicity test based on 10-day *A. abdita* survival	
2000 to 2005 NCA Program New York/New Jersey Harbor	August 2000 to August 2005	surface sediment grab samples (0 to 2 cm) collected from Jamaica Bay	9	metals, PAHs, SVOCs, PCB congeners, organochlorine pesticides, TOC, and grain size	
				sediment toxicity test based on 10-day *A. abdita* survival	
1993 to 2003 REMAP	September 1993 to August 1998 and July to September 2003	surface sediment grab samples (0 to 2 cm) collected from Jamaica Bay	84	metals, SEM metals, PAHs, SVOCs, PCDDs/PCDFs, organochlorine pesticides, general chemistry (e.g., AVS, ammonia, cyanide, Kjeldahl nitrogen, total sulfide), TOC, grain size	
				sediment toxicity test based on 10-day *A. abdita* survival	

Table 4-2. Sediment chemistry and toxicity data included in the BERA dataset

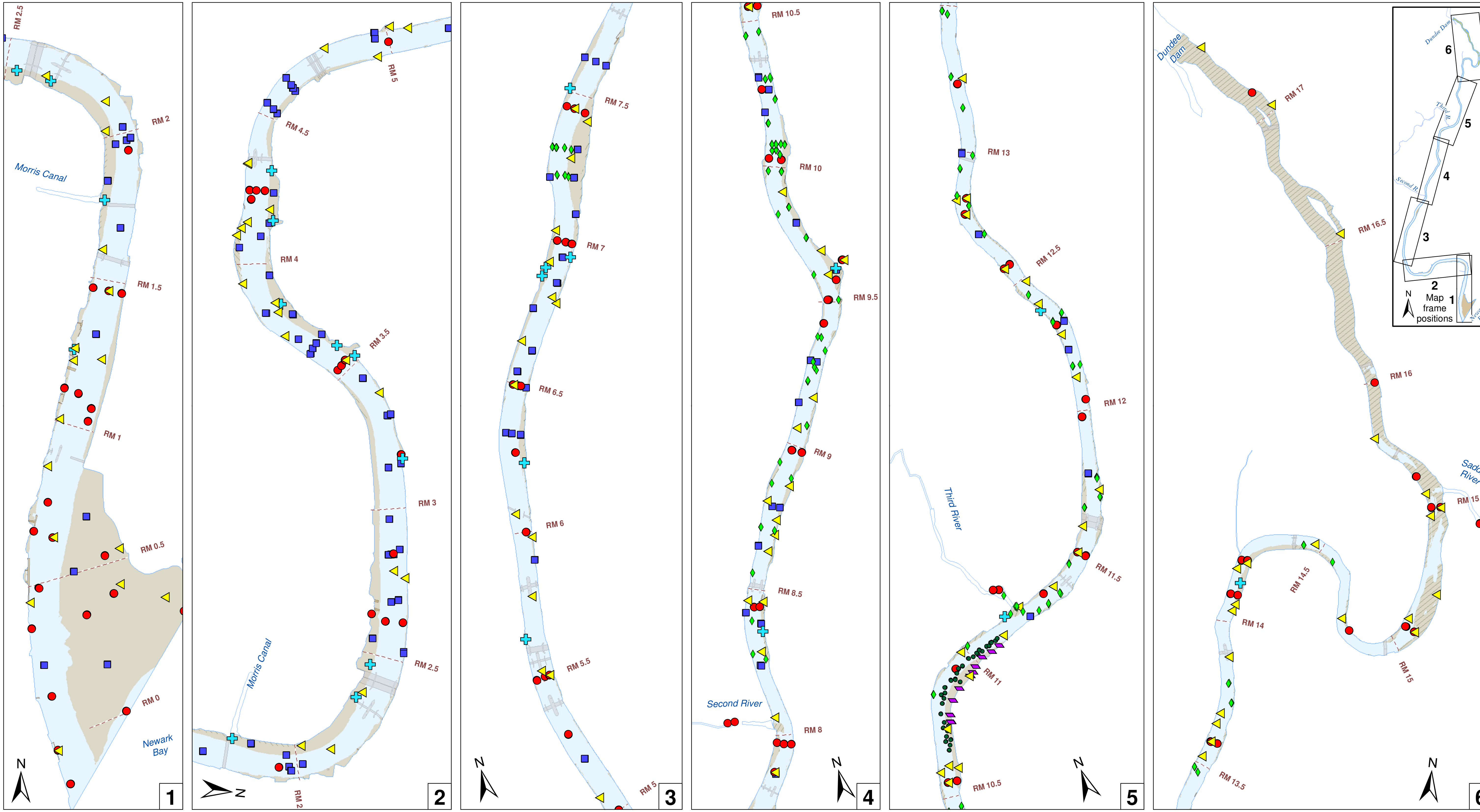
Sampling Event	Sampling Period	Description	Number of Locations	Chemical Group/Toxicity Test	Source
Mullica River and Great Bay					
1999 late summer/early fall RI-ESP sampling program	October 1999	surface sediment grab samples (0 to 15 cm) collected from Mullica River and Great Bay	3	metals, PAHs, PCB Aroclors, PCDDs/PCDFs, and organochlorine pesticides	NOAA (2013)
				toxicity test based on 10-day *A. abdita* survival	
1990 to 1991 EMAP-Delaware Bay	August 1990 to July 1991	surface sediment grab samples (0 to 2 cm) collected from Mullica River and Great Bay	4	metals, PAHs, SVOCs, PCB congeners, and organochlorine pesticides	NOAA (2013)
				toxicity test based on 10-day *A. abdita* survival	
2000 to 2006 NCA Program New Jersey Atlantic Coast	September 2000 to August 2006	surface sediment grab samples (0 to 2 cm) collected from Mullica River and Great Bay	17	metals, PAHs, SVOCs, PCB congeners, organochlorine pesticides, TOC, and grain size	NOAA (2013), USEPA (2016e)
				toxicity test based on 10-day *A. abdita* survival	
2010 NCCA Program	August 2010	surface sediment grab sample (0 to 2 cm) collected from Mullica River and Great Bay	1	metals, PAHs, SVOCs PCB congeners, organochlorine pesticides, TOC, and grain size	USEPA (2016f)
				toxicity test based on 10-day *A. abdita* survival	

[a] Two SQT sampling locations were in the RM 10.9 dredge area (LPRT11E and LPRT11G), and one SQT sampling location was in the Lister Avenue site dredge area (LPRT03G). These three SQT sampling locations are not in the chemistry dataset but have been retained for SQT analysis to understand relationships among synoptic data collected in 2009 (before dredging).

[b] The sediment from one location was collected for chemistry analysis and toxicity testing only (LPRT16B); samples from 97 locations with chemistry and toxicity test data were co-located with benthic invertebrate community survey samples.

[c] The 2003 REMAP data are only available from USEPA (2016h).

AVS – acid volatile sulfide
BERA – baseline ecological risk assessment
CARP – Contaminant Assessment and Reduction Project
EMAP – Environmental Monitoring and Assessment Program
ESP – ecological sampling program
LPRSA – Lower Passaic River Study Area
LRC – low-resolution coring
NCA – National Coastal Assessment
NCCA – National Coastal Condition Assessment
NOAA – National Oceanic and Atmospheric Administration
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
PCDD – polychlorinated dibenzo-*p*-dioxin
PCDF – polychlorinated dibenzofuran
REMAP – Regional Environmental Monitoring and Assessment Program
RI – remedial investigation
RM – river mile
SEM – simultaneously extracted metals
SFF – small forage fish
SQT – sediment quality triad
SVOC – semivolatile organic compound
TOC – total organic carbon
TPH – total petroleum hydrocarbons
VOC – volatile organic compound
Windward – Windward Environmental LLC



Event
- 2013 LRC Supplemental Investigation 2
- 2012 RM 10.9 Sediment Investigation (Addendum A)
- 2012 LRC Supplemental Investigation
- 2011 RM 10.9 Sediment Investigation
- 2010 Benthic Sediment Sampling
- 2009 Benthic Sediment Sampling
- 2008 LRC Program

- Bridge
- Abutment
- Dock
- River mile
- Mudflat
- Gravel with fines
- LPRSA

0 0.25 0.5 Kilometers

0 0.25 0.5 Miles

Mudflat areas for ecological receptors are those areas where the river bottom slope is ≤ 6° and the depth is ≥ -2 ft MLLW, based on the 2007 CPG bathymetric survey.

Figure 4-1. LPRSA locations for surface sediment chemistry samples

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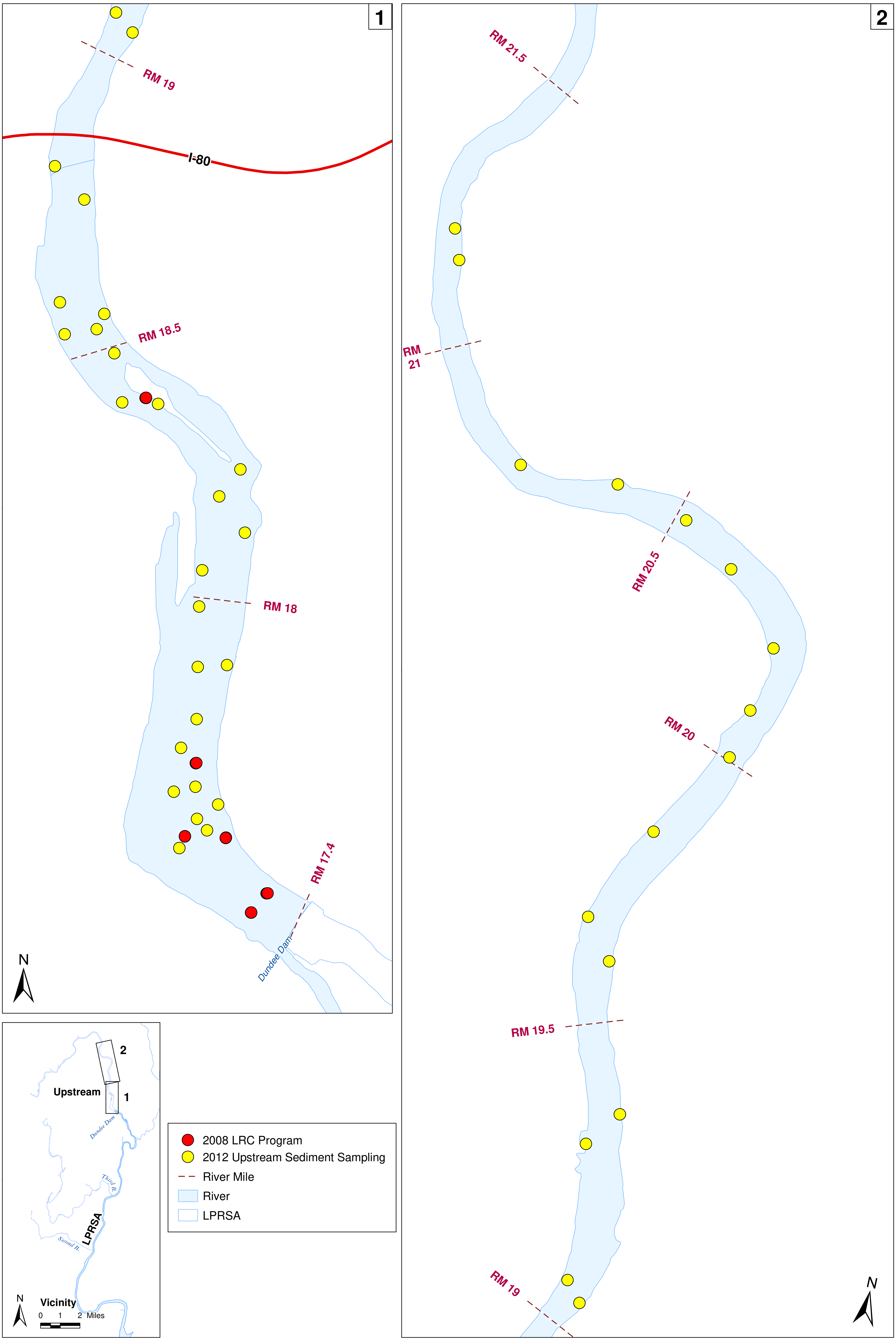

Figure 4-2. Locations above Dundee Dam for surface sediment chemistry samples
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1993-2003 REMAP
1998-1999 CARP Sediment Ambient Study
2000-2005 NCA Program New York/New Jersey Harbor

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Figure 4-3. Jamaica Bay locations for surface sediment chemistry samples and toxicity data

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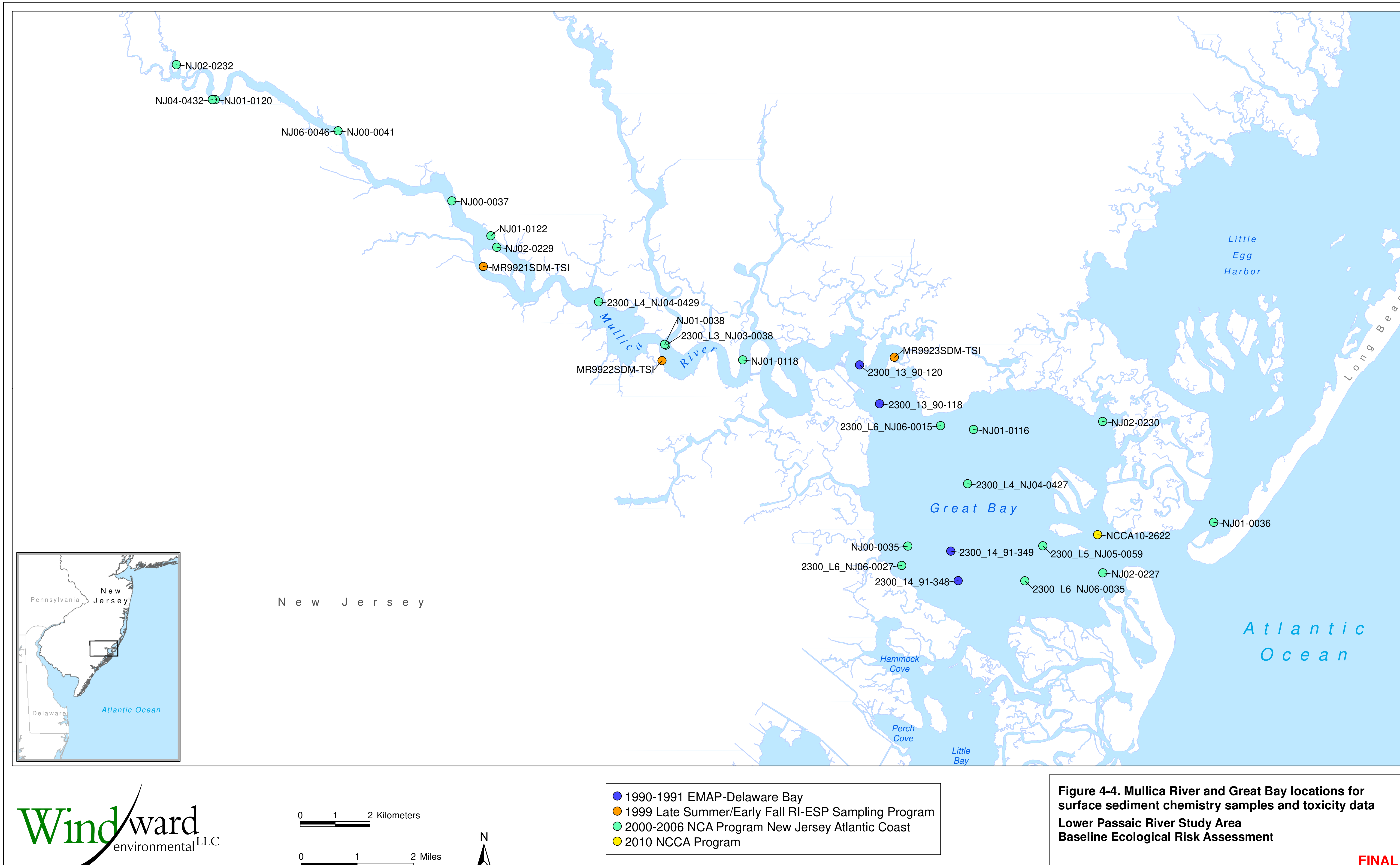

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Figure 4-4. Mullica River and Great Bay locations for surface sediment chemistry samples and toxicity data

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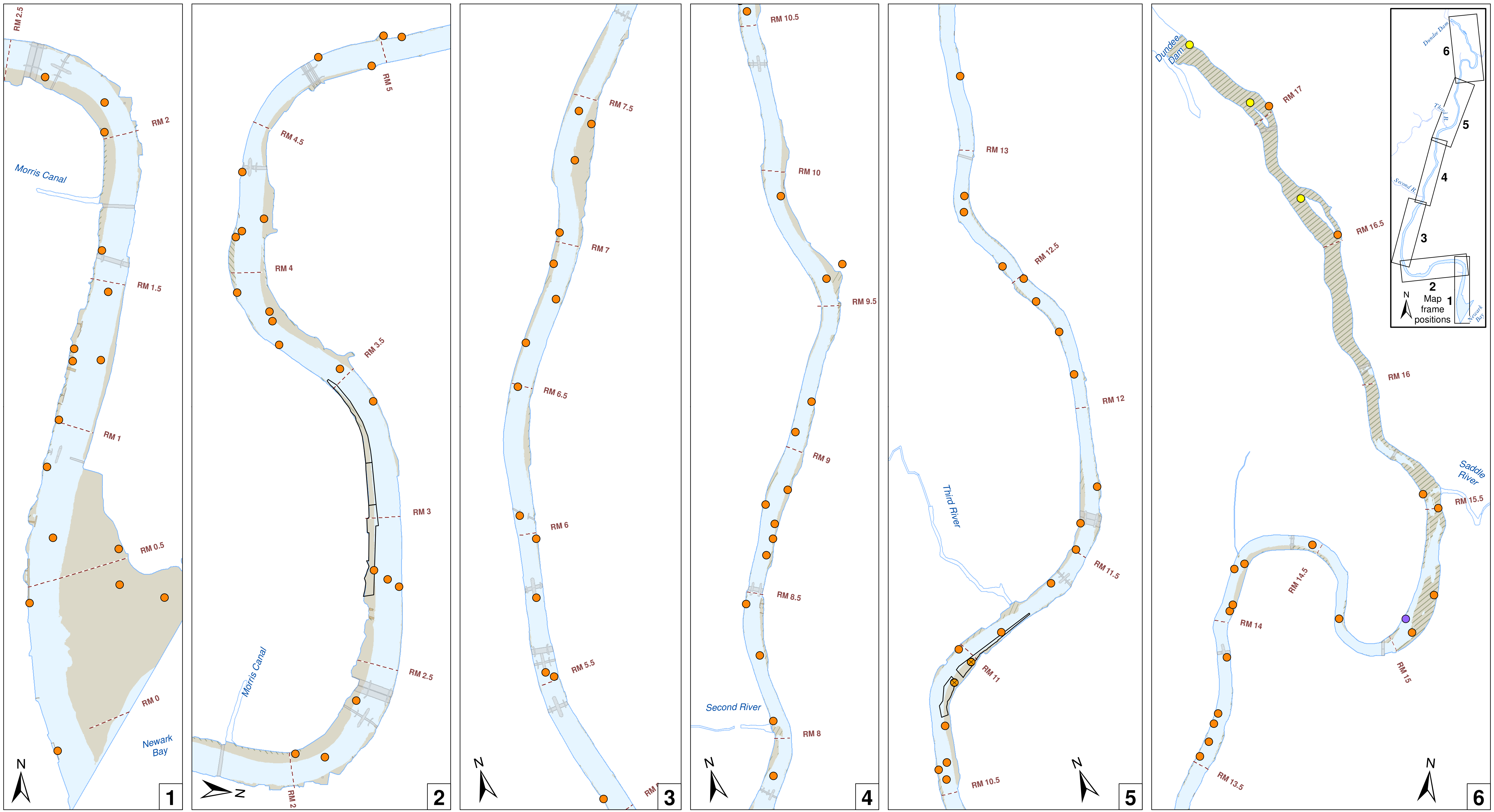

0 0.25 0.5 0.75 Kilometers

0 0.25 0.5 Miles

Benthic sediment sample
- Chemistry/toxicity only
- Benthic community only
- Complete SQT
- Remediated location[a]

- Dredge zone
- Bridge
- Abutment
- Dock
- River mile
- Mudflat
- Gravel with fines
- LPRSA

[a]One sample was collected in the Lister Ave. dredge area at RM 2.8 and two were collected in the RM 10.9 dredge area.

Note: SQT samples were analyzed for sediment chemistry, sediment toxicity, and benthic invertebrate community structure.

Mudflat areas for ecological receptors are those areas where the river bottom slope is ≤ 6° and the depth is ≥ -2 ft MLLW, based on the 2007 CPG bathymetric survey.

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Figure 4-5. LPRSA locations for SQT samples

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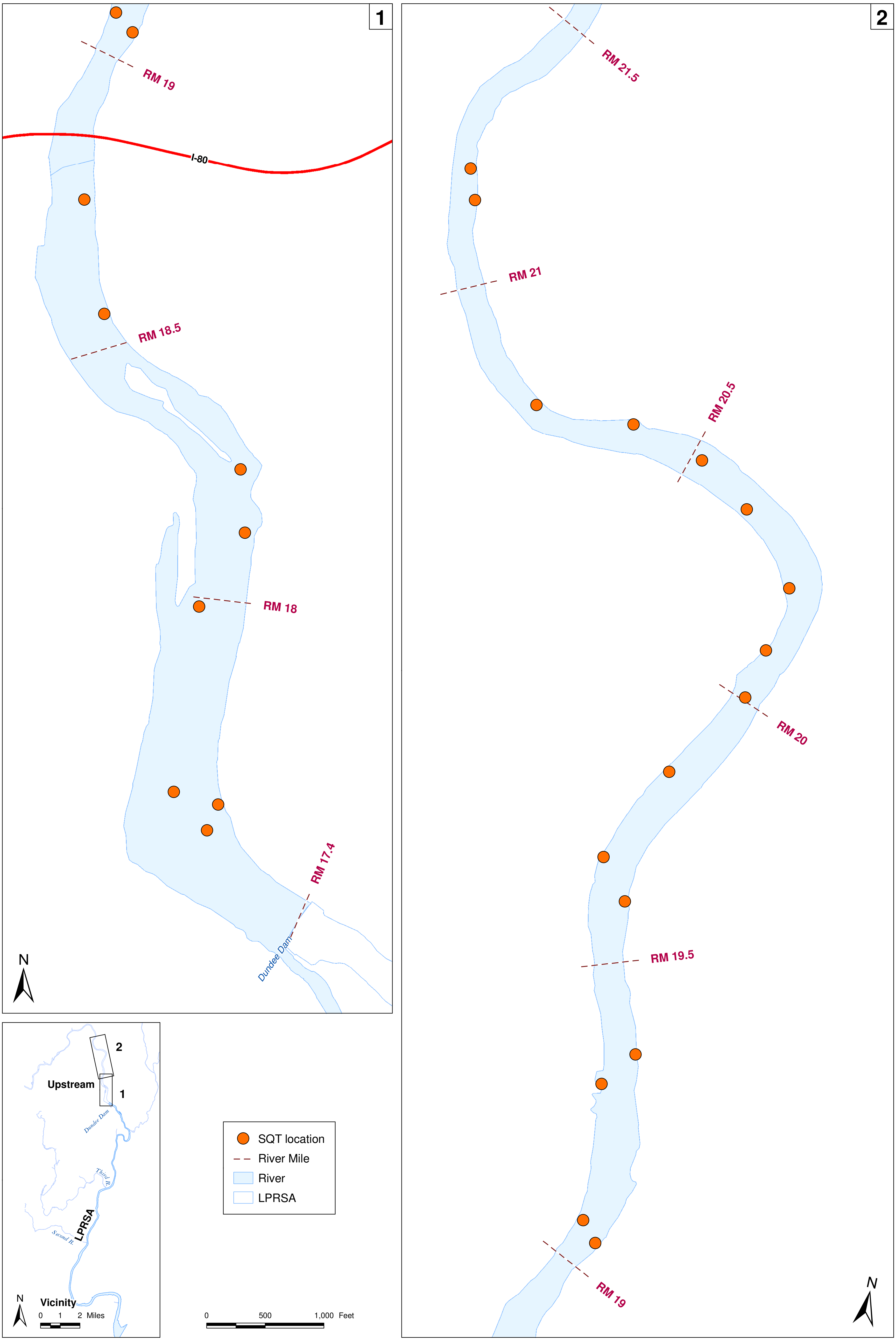

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Figure 4-6. Locations above Dundee Dam for SQT samples

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4.2.3 Tissue chemistry data

Fish and crab tissue chemistry data used in this BERA were from samples collected from the LPRSA in 2009 and 2010 for the following species: American eel, blue crab, brown bullhead, common carp, channel catfish, largemouth bass, northern pike, smallmouth bass, white catfish, white perch, white sucker [*Catastomus commersoni*], and SFF (i.e., gizzard shad, mixed forage fish, mummichog, pumpkinseed, silver shiner [*Notropis photogenis*], spottail shiner [*Notropis hudsonius*], and white perch)[57] (Table 4-3; Figures 4-7 through 4-15). The LPRSA tissue chemistry dataset also included estuarine worm (*Nereis virens)* and freshwater worm (*Lumbriculus variegatus*) data from 28-day laboratory bioaccumulation studies conducted using sediment collected from throughout the LPRSA in 2010 (Figure 4-16) and Eastern elliptio mussel and ribbed mussel data from an *in situ* caged bivalve study conducted in 2011 (Figure 4-17).[58] Only whole-body concentrations were used to develop exposure concentrations in this BERA. Whole-body concentrations were estimated from individual fillet and carcass concentrations for some fish samples and from muscle/hepatopancreas for some blue crab samples (see Section 4.3.4).

Background datasets were developed using tissue samples collected by CPG from above Dundee Dam in 2012 and tissue samples collected by non-CPG parties from Jamaica Bay and Lower Harbor in 1999 and from the Mullica River and Great Bay in 1999 and 2000 (Table 4-3).[59] Species collected from above Dundee Dam included American eel, brown bullhead, common carp, channel catfish, northern pike, smallmouth bass, white perch, white sucker, and SFF (i.e., pumpkinseed, silver shiner, and banded killifish) at sampling locations shown in Figures 4-18 through 4-24. Species collected from Jamaica Bay and Lower Harbor included banded killifish, mummichog, and other killifish species at sampling locations shown in Figure 4-25. Mummichog was the only type of fish collected from the Mullica River and Great Bay; sampling locations are shown in Figure 4-26.

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[57] Mixed-species composites were composed of multiple SFF species, including Atlantic silverside, bluegill, gizzard shad, inland silverside, spottail shiner, smallmouth bass, striped bass, striped mullet (*Mugil cephalus*), tessellated darted, and white perch (Windward 2018c).

[58] Mussel tissue data were normalized to Day 0 of the caged bivalve study.

[59] Background datasets included only species and tissue types that were used in this BERA dataset.

Table 4-3. Tissue data included in the BERA dataset

Sampling Event	Sampling Period	Species	Tissue Type	Number of Samples	Chemical Group	Source
LPRSA						
2009 fish and decapod tissue collection	August to September 2009	American eel	whole-body composites and individuals	19	metals, butyltins, PAHs, alkylated PAHs, SVOCs, PCB Aroclors, PCB congeners, PCDDs/PCDFs, organochlorine pesticides, lipids, and percent moisture	Windward (2018b)
			whole-body (calculated) composite and individual[a]	2		
		blue crab	whole-body (calculated) composites[b]	24		
			muscle-only composites	21		
			hepatopancreas-only composites	7		
		brown bullhead	whole-body individuals	6		
		common carp	whole-body individuals	12		
		channel catfish	whole-body individuals (calculated)	11		
		largemouth bass	whole-body (calculated) composites and individuals[a]	3		
		northern pike	whole-body (calculated) individual[a]	1		
		smallmouth bass	whole-body (calculated) composites[a]	3		
		white catfish	whole-body (calculated) individuals[a]	19		
		white perch	whole-body composites and individuals	19		
			whole-body (calculated) individual[a]	1		
		white sucker	whole-body (calculated) individuals[a]	5		

Table 4-3. Tissue data included in the BERA dataset

Sampling Event	Sampling Period	Species	Tissue Type	Number of Samples	Chemical Group	Source
2009 laboratory bioaccumulation evaluation	December 2009 to January 2010	estuarine worm (*Nereis virens)*	whole-body composites	5	metals, butyltins, PAHs, SVOCs, PCB congeners, PCDDs/PCDFs, organochlorine pesticides, lipids, and percent moisture[d]	Windward (2018a)
		freshwater worm (*Lumbriculus variegatus*)		14[c]		
2010 spring SFF reconnaissance sampling	May 2010	mummichog	egg composites	10	lipids	Windward (2018b)
2010 SFF tissue collection	June to August 2010	mummichog	whole-body composites	18	metals, butyltins, PAHs, alkylated PAHs, organochlorine pesticides, PCB Aroclors, PCB congeners, PCDDs/PCDFs, SVOCs, lipids, and percent moisture	Windward (2018c)
		gizzard shad		3		
		pumpkinseed		1		
		silver shiner		1		
		spottail shiner		1		
		mixed forage fish[e]		4		
		white perch		2		
2011 caged bivalve study	March to June 2011	Eastern elliptio mussel (freshwater)	soft-tissue composites[f]	5[f]	metals, butyltins, PAHs, alkylated PAHs, SVOCs, PCB Aroclors, PCB congeners, PCDDs/PCDFs, organochlorine pesticides, lipids, and percent moisture	Windward (2019a)
		ribbed mussel (estuarine)		3[f]		

Table 4-3. Tissue data included in the BERA dataset

Sampling Event	Sampling Period	Species	Tissue Type	Number of Samples	Chemical Group	Source
Passaic River above Dundee Dam						
2012 upstream tissue sampling	October 2012	American eel	whole-body individuals	6	metals, butyltins, SVOCs, PAHs, alkylated PAHs, PCB Aroclors, PCB congeners, PCDDs/PCDFs, organochlorine pesticides, lipids, and percent moisture	Windward (2019c)
			whole-body (calculated) composites and individuals[a]	10		
		banded killifish	whole-body composite	1		
		brown bullhead	whole-body individuals	6		
		common carp	whole-body individuals	5		
			whole-body (calculated) individuals[a]	5		
		channel catfish	whole-body (calculated) individuals[a]	4		
		Northern pike	whole-body (calculated)[a] individual	1		
		pumpkinseed	whole-body composite	1		
		silver shiner	whole-body composite	1		
		smallmouth bass	whole-body (calculated) composites[a]	3		
		white perch	whole-body (calculated) composites	8		
		white sucker	whole-body (calculated) individuals[a]	5		
Jamaica Bay/Lower Harbor						
Fall1999 harbor fish collection	November 1999	mummichog	whole-body composites	2	metals, PAHs, PCB Aroclors, PCB congeners, PCDDs/PCDFs, and organochlorine pesticides	Litten (2003)

Table 4-3. Tissue data included in the BERA dataset

Sampling Event	Sampling Period	Species	Tissue Type	Number of Samples	Chemical Group	Source
Summer 1999 harbor fish collection	September 1999	mummichog	whole-body composites	5	metals, PAHs, PCB Aroclors, PCB congeners, PCDDs/PCDFs, and organochlorine pesticides	Litten (2003)
Mullica River/Great Bay						
1999 late summer/early fall RI-ESP sampling program	October 1999	mummichog	whole-body composites	9	metals, PAHs, PCB Aroclors, PCB congeners, PCDDs/PCDFs, organochlorine pesticides, and lipids	NOAA (2013)
2000 spring RI-ESP sampling program	May 2000			3		

[a] Fish whole-body tissue concentrations were calculated using fillet and carcass tissue concentrations as described in Section 4.2.3.

[b] Crab whole-body tissue concentrations were calculated using muscle/hepatopancreas and carcass concentrations as described in Section 4.2.4. Seventeen muscle/hepatopancreas crab samples collected above RM 10 did not have corresponding carcass samples to calculate whole-body concentrations. These samples were evaluated in the uncertainty section.

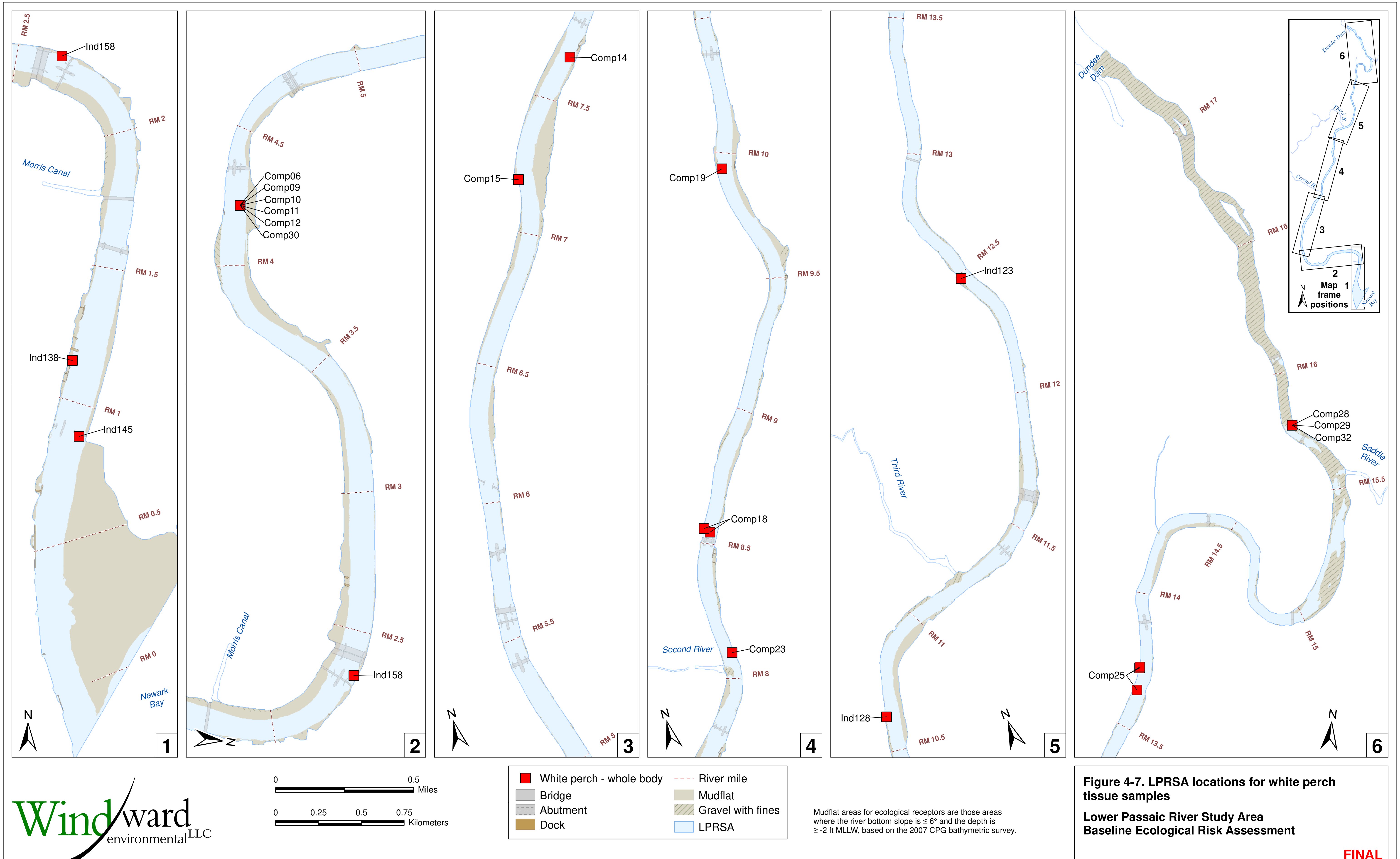
[c] Among these 14 samples, the sediment used in the laboratory bioaccumulation study from 1 sample (LPRT11E) was collected in the RM 10.9 dredge area and was excluded when calculating freshwater worm tissue EPCs.

[d] The five *N. virens* tissue samples had sufficient mass for analysis of the full set of analytes; the reduced analyte priority list presented in the benthic tissue analysis plan (Windward 2010h) was followed for *Lumbriculus variegatus* tissue samples because of mass limitations for some samples.

[e] Mixed-species composites were composed of multiple SFF species, including Atlantic silverside, bluegill, gizzard shad, inland silverside, spottail shiner, smallmouth bass, striped bass, striped mullet, tessellated darted, and white perch (Windward 2018c).

[f] Mussel tissue data were normalized to Day 0 of the caged bivalve study.

BERA – baseline ecological risk assessment
EPC – exposure point concentration
ESP – ecological sampling program
LPRSA – Lower Passaic River Study Area
NOAA – National Oceanic and Atmospheric Administration
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
PCDD – polychlorinated dibenzo-*p*-dioxin
PCDF – polychlorinated dibenzofuran
RI – remedial investigation
RM – river mile
SFF – small forage fish
SVOC – semivolatile organic compound
Windward – Windward Environmental LLC



Mudflat areas for ecological receptors are those areas where the river bottom slope is ≤ 6° and the depth is ≥ -2 ft MLLW, based on the 2007 CPG bathymetric survey.

Figure 4-7. LPRSA locations for white perch tissue samples

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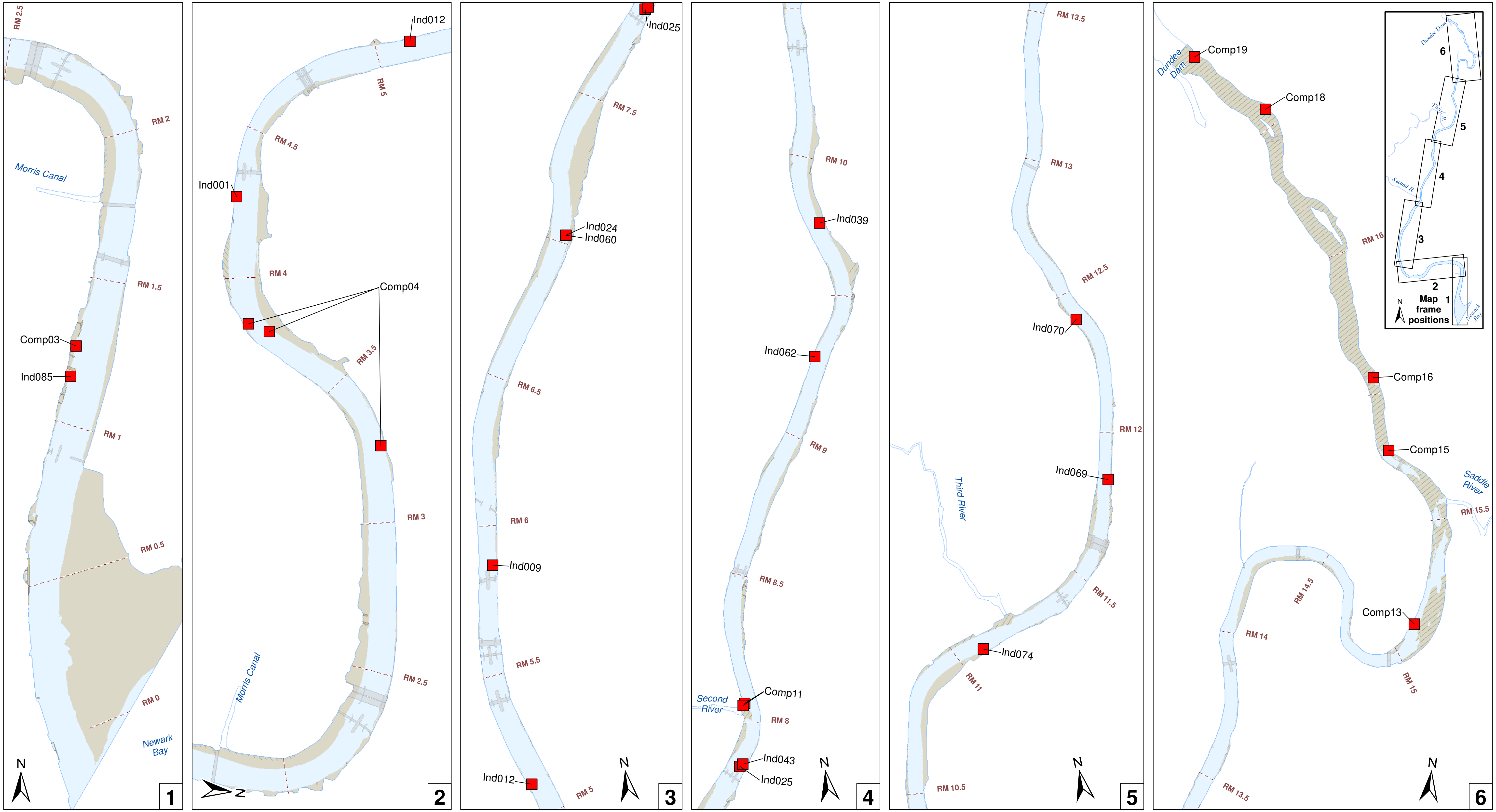

0 0.25 Miles

0 0.5 Kilometers

American eel - whole body	River mile
Bridge	Mudflat
Abutment	Gravel with fines
Dock	LPRSA

Mudflat areas for ecological receptors are those areas where the river bottom slope is ≤ 6° and the depth is ≥ -2 ft MLLW, based on the 2007 CPG bathymetric survey.

Figure 4-8. LPRSA locations for American eel tissue samples

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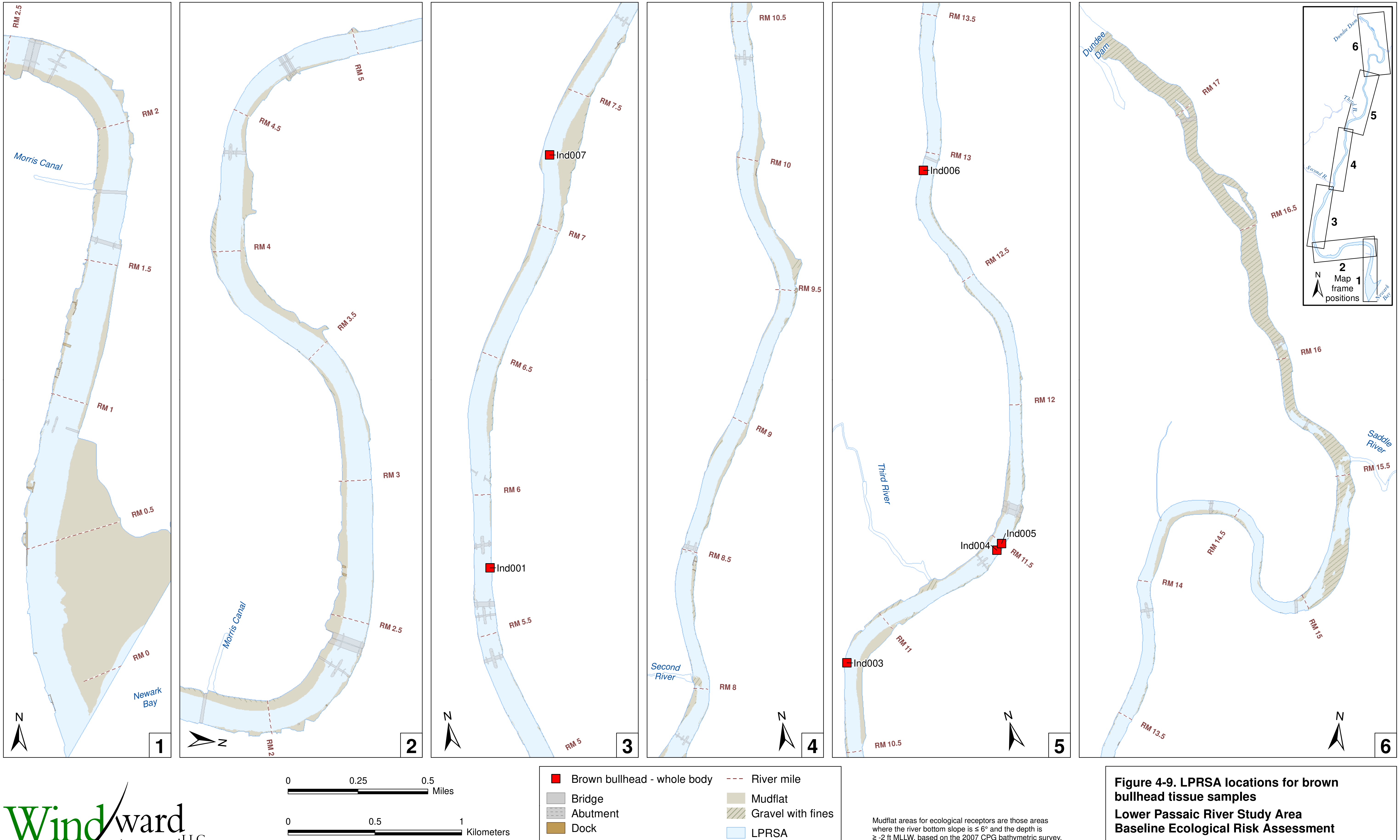

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Mudflat areas for ecological receptors are those areas where the river bottom slope is ≤ 6° and the depth is ≥ -2 ft MLLW, based on the 2007 CPG bathymetric survey.

Figure 4-9. LPRSA locations for brown bullhead tissue samples

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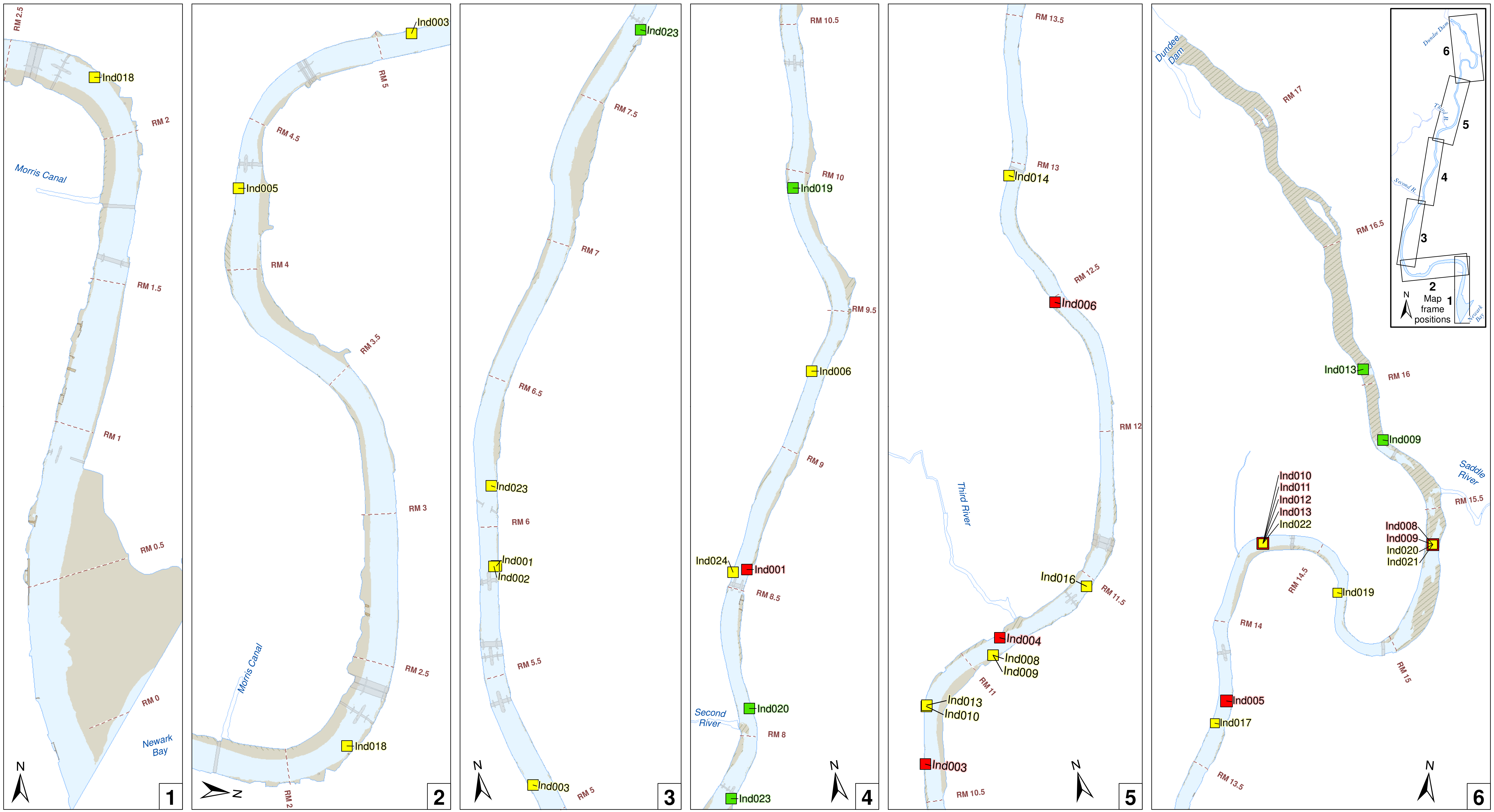
Prepared by mikey 5/30/2019; W:\Projects\06-58-01 Passaic RI\Data\GIS\Maps_and_Analysis\BERA\Revised BERA 2016\4069_Bullhead composite samples 2009_CEH_20131204.mxd



Channel catfish - whole body
White catfish - whole body
White sucker - whole body

Bridge
Abutment
Dock

River mile
Mudflat
Gravel with fines
LPRSA

0 | 0.25 | 0.5 Miles

0 | 0.5 | 1 Kilometers

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Mudflat areas for ecological receptors are those areas where the river bottom slope is ≤ 6° and the depth is ≥ -2 ft MLLW, based on the 2007 CPG bathymetric survey.

Figure 4-10. LPRSA locations for channel catfish, white catfish, and white sucker samples

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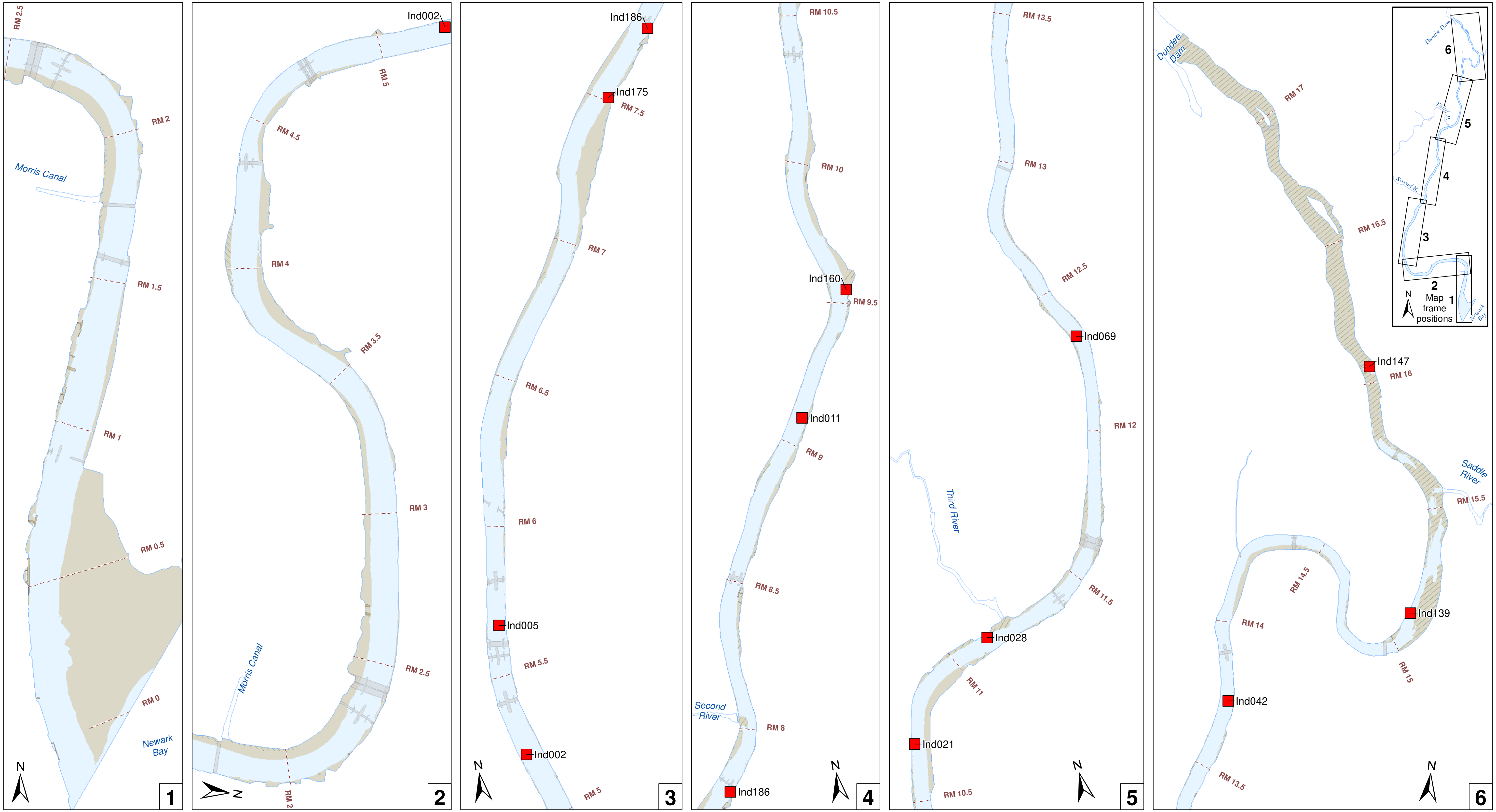

0 0.25 0.5 Miles

0 0.5 1 Kilometers

Carp - whole body
River mile
Bridge
Mudflat
Abutment
Gravel with fines
Dock
LPRSA

Mudflat areas for ecological receptors are those areas where the river bottom slope is ≤ 6° and the depth is ≥ -2 ft MLLW, based on the 2007 CPG bathymetric survey.

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Figure 4-11. LPRSA locations for common carp samples

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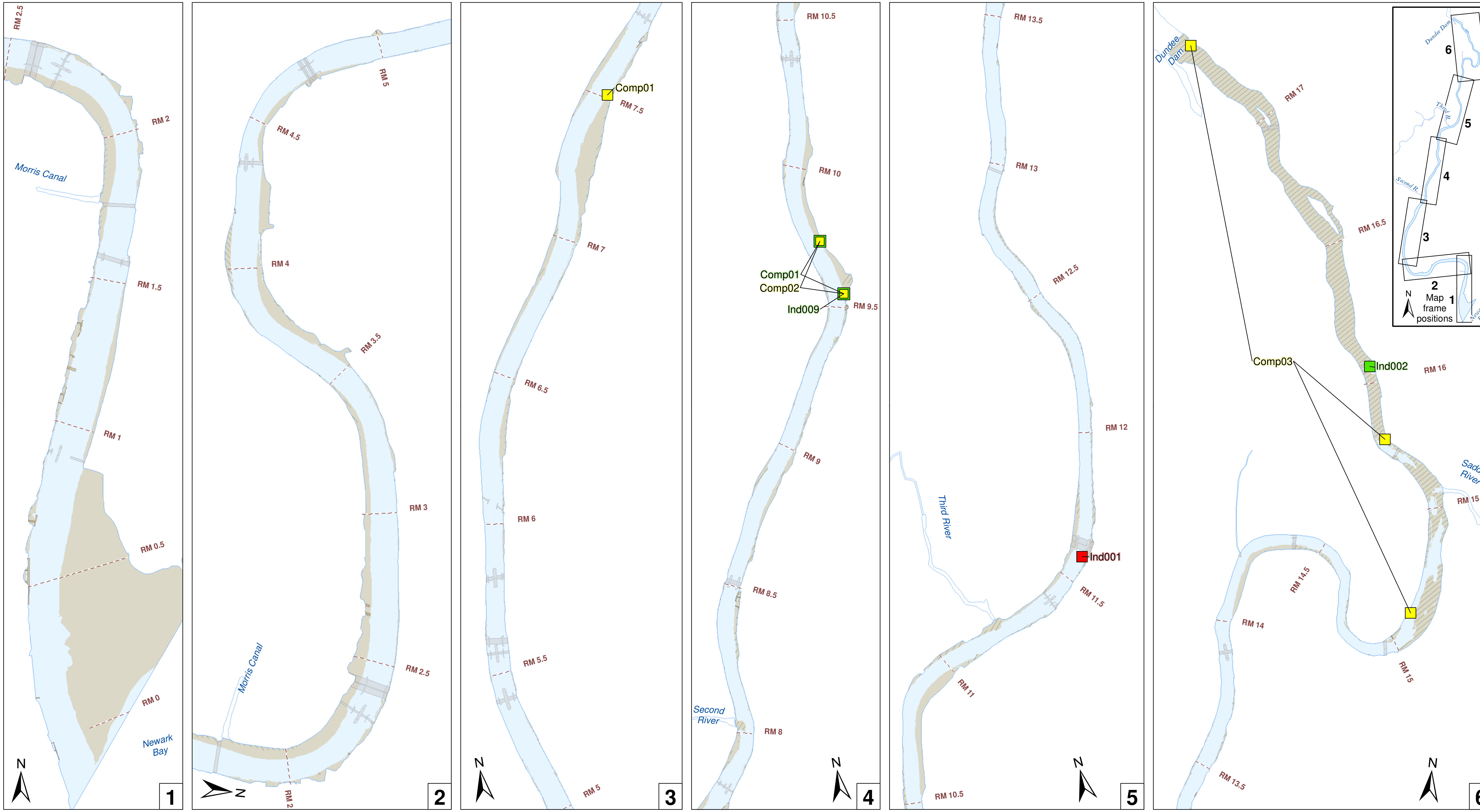
Prepared by mikey 5/30/2019; W:\Projects\06-58-01 Passaic RI\Data\GIS\Maps_and_Analysis\BERA\Revised BERA 2016\3978_Carp composite samples 2009_CEH_20131111.mxd



0 0.25 0.5 Miles

0 0.25 0.5 0.75 Kilometers

Northern pike - whole body
Smallmouth bass - whole body
Largemouth bass - whole body
Bridge
Abutment
Dock
River mile
Mudflat
Gravel with fines
LPRSA

Mudflat areas for ecological receptors are those areas where the river bottom slope is ≤ 6° and the depth is ≥ -2 ft MLLW, based on the 2007 CPG bathymetric survey.

Figure 4-12. LPRSA locations for largemouth bass, smallmouth bass, and northern pike samples

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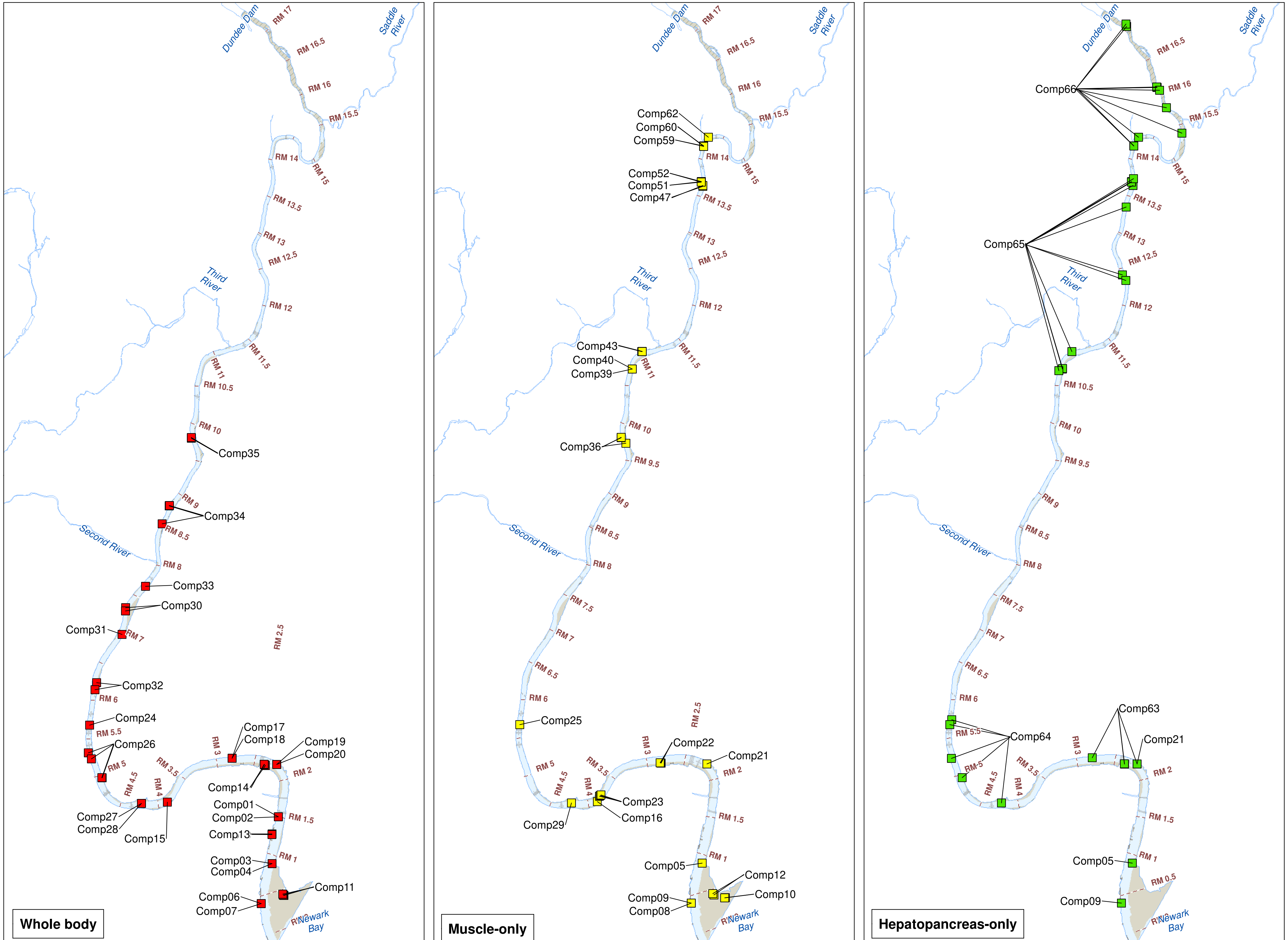

Figure 4-13. LPRSA locations for blue crab samples

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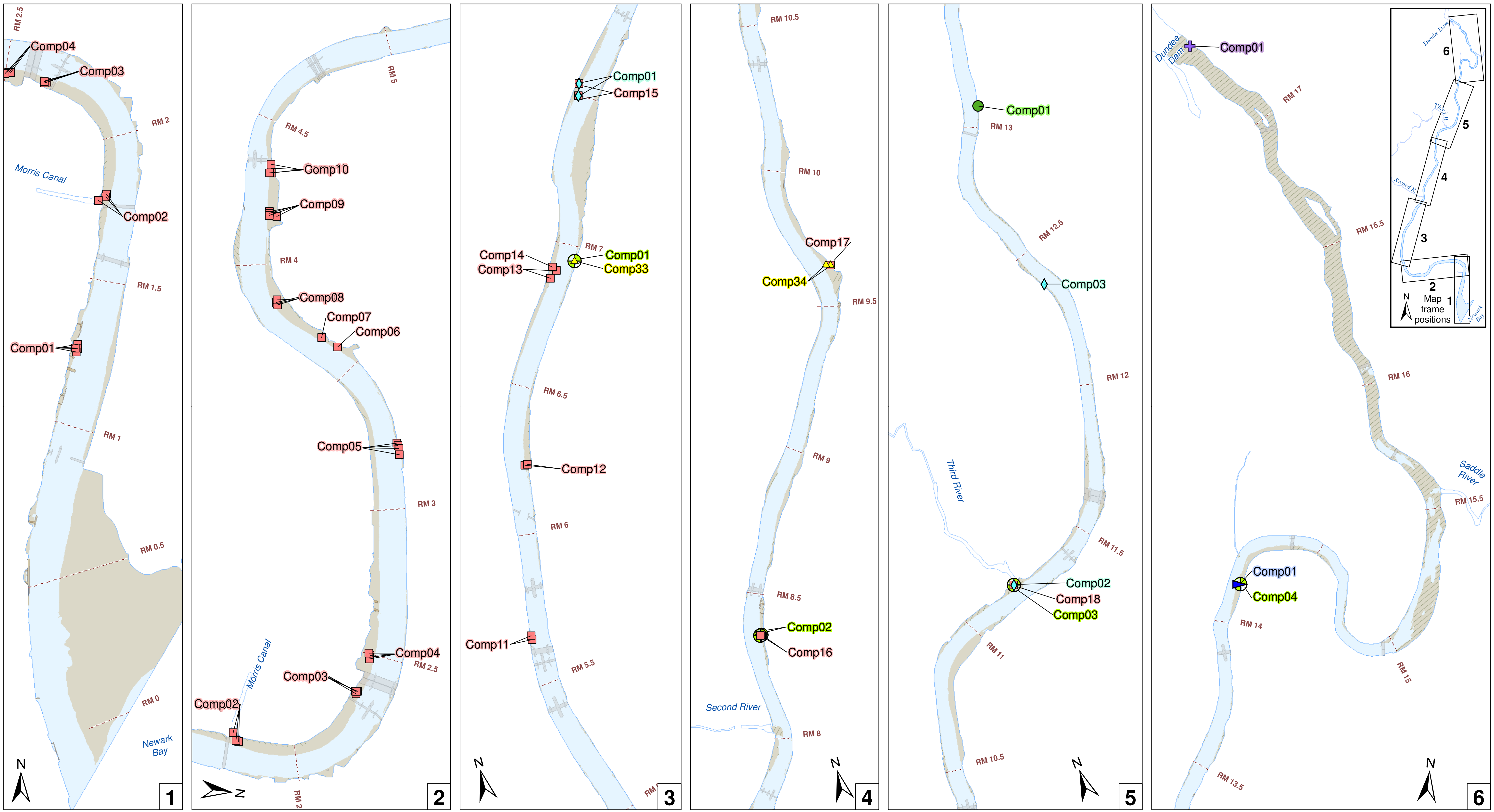

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0 0.25 0.5 Miles

0 0.25 0.5 0.75 Kilometers

Whole body sample
- Pumpkinseed
- Spottail shiner
- Silver shiner
- White perch
- Gizzard shad
- Mummichog
- Mixed forage fish

- River mile
- Bridge
- Abutment
- Dock
- Mudflat
- Gravel with fines
- LPRSA

Mudflat areas for ecological receptors are those areas where the river bottom slope is ≤ 6° and the depth is ≥ -2 ft MLLW, based on the 2007 CPG bathymetric survey.

Figure 4-14. LPRSA locations for mummichog and other small forage fish samples

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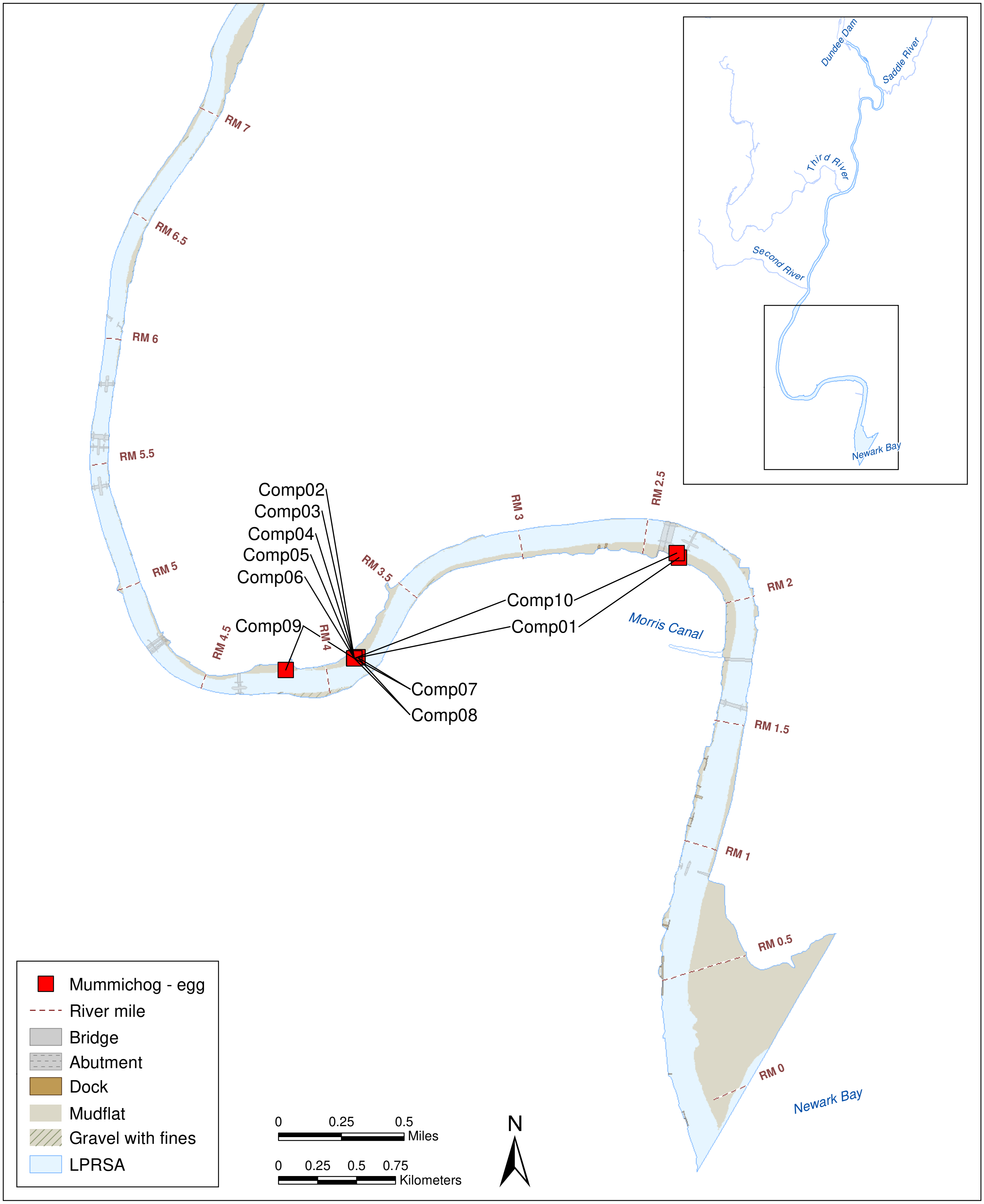

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Mudflat areas for ecological receptors are those areas where the river bottom slope is ≤ 6° and the depth is ≥ -2 ft MLLW, based on the 2007 CPG bathymetric survey.

Figure 4-15. LPRSA locations for mummichog egg composite samples
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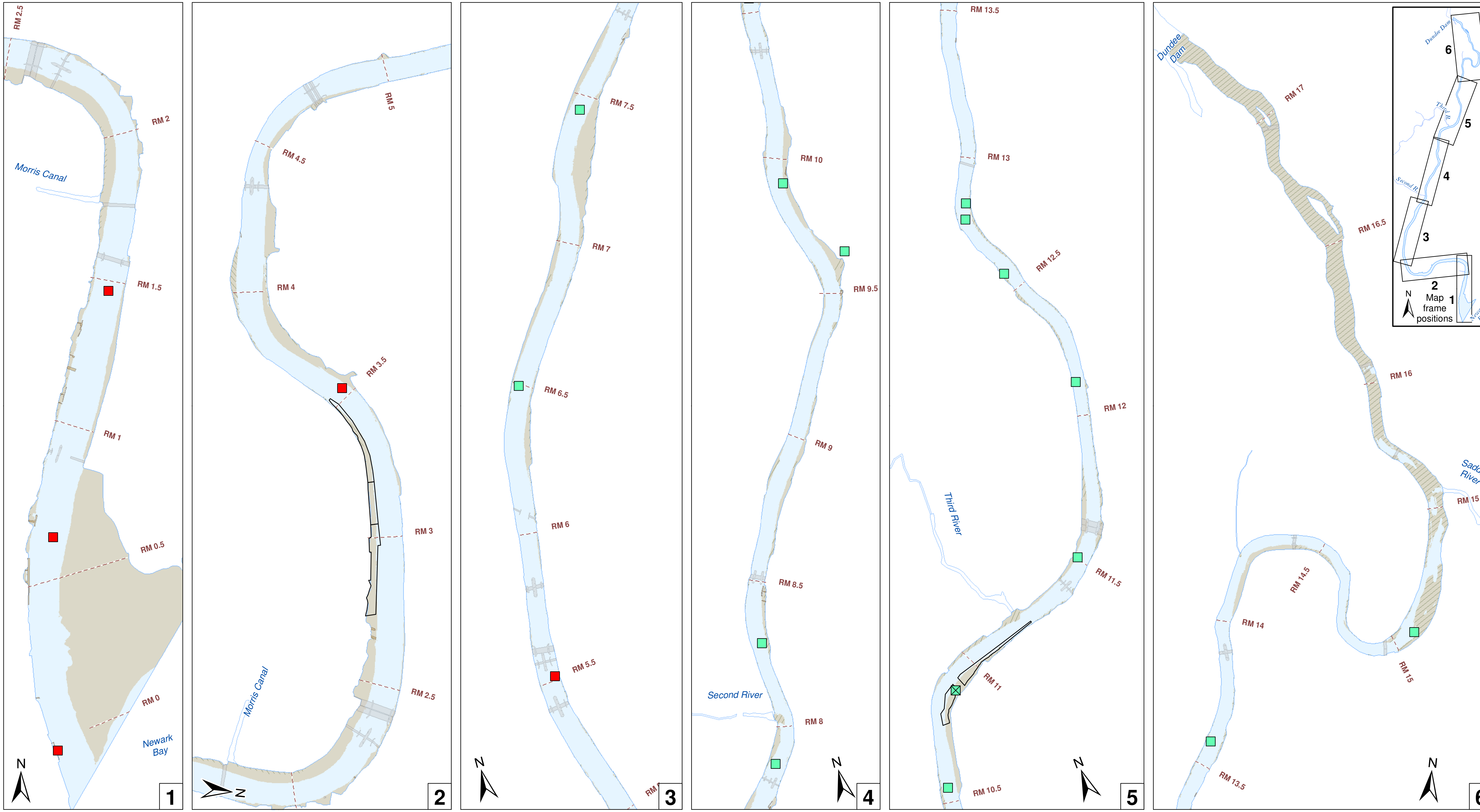

Bioaccumulation	Dredge zone	River mile
Estuarine	Bridge	Mudflat
Freshwater	Abutment	Gravel with fines
Remediated location[a]	Dock	LPRSA

[a]One sample was collected in the RM 10.9 dredge area.

Mudflat areas for ecological receptors are those areas where the river bottom slope is ≤ 6° and the depth is ≥ -2 ft MLLW, based on the 2007 CPG bathymetric survey.

Figure 4-16. LPRSA sediment locations for bioaccumulation samples

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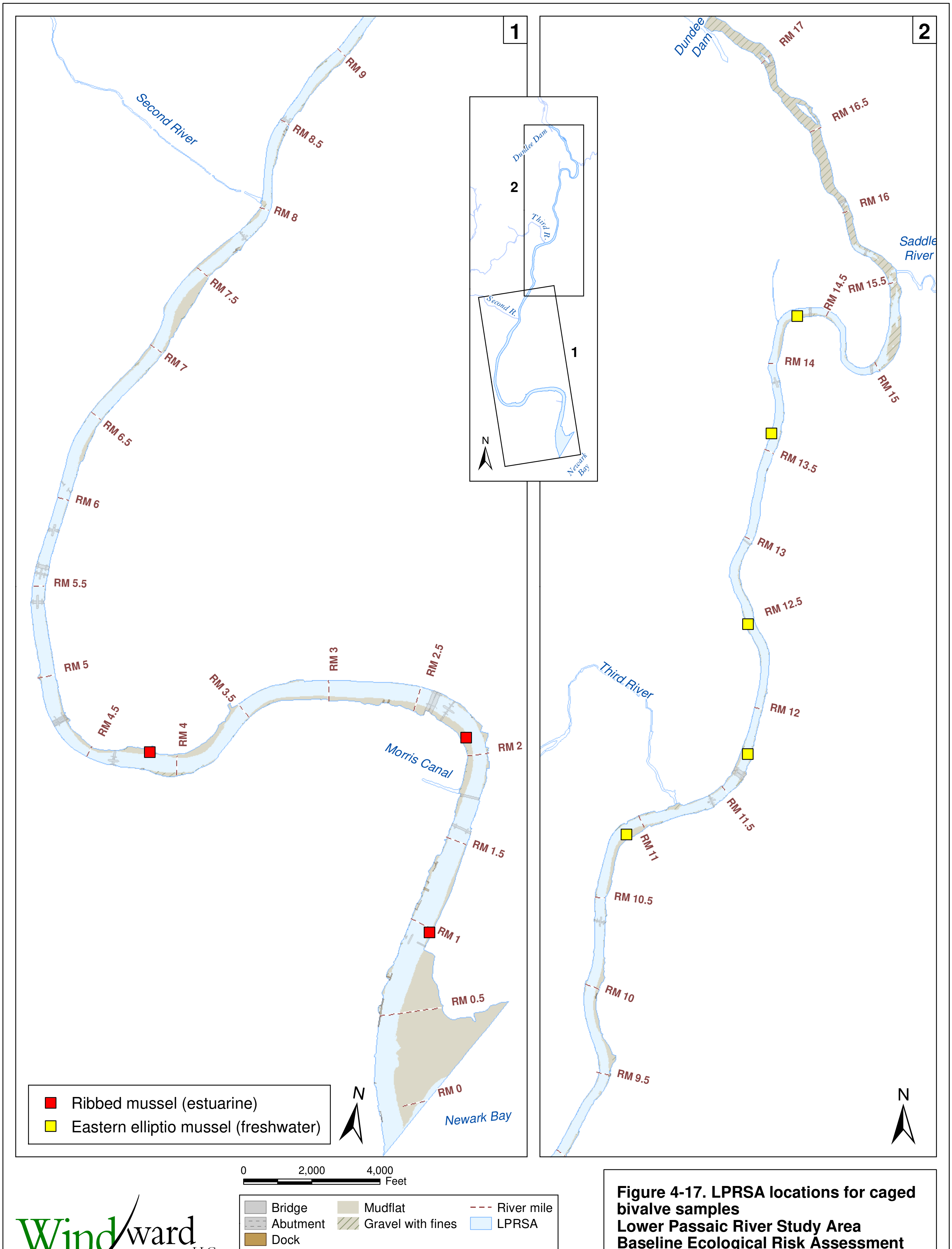

Mudflat areas for ecological receptors are those areas where the river bottom slope is ≤ 6° and the depth is ≥ -2 ft MLLW, based on the 2007 CPG bathymetric survey.

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Figure 4-17. LPRSA locations for caged bivalve samples
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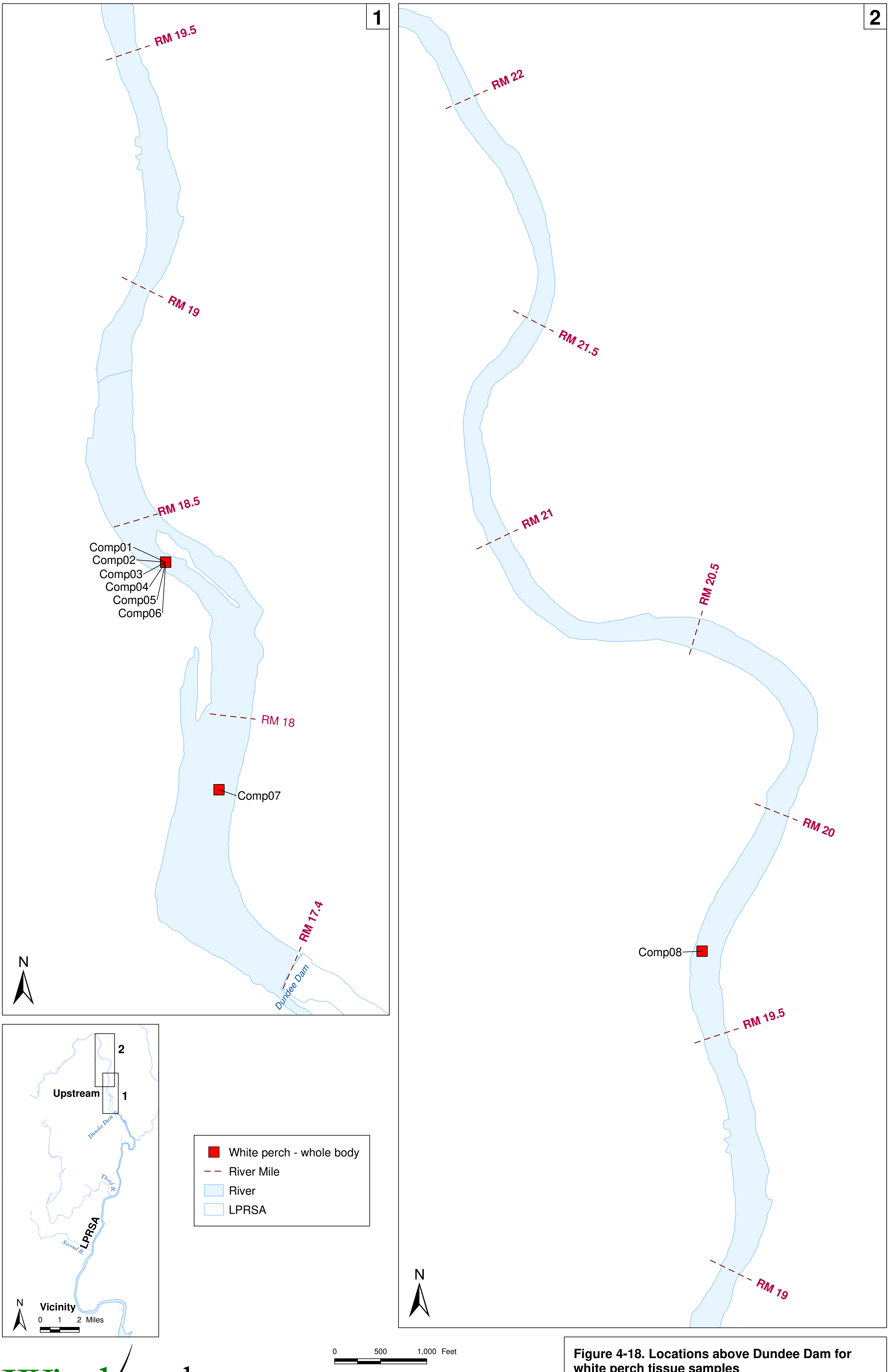

Figure 4-18. Locations above Dundee Dam for white perch tissue samples
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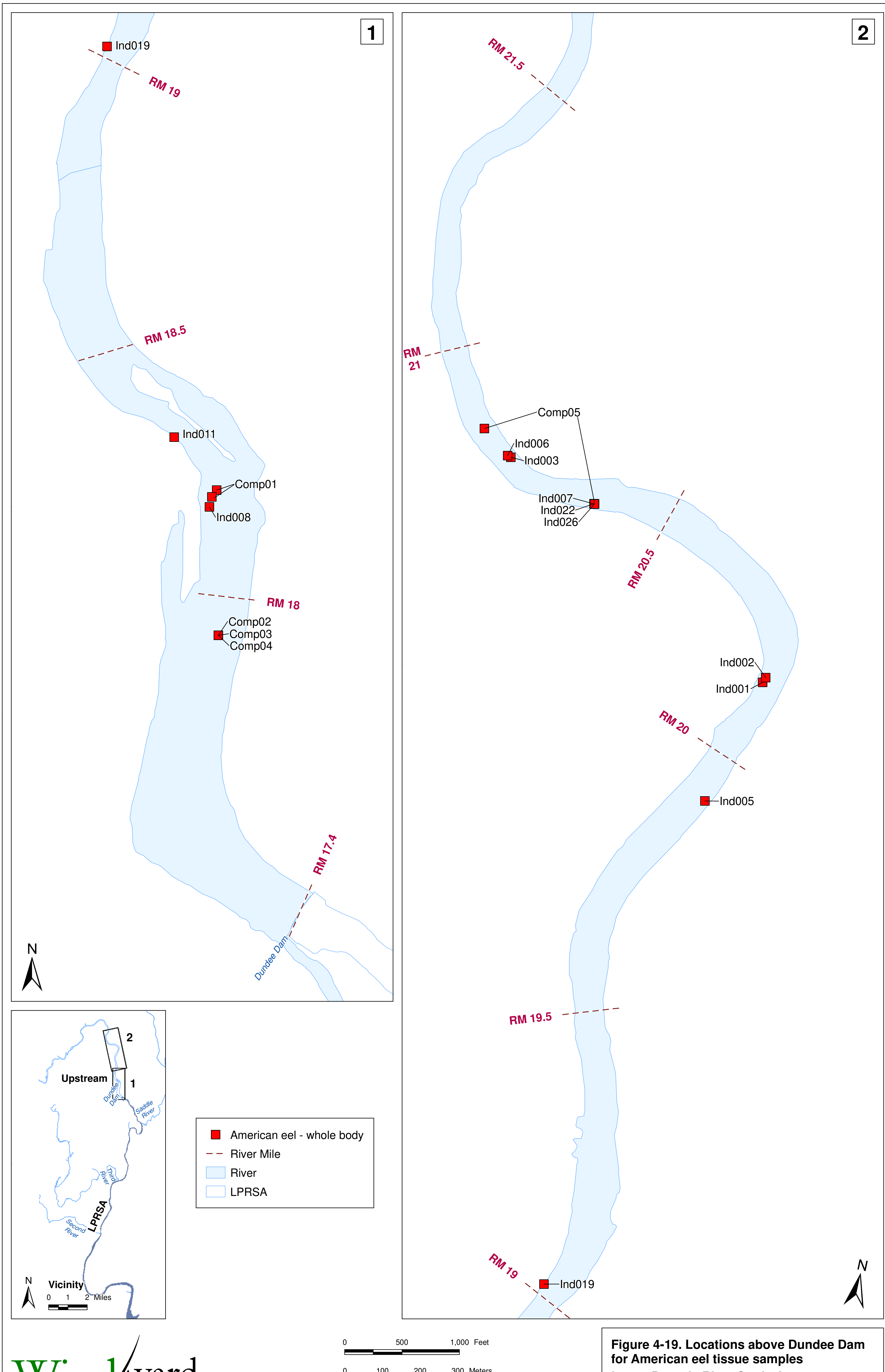

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Figure 4-19. Locations above Dundee Dam for American eel tissue samples

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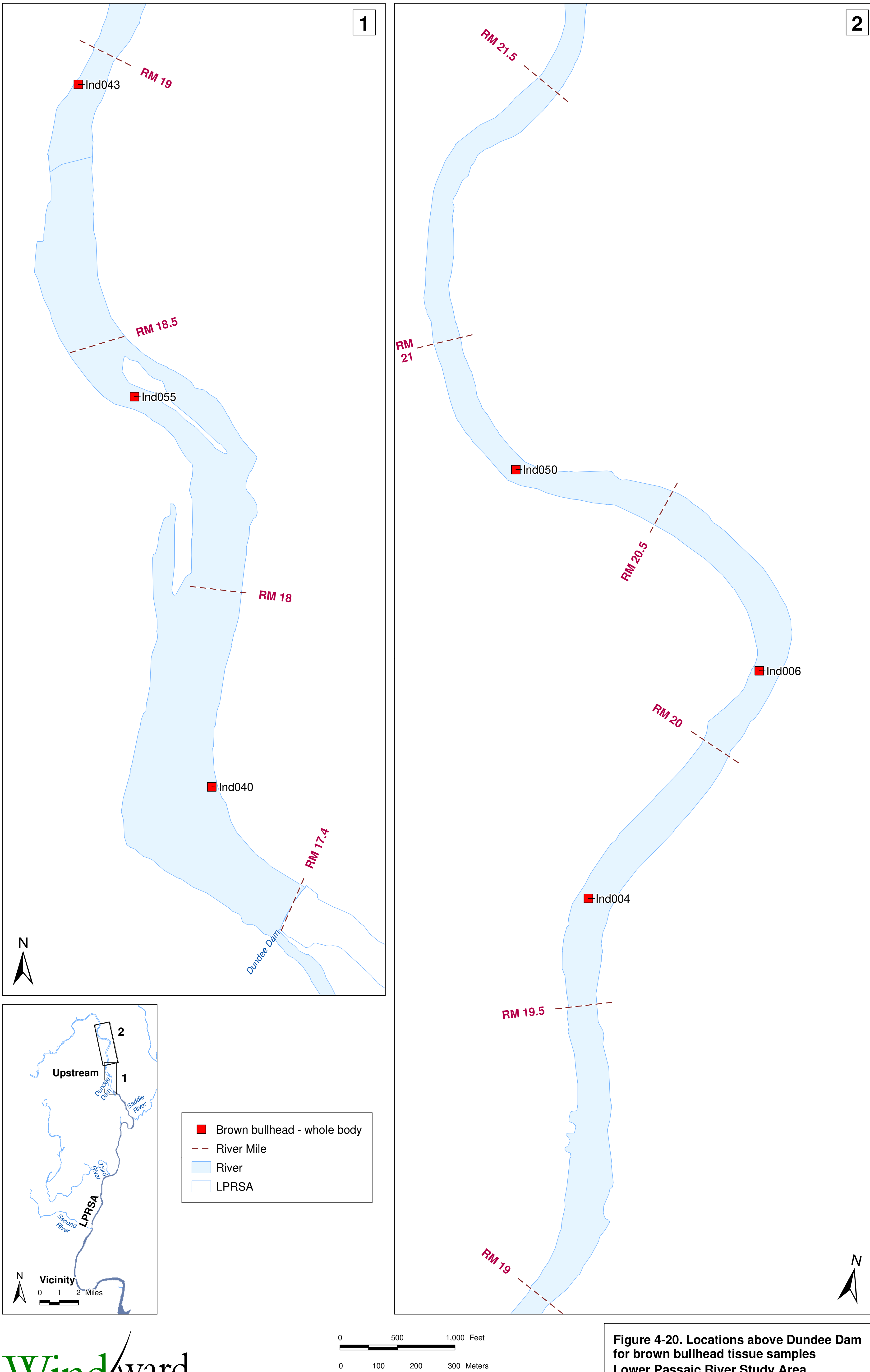

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Figure 4-20. Locations above Dundee Dam for brown bullhead tissue samples
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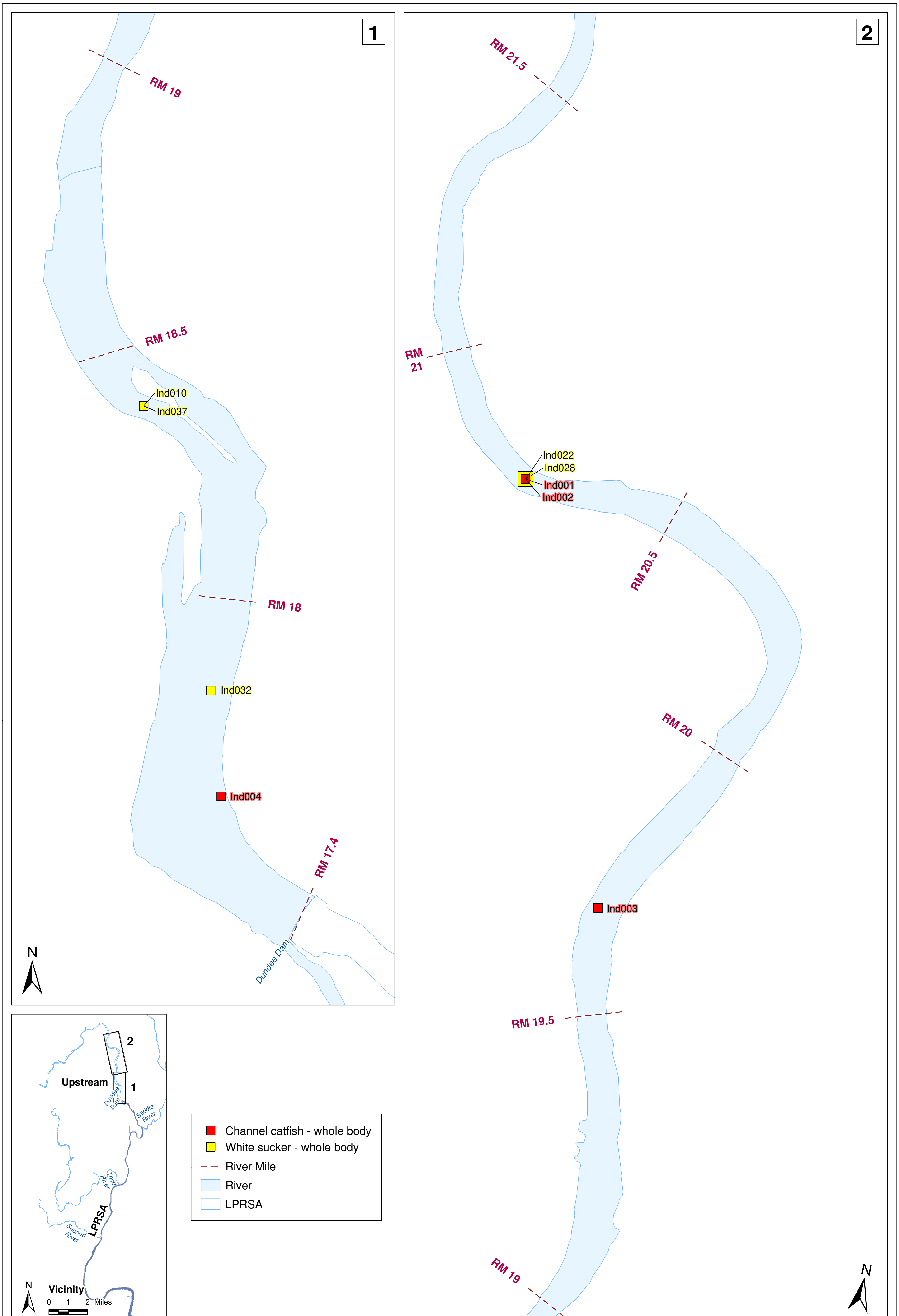

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Figure 4-21. Locations above Dundee Dam for channel catfish and white sucker samples

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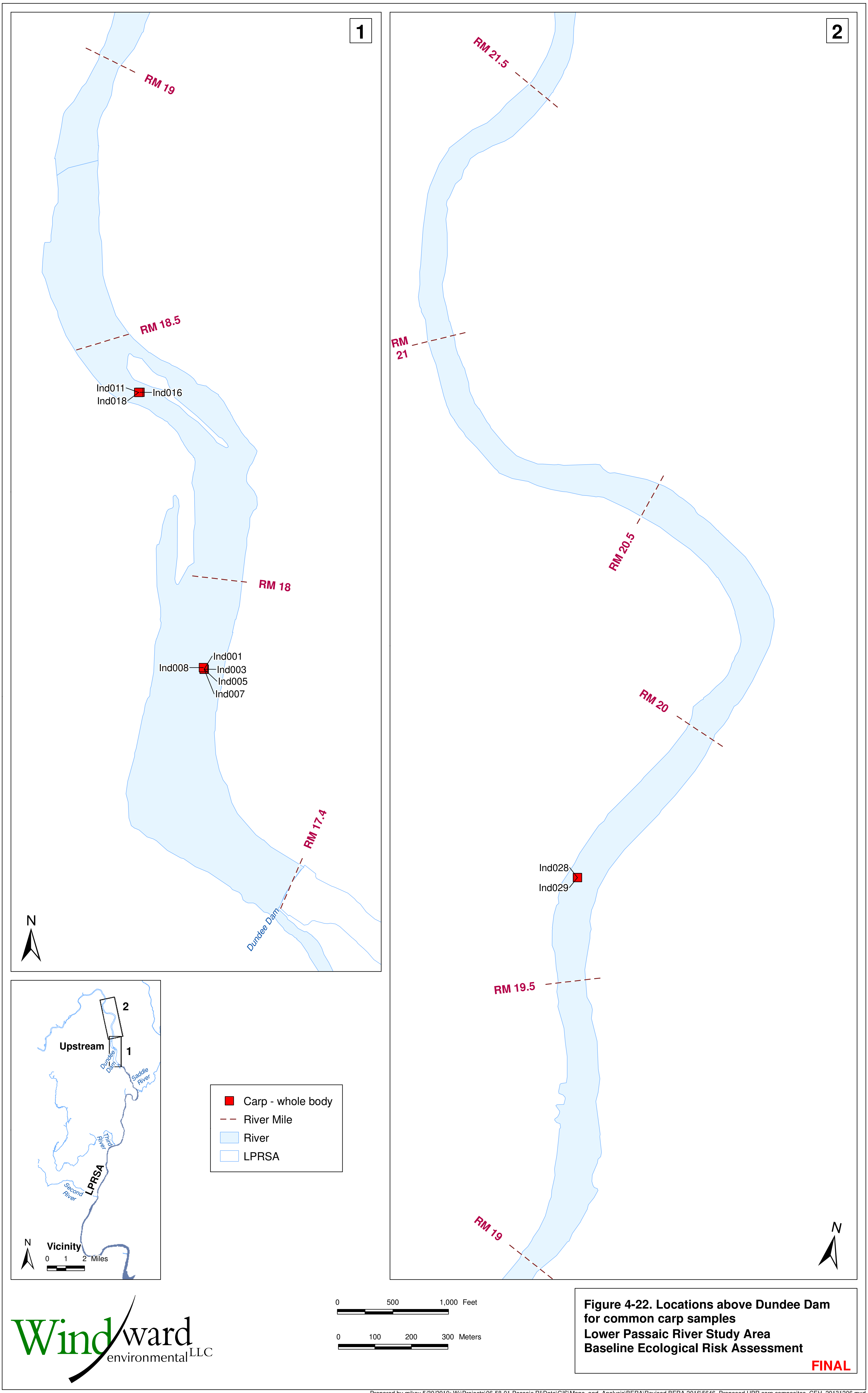

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Figure 4-22. Locations above Dundee Dam for common carp samples
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Baseline Ecological Risk Assessment

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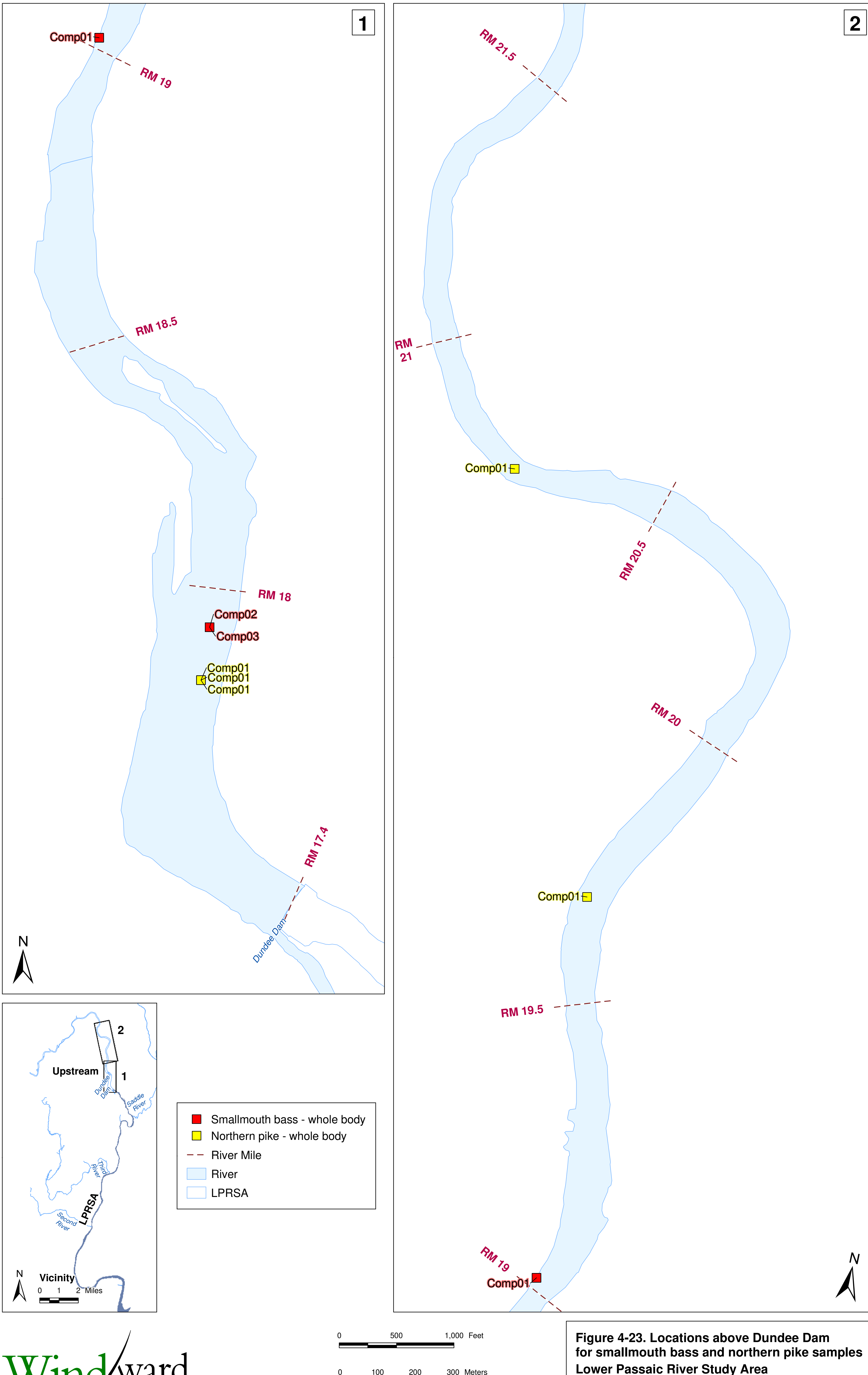

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Figure 4-23. Locations above Dundee Dam for smallmouth bass and northern pike samples
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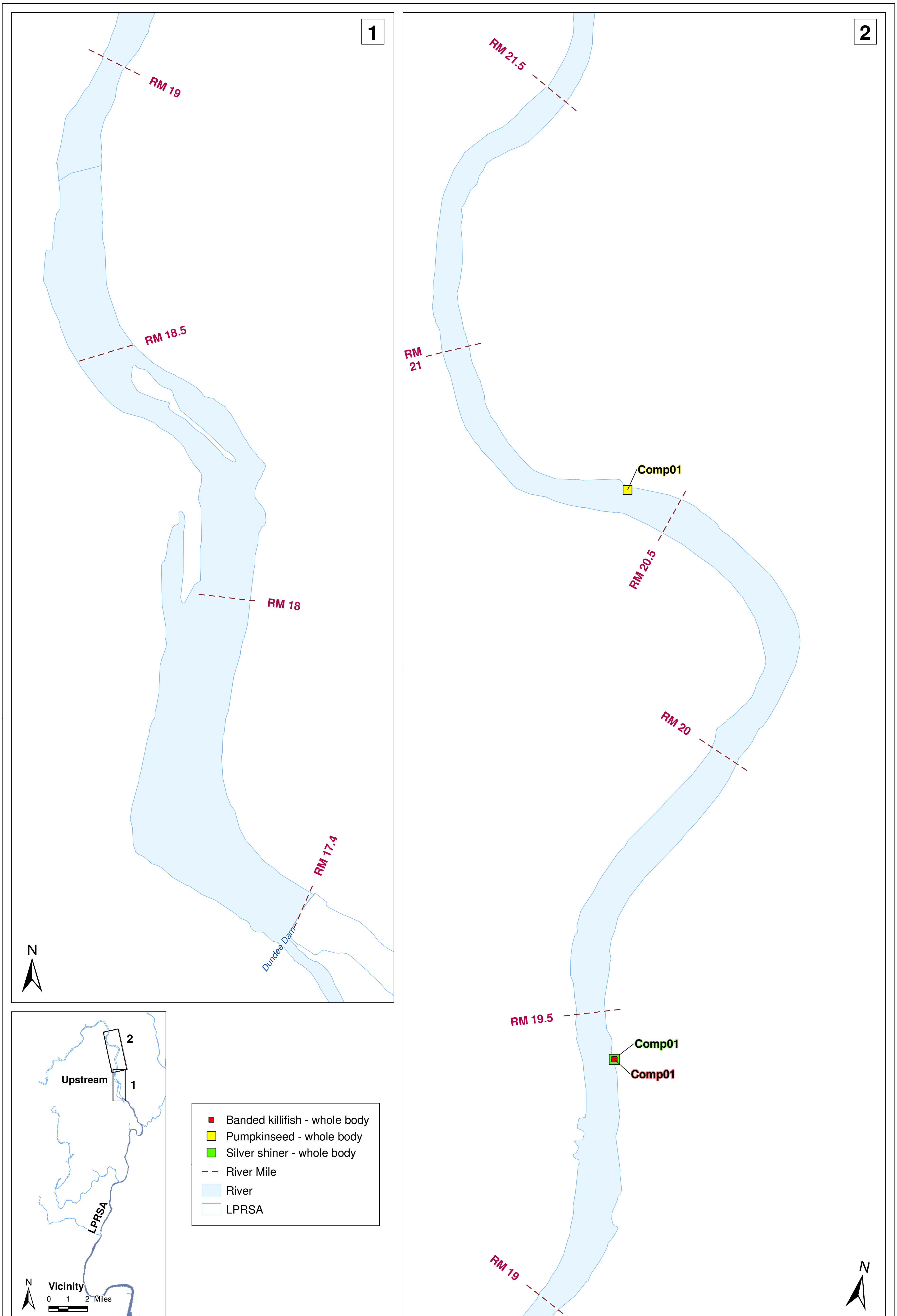

Figure 4-24. Locations above Dundee Dam for small forage fish samples
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Prepared by mikey 5/30/2019; W:\Projects\06-58-01 Passaic RI\Data\GIS\Maps_and_Analysis\BERA\Revised BERA 2016\5405_Locations above Dundee Dam for small forage fish samples_MTY_20131206.mxd

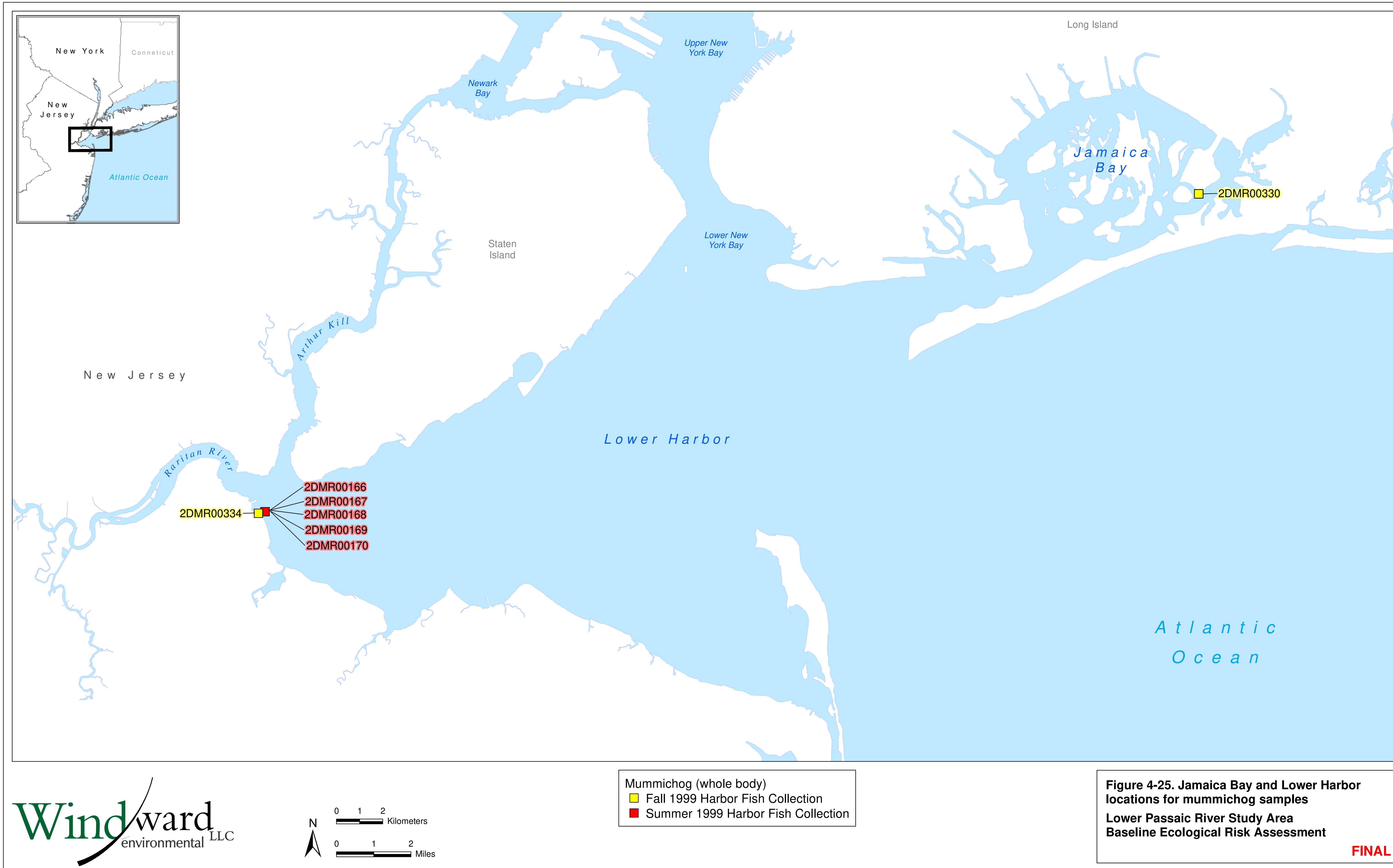

Figure 4-25. Jamaica Bay and Lower Harbor locations for mummichog samples

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Prepared by mikey 5/30/2019; W:\Projects\06-58-01 Passaic RI\Data\GIS\Maps_and_Analysis\BERA\Revised BERA 2016\5729_Jamaica Bay and Lower Harbor locs for killifish species_LSM_20151007.mxd

Windward environmental LLC

N

0 1 2 Kilometers

0 1 2 Miles

Mummichog - whole body
- 1999 Late Summer/Early Fall RI-ESP Sampling Program
- 2000 Spring RI-ESP Sampling Program

Figure 4-26. Mullica River and Great Bay locations for mummichog samples

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Prepared by mikey 5/30/2019; W:\Projects\06-58-01 Passaic RI\Data\GIS\Maps_and_Analysis\BERA\Revised BERA 2016\5730_Mullica River and Great Bay mummichog locs_MTY_20131203.mxd

4.2.4 Surface water chemistry data

Surface water chemistry data used in this BERA (Table 4-4) were from LPRSA samples collected during the following sampling events :

- Five routine chemical water column monitoring events in 2011 and 2012 (Figure 4-27) under normal flow conditions (i.e., 400 to 3,000 cfs at Dundee Dam):
  - Event 1 was conducted from August 15 to 17, 2011, during average tide (median flow at Dundee Dam was 2,650 cfs).
  - Event 2 was conducted from February 20 to 21, 2012, during spring tide (median flow at Dundee Dam was 699 cfs).
  - Event 3 was conducted from March 26 to 27, 2012, during neap tide[60] (median flow at Dundee Dam was 392 cfs).
  - Event 4 was conducted from June 4 to 5, 2012, during spring tide (median flow at Dundee Dam was 1,389 cfs).
  - Event 5 was conducted from December 10 to 11, 2012, during average tide (median flow at Dundee Dam was 664 cfs).
- A single low-flow (i.e., < 400 cfs at Dundee Dam) water column monitoring event in August 2012 during spring tide (median flow at Dundee Dam was 253 cfs) (Figure 4-27)
- Two high-flow events (i.e., > 3,000 cfs at Dundee Dam) conducted in 2013:
  - Event 1 was conducted from February to March 2013.
  - Event 2 was conducted in June 2013.

At each location from RM 0 to RM 10.2 and during every sampling event, grab samples were collected from two depths in the water column: 3 ft (0.9 m) above the bottom and 3 ft (0.9 m) below the surface (AECOM 2019b). Samples were collected at four intervals (i.e., high water slack tide, low water slack tide, maximum ebb tide, and maximum flood tide) at each location and depth.

The background dataset was developed using surface water chemistry samples collected from 2011 to 2013 from one location above Dundee Dam during the five routine monitoring events, single low-flow event, and two high-flow events detailed above (Table 4-4, Figure 4-27).

[60] The period of neap tide started on the last day of sampling, when the boats were in Newark Bay.

Table 4-4. Surface water data included in the BERA dataset

Sampling Event	Sampling Period	Description	Number of Locations	Chemical Group	Source
LPRSA					
2011 chemical water column monitoring	August 2011	surface water collection during average tide	5	total and dissolved metals,[a] butyltins, PAHs, alkylated PAHs, SVOCs, PCB congeners, PCDDs/PCDFs, organochlorine pesticides, VOCs, cyanide, DOC, POC, TOC, TDS, chlorophyll-a, and water quality parameters	AECOM (2019b)
2012 chemical water column monitoring 2	February 2012	surface water collection during spring tide	5	total and dissolved metals,[a] PCB congeners, PCDDs/PCDFs, DOC, POC, TOC, TDS, chlorophyll-a, and water quality parameters	AECOM (2019b)
2012 chemical water column monitoring 3	March 2012	surface water collection during neap tide[b]	5	total and dissolved metals,[a] butyltins, PAHs, alkylated PAHs, SVOCs, PCB congeners, PCDDs/PCDFs, organochlorine pesticides, VOCs, cyanide, DOC, POC, TOC, TDS, chlorophyll-a, and water quality parameters	AECOM (2019b)
2012 chemical water column monitoring 4	June 2012	surface water collection during spring tide	5	total and dissolved metals,[a] PCB congeners, PCDDs/PCDFs, DOC, POC, TOC, TDS, and chlorophyll-a, and water quality parameters	AECOM (2019b)
2012 low-flow chemical water column monitoring	August 2012	surface water collection during low flow	5	total and dissolved metals,[a] butyltins, PAHs, SVOCs, alkylated PAHs, PCB congeners, PCDDs/PCDFs, organochlorine pesticides, VOCs, cyanide, DOC, POC, TOC, TDS, chlorophyll-a, and water quality parameters	AECOM (2019b)
2012 chemical water column monitoring 5	December 2012	surface water collection during average tide	5	total and dissolved metals,[a] butyltins, PAHs, alkylated PAHs, SVOCs, PCB congeners, PCDDs/PCDFs, organochlorine pesticides, VOCs, cyanide, DOC, POC, TOC, TDS, TSS, chlorophyll-a, and water quality parameters	AECOM (2019b)
2013 high-flow chemical water column monitoring 1	February to March 2013	surface water collection during high flow	5	total and dissolved metals,[a] butyltins, PAHs, alkylated PAHs, SVOCs, PCB congeners, PCDDs/PCDFs, organochlorine pesticides, VOCs, cyanide, DOC, POC, TOC, TDS, TSS, chlorophyll-a, and water quality parameters	AECOM (2014b)
2013 high-flow chemical water column monitoring 2	June 2013	surface water collection during high flow	5	total and dissolved metals,[a] PCB congeners, PCDDs/PCDFs, DOC, POC, TOC, TDS, TSS, chlorophyll-a, and water quality parameters	AECOM (2014b)

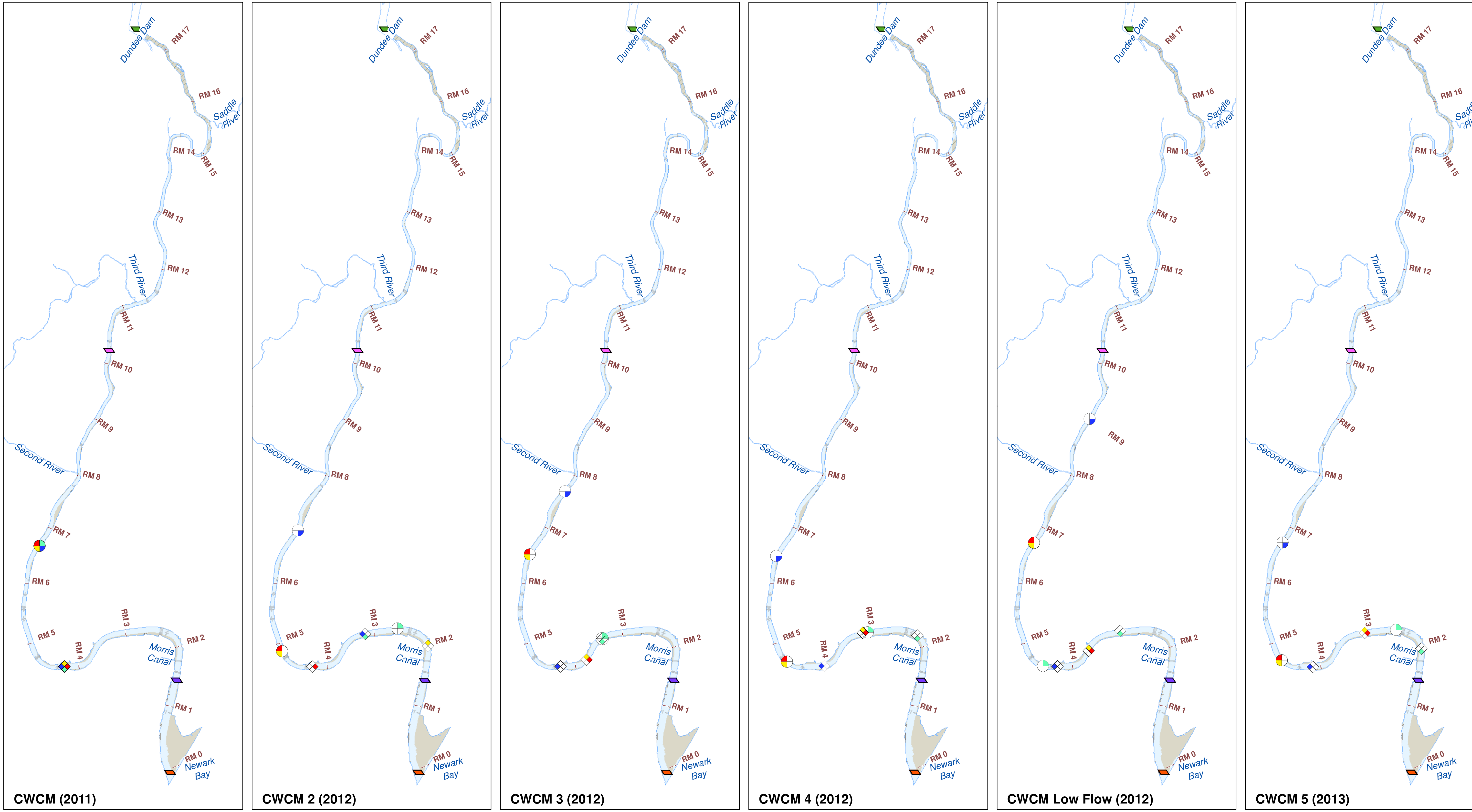
Table 4-4. Surface water data included in the BERA dataset

Sampling Event	Sampling Period	Description	Number of Locations	Chemical Group	Source
Passaic River Above Dundee Dam					
2011 chemical water column monitoring	August 2011	surface water collection during average tide	1	total and dissolved metals,[a] butyltins, PAHs, alkylated PAHs, SVOCs, PCB congeners, PCDDs/PCDFs, organochlorine pesticides, VOCs, cyanide, DOC, POC, TOC, TDS, chlorophyll-a, and water quality parameters	AECOM (2019b)
2012 chemical water column monitoring 2	February 2012	surface water collection during spring tide	1	total and dissolved metals,[a] PCB congeners, PCDDs/PCDFs, DOC, POC, TOC, TDS, chlorophyll-a, and water quality parameters	AECOM (2019b)
2012 chemical water column monitoring 3	March 2012	surface water collection during neap tide[a]	1	total and dissolved metals,[a] butyltins, PAHs, alkylated PAHs, SVOCs, PCB congeners, PCDDs/PCDFs, organochlorine pesticides, VOCs, cyanide, DOC, POC, TOC, TDS, chlorophyll-a, and water quality parameters	AECOM (2019b)
2012 chemical water column monitoring 4	June 2012	surface water collection during spring tide	1	total and dissolved metals,[a] PCB congeners, PCDDs/PCDFs, DOC, POC, TOC, TDS, and chlorophyll-a, and water quality parameters	AECOM (2019b)
2012 low-flow chemical water column monitoring	August 2012	surface water collection during low flow	1	total and dissolved metals,[a] butyltins, PAHs, SVOCs, alkylated PAHs, PCB congeners, PCDDs/PCDFs, organochlorine pesticides, VOCs, cyanide, DOC, POC, TOC, TDS, chlorophyll-a, and water quality parameters	AECOM (2019b)
2012 chemical water column monitoring 5	December 2012	surface water collection during average tide	1	total and dissolved metals,[a] butyltins, PAHs, alkylated PAHs, SVOCs, PCB congeners, PCDDs/PCDFs, organochlorine pesticides, VOCs, cyanide, DOC, POC, TOC, TDS, TSS, chlorophyll-a, and water quality parameters	AECOM (2019b)
2013 high-flow chemical water column monitoring 1	February to March 2013	surface water collection during high flow	1	total and dissolved metals,[a] butyltins, PAHs, alkylated PAHs, SVOCs, PCB congeners, PCDDs/PCDFs, organochlorine pesticides, VOCs, cyanide, DOC, POC, TOC, TDS, TSS, chlorophyll-a, and water quality parameters	AECOM (2014b)
2013 high-flow chemical water column monitoring 2	June 2013	surface water collection during high flow	1	total and dissolved metals,[a] PCB congeners, PCDDs/PCDFs, DOC, POC, TOC, TDS, TSS, chlorophyll-a, and water quality parameters	AECOM (2014b)

[a] Surface water data were evaluated based on dissolved metal concentrations, unless criteria were based on total metal concentrations (see Sections 5.3.1, 6.5, and 7.3 for additional information about surface water assessment criteria).

[b] The period of neap tide started on the last day of sampling, when the boats were in Newark Bay.

BERA – baseline ecological risk assessment
DOC – dissolved organic carbon
LPRSA – Lower Passaic River Study Area
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
PCDD – polychlorinated dibenzo-*p*-dioxin
PCDF – polychlorinated dibenzofuran
POC – particulate organic carbon
SVOC – semivolatile organic compound
TDS – total dissolved solids
TOC – total organic carbon
VOC – volatile organic compound



Windward environmental LLC

0 1 2 Kilometers

0 1 2 Miles

N

Mudflat areas for ecological receptors are those areas where the river bottom slope is ≤ 6° and the depth is ≥ -2 ft MLLW, based on the 2007 CPG bathymetric survey.

Surface water sampling station
- RM 0
- RM 1.4
- RM 10.2
- Above Dundee Dam location

Tidal river 1
- Ebb tide
- Flood tide
- High tide
- Low tide

Tidal river 2
- Ebb tide
- Flood tide
- High tide
- Low tide

- Bridge
- Abutment
- Dock

- River mile
- Mudflat
- Gravel with fines
- LPRSA

Figure 4-27. LPRSA locations and above Dundee Dam locations for surface water chemistry samples

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Baseline Ecological Risk Assessment

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4.2.5 Biological survey data

Survey data were collected to provide qualitative information about fish and avian communities, as well as the shoreline habitat and water quality of the LPRSA (Table 4-5). These data were used in this BERA to provide additional information in the evaluation of potential risks and the overall health of ecological receptors. Benthic invertebrate community survey data were also collected for use in the SQT analysis.

Data from the following surveys conducted within the LPRSA were included:

- Seasonal fish community surveys, including the evaluation of external and internal gross pathology from 2009 to 2010 (sampling methods and locations shown in Figures 4-28 to 4-31)
- Benthic invertebrate community surveys conducted in 2009 and 2010 using surface sediment from the locations shown in Figure 4-32[61]
- Seasonal avian community surveys from 2009 to 2011 (Figure 4-33)
- A habitat survey of the LPRSA and select tributary shoreline features and vegetation conducted in 2010
- Continuous near-bottom (i.e., 8 in. [0.2 m] above bottom) DO monitoring conducted at 11 LPRSA locations in 2012 (Figure 4-34)

Survey data were collected above Dundee Dam to provide qualitative background and reference information about the fish community and water quality, and to provide reference information for the benthic invertebrate community in the freshwater portion of the LPRSA. Data from the following surveys conducted above Dundee Dam were included:

- A fish community survey, including the evaluation of gross pathology, conducted in October 2012 (sampling methods and locations shown in Figures 4-35 and 4-36)
- Continuous near-bottom DO monitoring conducted at two locations above Dundee Dam in 2012 (Figure 4-34)
- A benthic invertebrate community survey conducted in 2012 using surface sediment from the SQT (analyzed for chemistry and toxicity) sediment samples (Figure 4-6)

Regional reference information also included benthic invertebrate community survey data collected from 1993 to 2003 from multiple locations in Jamaica Bay and from 1995 to 2006 from multiple locations in the Mullica River and Great Bay (Figures 4-37 and 4-38, respectively).

[61] The 2009 fall benthic invertebrate community survey was conducted using SQT sediment samples that were analyzed for chemistry and toxicity (Figure 4-5).

Table 4-5. Biological survey data included in the BERA dataset

Sampling Event	Survey Period	Description	Source
LPRSA			
Fish community seasonal surveys	August to September 2009 (late summer/early fall)	surveys of the fish community, including gross internal and external pathology evaluations on select fish	Windward (2010c)
	January to February 2010 (winter)		Windward (2011c)
	June to July 2010 (late spring/early summer)		
	July and August 2010 (summer)	collection effort targeted for SFF tissue	Windward (2011c, 2018c)
Benthic invertebrate community seasonal surveys	October to November 2009 (fall)	benthic invertebrate community data from surface sediment grab samples (0 to 15 cm) at 100 locations	Windward (2014a)
	June 2010 (spring)	benthic invertebrate community data from surface sediment grab samples (0 to 15 cm) at 33 locations	Windward (2014c)
	July to August 2010 (summer)	benthic invertebrate community data from surface sediment grab samples (0 to 15 cm) at 33 locations	
Avian community seasonal surveys	August 2010 (summer)	qualitative survey of birds observed in habitats using transects that were surveyed a total of three times (i.e., at sunrise, midday, and sunset)	Windward (2011a)
	October 2010 (fall)		
	January 2011(winter)		Windward (2019e)
	May 2011 (spring)		
Habitat survey	September 2010	qualitative survey of shoreline features and vegetation within the LPRSA and LPRSA tributaries	(Windward 2014b)
DO monitoring	August 7 to December 9, 2012	continuous near-bottom (i.e., 8 in. above bottom) monitoring for DO, temperature, turbidity, and salinity at 11 locations	Windward (2018e)
Passaic River Above Dundee Dam			
Fish community survey	October 2012	survey of the fish community; gross internal and external pathology evaluations on select fish	Windward (2019c)
Benthic invertebrate community survey	November 2012	benthic invertebrate community data from surface sediment grab samples (0 to 15 cm) at 24 locations	Windward (2019b)
DO monitoring	August 7 to December 9, 2012	continuous near-bottom (i.e., 8 in. above bottom) monitoring for DO, temperature, turbidity, and salinity at two locations	Windward (2018e)

Table 4-5. Biological survey data included in the BERA dataset

Sampling Event	Survey Period	Description	Source
Jamaica Bay			
1993 to 2003 REMAP	September 1993 to August 1998 and July to September 2003	benthic invertebrate community data from 56 surface sediment grab samples (0 to 15 cm) collected from Jamaica Bay (samples were co-located with chemistry analysis and toxicity testing)	USEPA (2011) and USEPA (2016h)
2000 to 2004 NCA Program New Jersey Atlantic Coast	August 2000 to August 2004	benthic invertebrate community data from 7 surface sediment grab samples (0 to 10 cm) collected from Jamaica Bay (samples were co-located with chemistry analysis and toxicity testing)	USEPA (2016e)
Mullica River/Great Bay Estuary and Mullica River Freshwater Area			
2000 to 2006 NCA Program New Jersey Atlantic Coast	September 2000 to August 2006	benthic invertebrate community data from surface sediment grab samples (0 to 10 cm) collected from Mullica River and Great Bay	(USEPA 2016e)
2010 NCCA Program	August 2010	benthic invertebrate community data from surface sediment grab samples (0 to 10 cm) collected from Mullica River and Great Bay	USEPA (2016f)
1995 to 2006 NJDEP	February 1995 to April 2006	benthic invertebrate community data from surface sediment grab samples collected from Mullica River and Great Bay	USEPA (2011)

BERA – baseline ecological risk assessment
EMAP – Environmental Monitoring and Assessment Program
NCA – National Coastal Assessment
NCCA – National Coastal Condition Assessment
NJDEP – New Jersey Department of Environmental Protection
REMAP – Regional Environmental Monitoring and Assessment Program
SFF – small forage fish
USEPA – US Environmental Protection Agency
Windward – Windward Environmental LLC

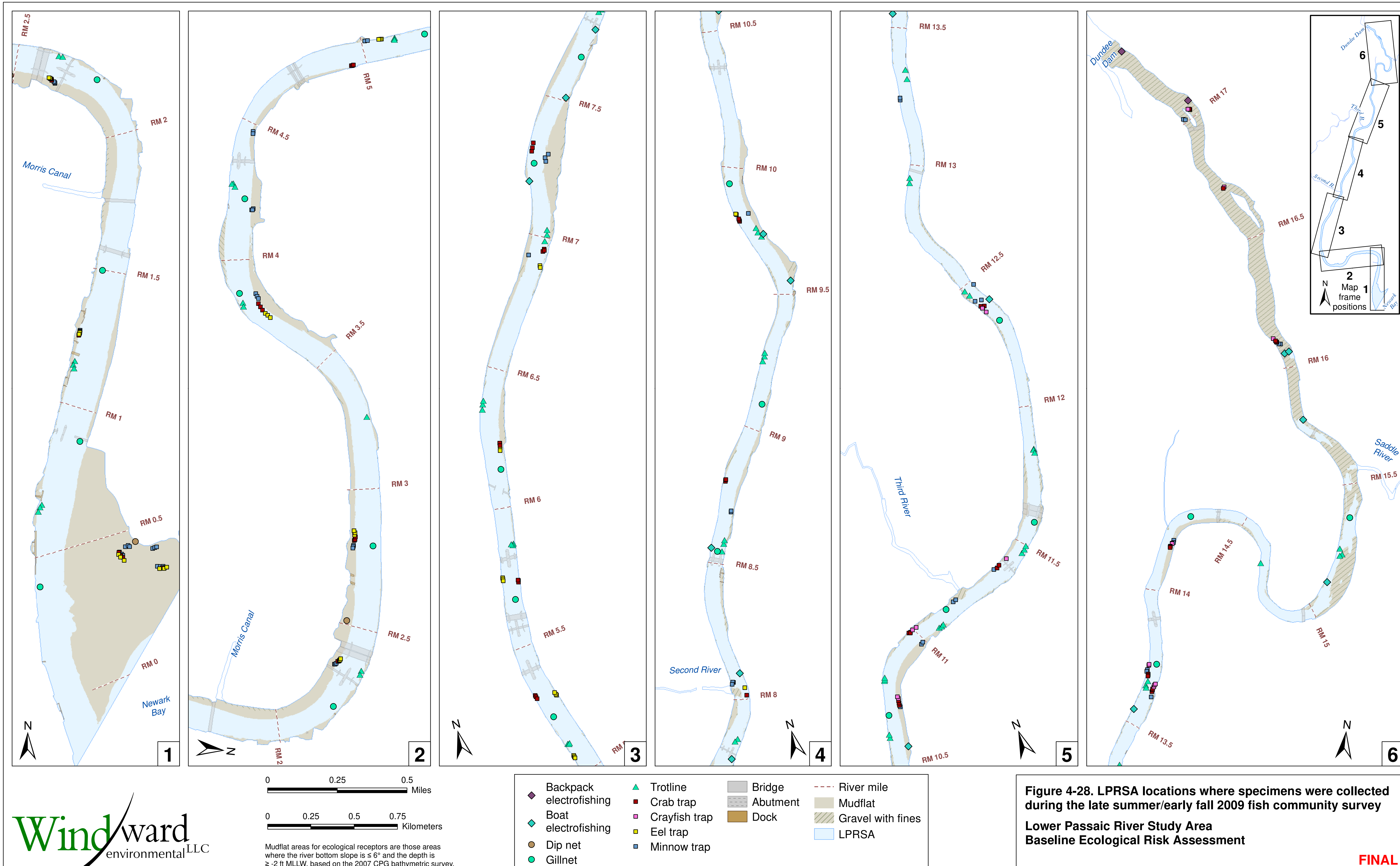

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Figure 4-28. LPRSA locations where specimens were collected during the late summer/early fall 2009 fish community survey

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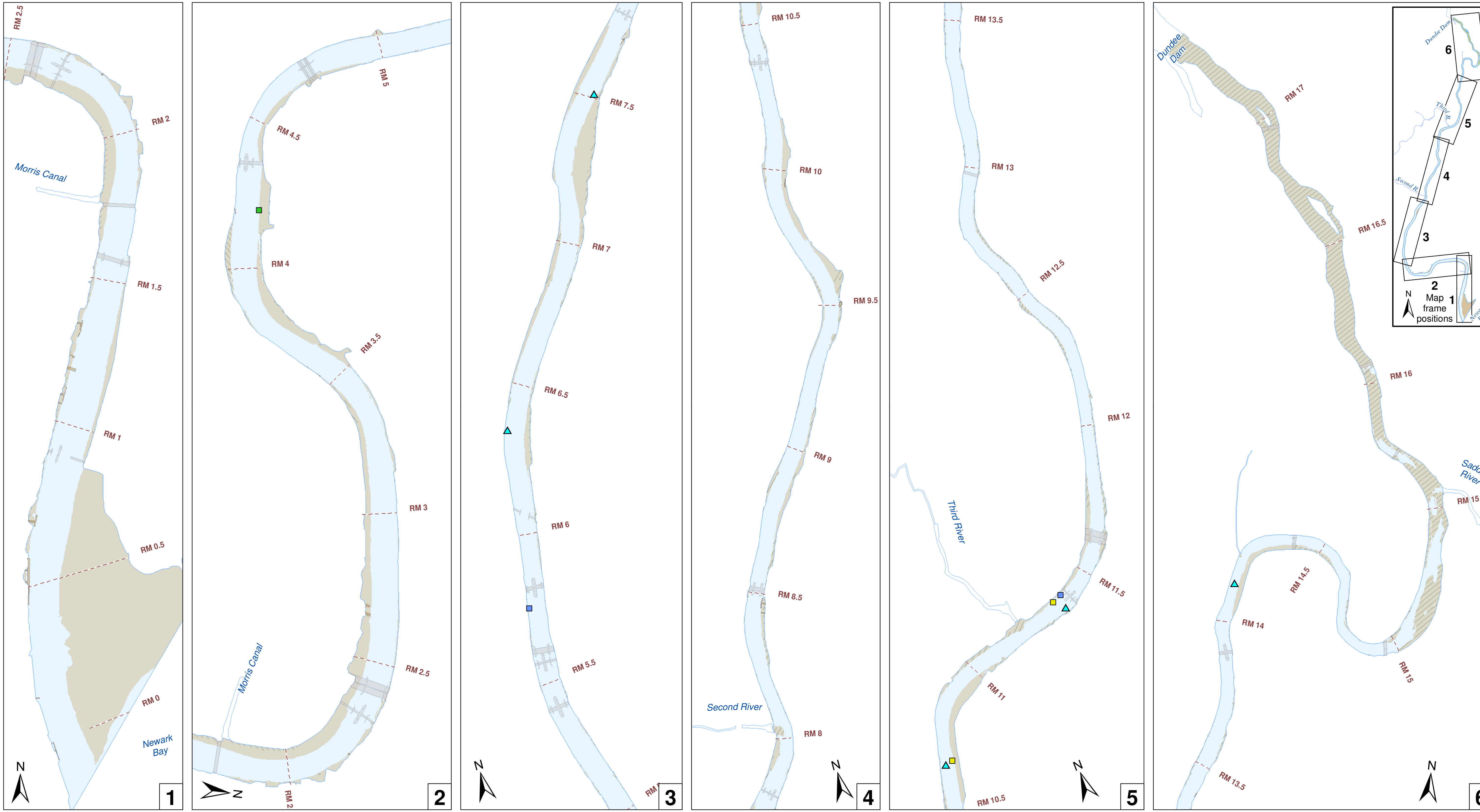

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Mudflat areas for ecological receptors are those areas where the river bottom slope is ≤ 6° and the depth is ≥ -2 ft MLLW, based on the 2007 CPG bathymetric survey.

Figure 4-29. LPRSA locations where specimens were collected during the winter 2010 fish community survey

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Prepared by mikey 5/30/2019; W:\Projects\06-58-01 Passaic RI\Data\GIS\Maps_and_Analysis\BERA\Revised BERA 2016\4355_Catch locs from winter 2010 fish community survey_LSM_20131206.mxd

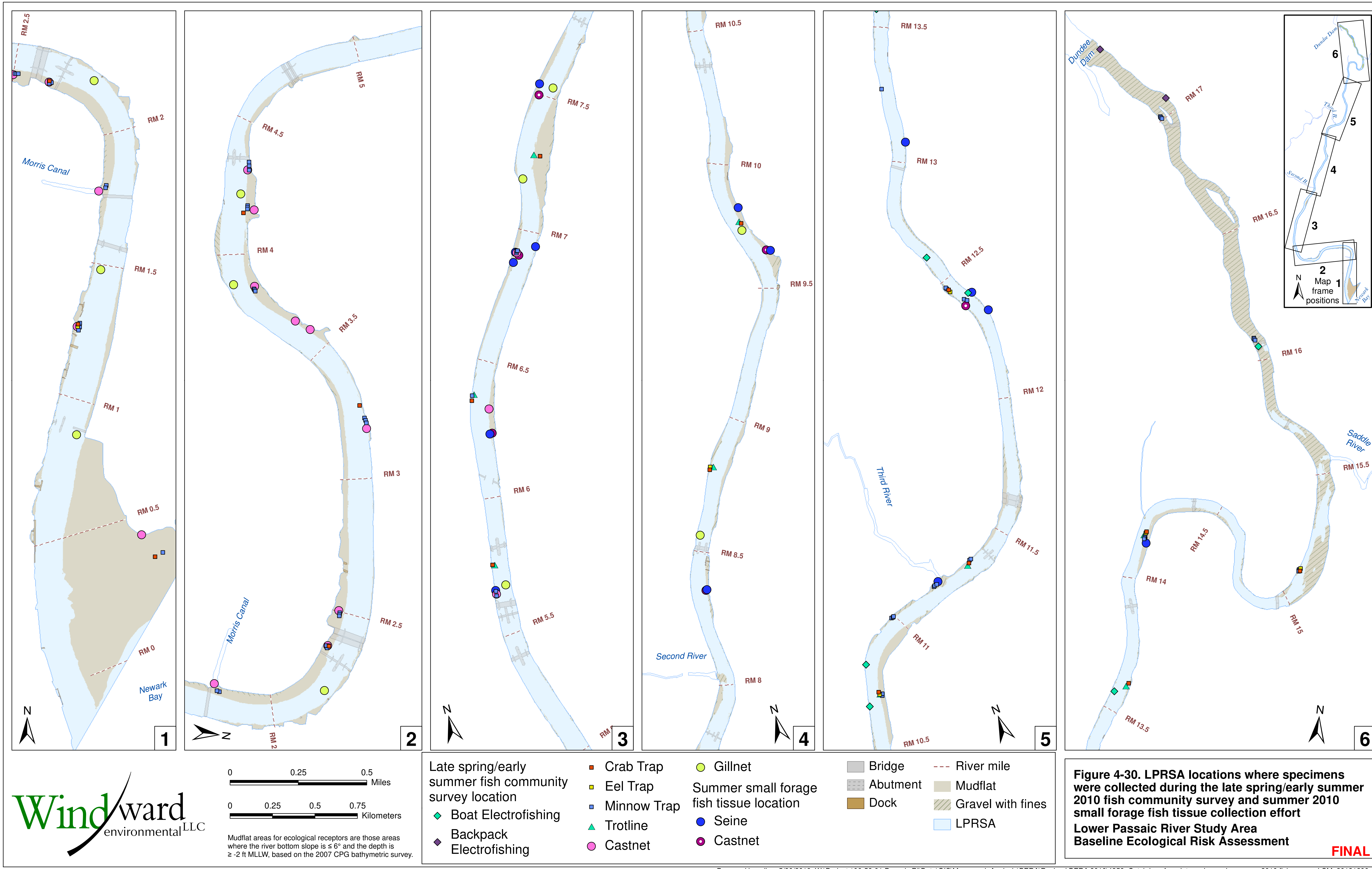

Figure 4-30. LPRSA locations where specimens were collected during the late spring/early summer 2010 fish community survey and summer 2010 small forage fish tissue collection effort

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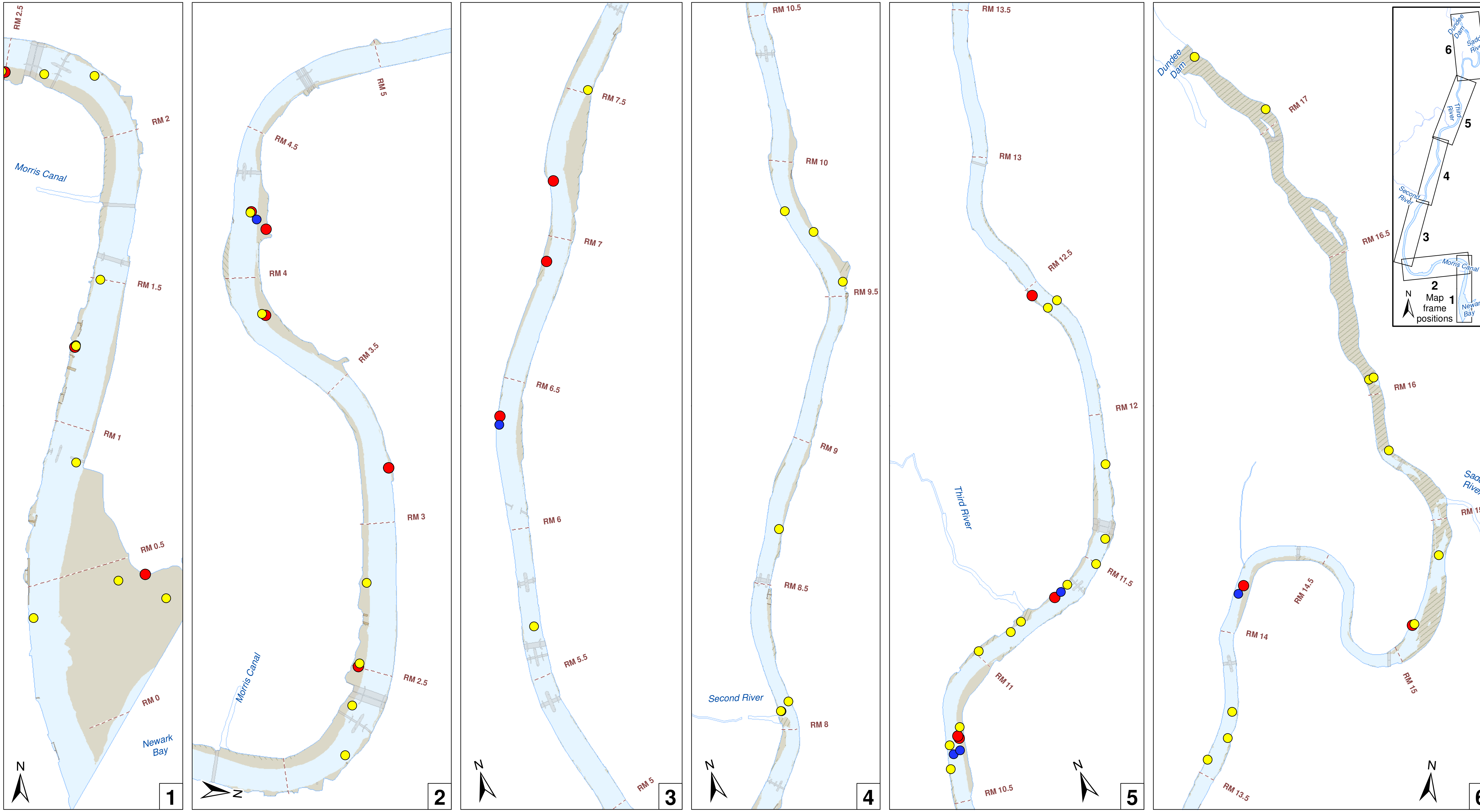

Late summer/early fall 2009
Winter 2010
Late spring/early summer 2010

Bridge
Abutment
Dock

River mile
Mudflat
Gravel with fines
LPRSA

0 0.25 0.5 Miles

0 0.25 0.5 0.75 Kilometers

Windward environmental LLC

Mudflat areas for ecological receptors are those areas where the river bottom slope is ≤ 6° and the depth is ≥ -2 ft MLLW, based on the 2007 CPG bathymetric survey.

Figure 4-31. LPRSA locations where specimens were collected for pathology evaluation during the late summer/early fall 2009, winter 2010, and spring/early summer 2010 fish community surveys

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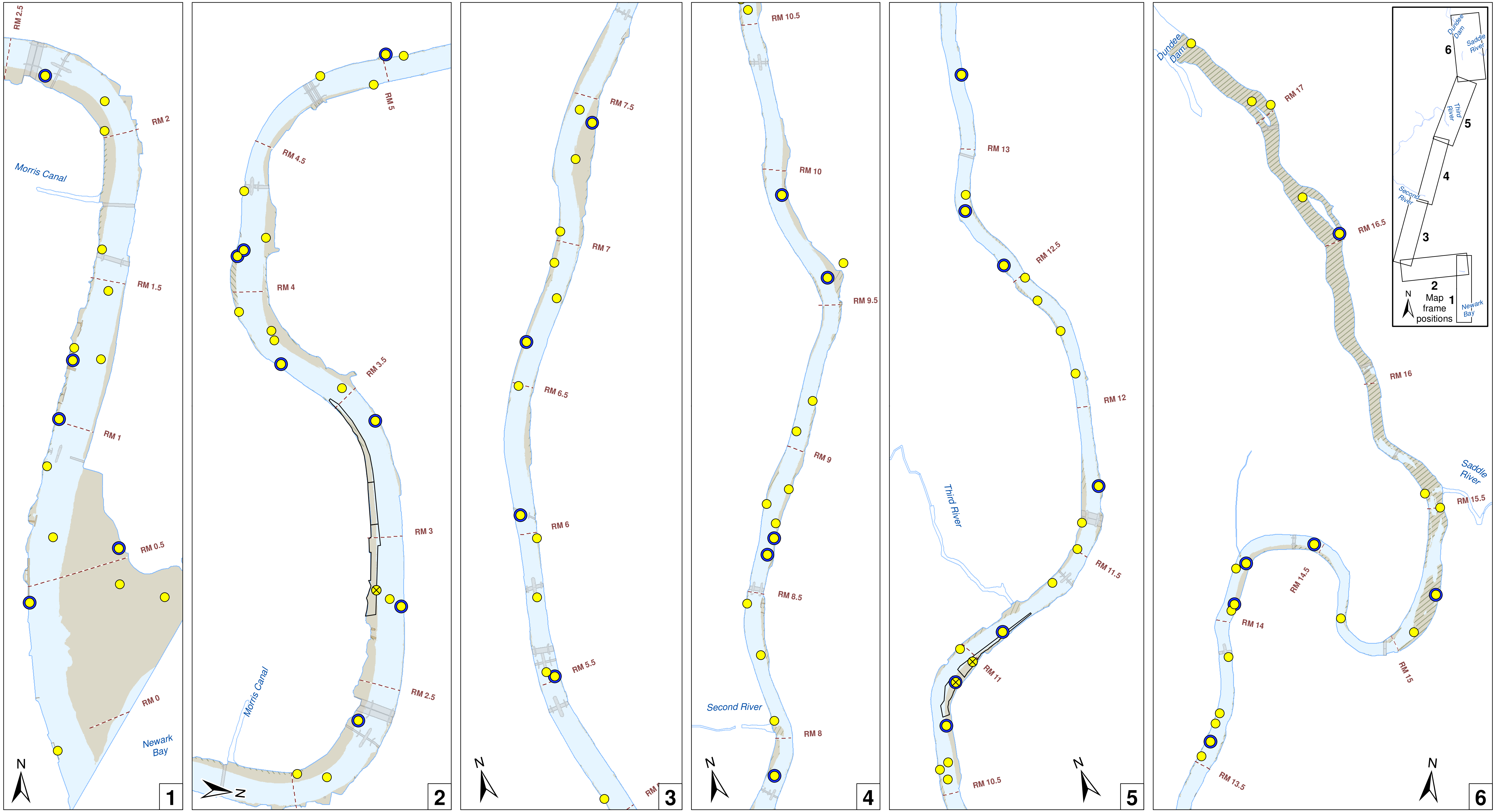

0	0.25	0.5 Miles

0	0.25	0.5	0.75 Kilometers

Legend:

- 2009 Benthic community
- 2010 Benthic community
- Remediated location[a]
- Dredge zone
- Bridge
- Abutment
- Dock
- River mile
- Mudflat
- Gravel with fines
- LPRSA

[a]One sample was collected in the Lister Ave. dredge area at RM 2.8 and two were collected in the RM 10.9 dredge area.

Mudflat areas for ecological receptors are those areas where the river bottom slope is ≤ 6° and the depth is ≥ -2 ft MLLW, based on the 2007 CPG bathymetric survey.

Figure 4-32. LPRSA locations where surface sediment samples were collected during the fall 2009 and spring and summer 2010 benthic invertebrate community surveys

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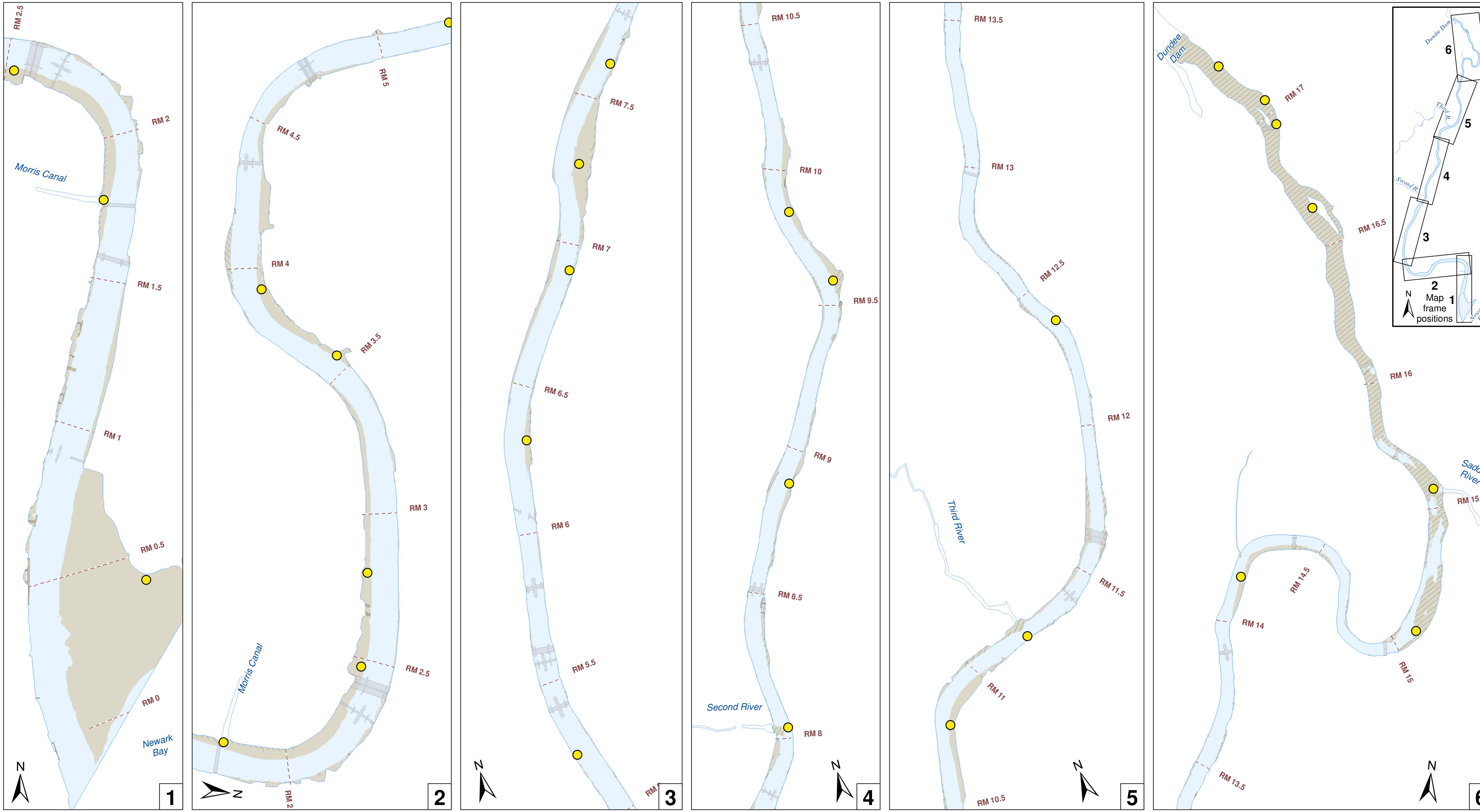

Avian survey location*
Bridge
Abutment
Dock
River mile
Mudflat
Gravel with fines
LPRSA

0 0.25 0.5 Miles

0 0.25 0.5 0.75 Kilometers

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* Some LPRSA survey locations were inaccessible and relocated due to ice blockages during the winter survey. For details, please refer to the Avian community survey data report for the Lower Passaic River Study Area winter and spring 2011 field efforts (Windward [in prep]-i).

Mudflat areas for ecological receptors are those areas where the river bottom slope is ≤ 6° and the depth is ≥ -2 ft MLLW, based on the 2007 CPG bathymetric survey.

Figure 4-33. LPRSA avian survey locations

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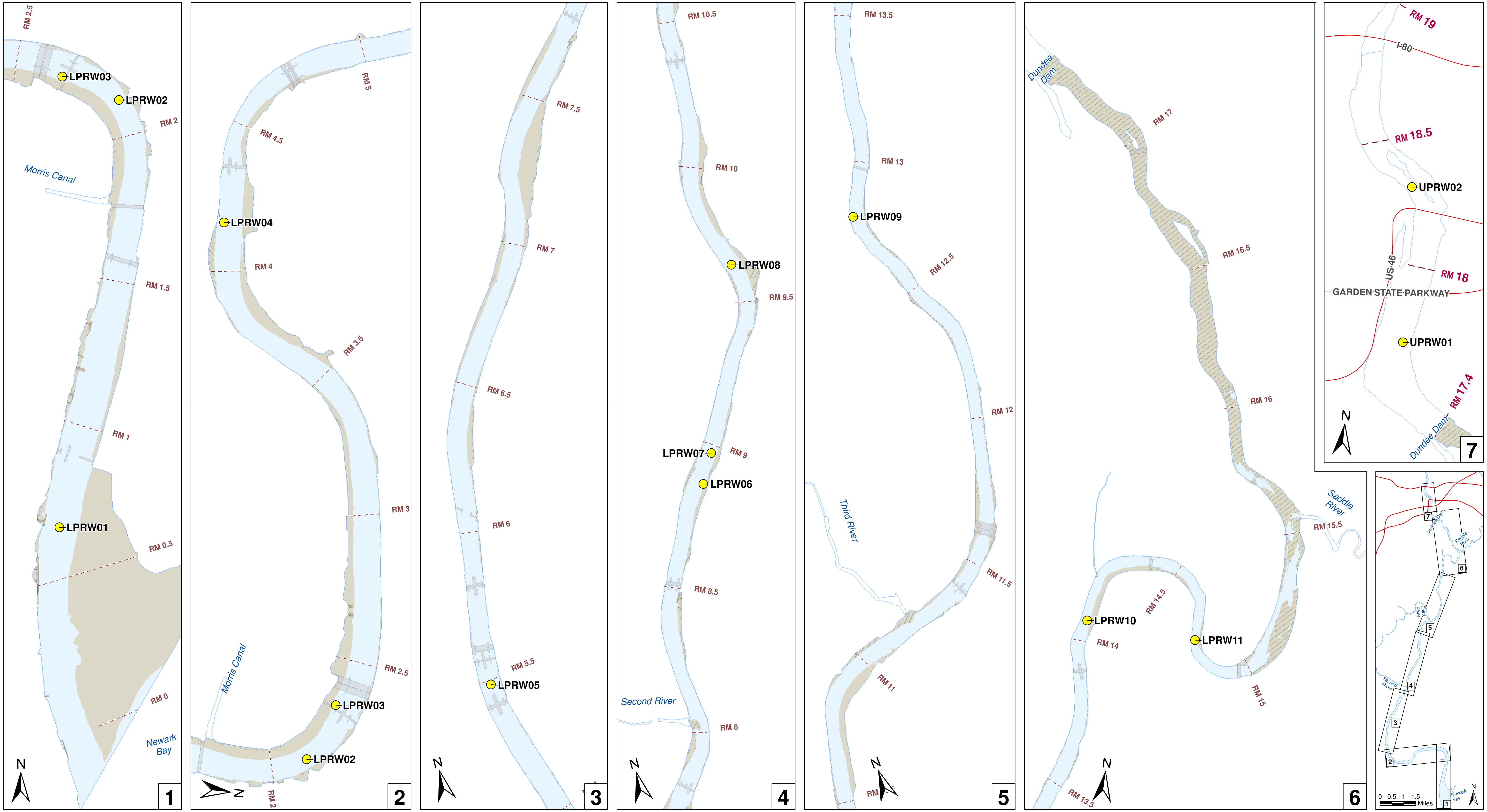

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0 0.25 0.5 Miles

0 0.25 0.5 0.75 Kilometers

Dissolved oxygen monitoring station
River mile
Bridge
Mudflat
Abutment
Gravel with fines
Dock
LPRSA

Mudflat areas for ecological receptors are those areas where the river bottom slope is ≤ 6° and the depth is ≥ -2 ft MLLW, based on the 2007 CPG bathymetric survey.

Figure 4-34. LPRSA and above Dundee Dam locations for monitoring dissolved oxygen

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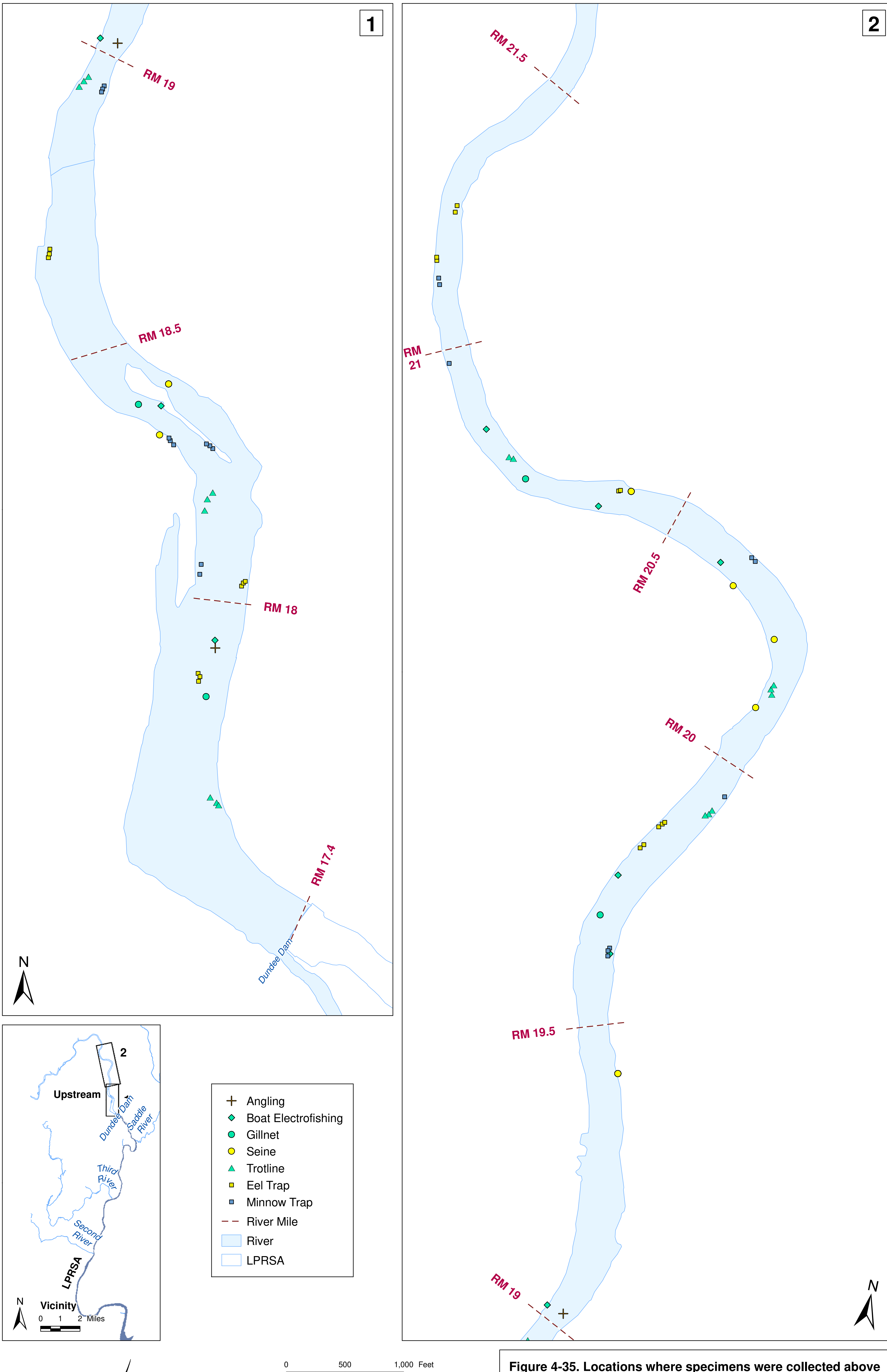

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Figure 4-35. Locations where specimens were collected above Dundee Dam during the fall 2012 fish community survey

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Baseline Ecological Risk Assessment

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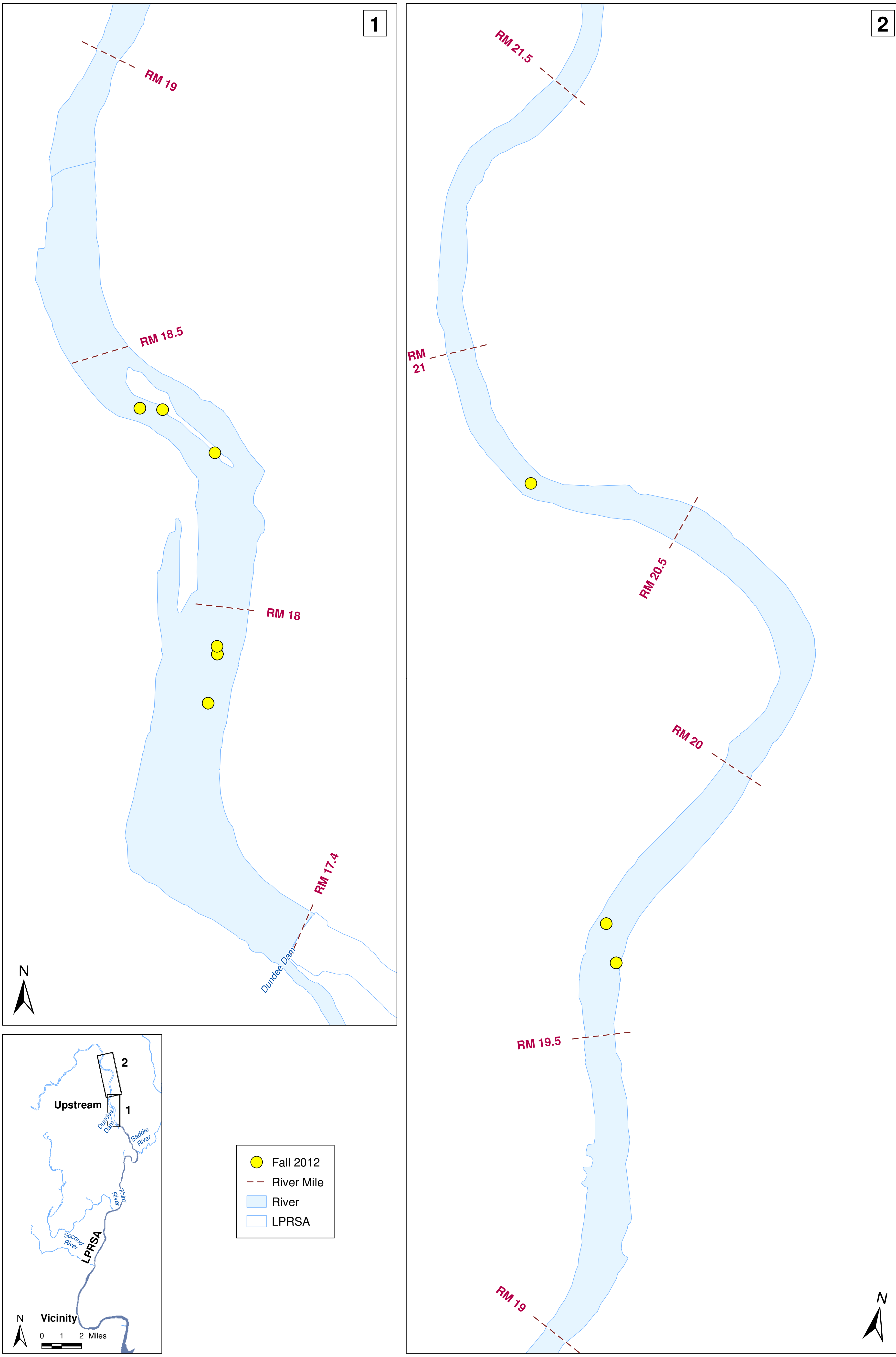

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Figure 4-36. Locations where specimens were collected above Dundee Dam for pathology evaluation during the fall 2012 fish community survey

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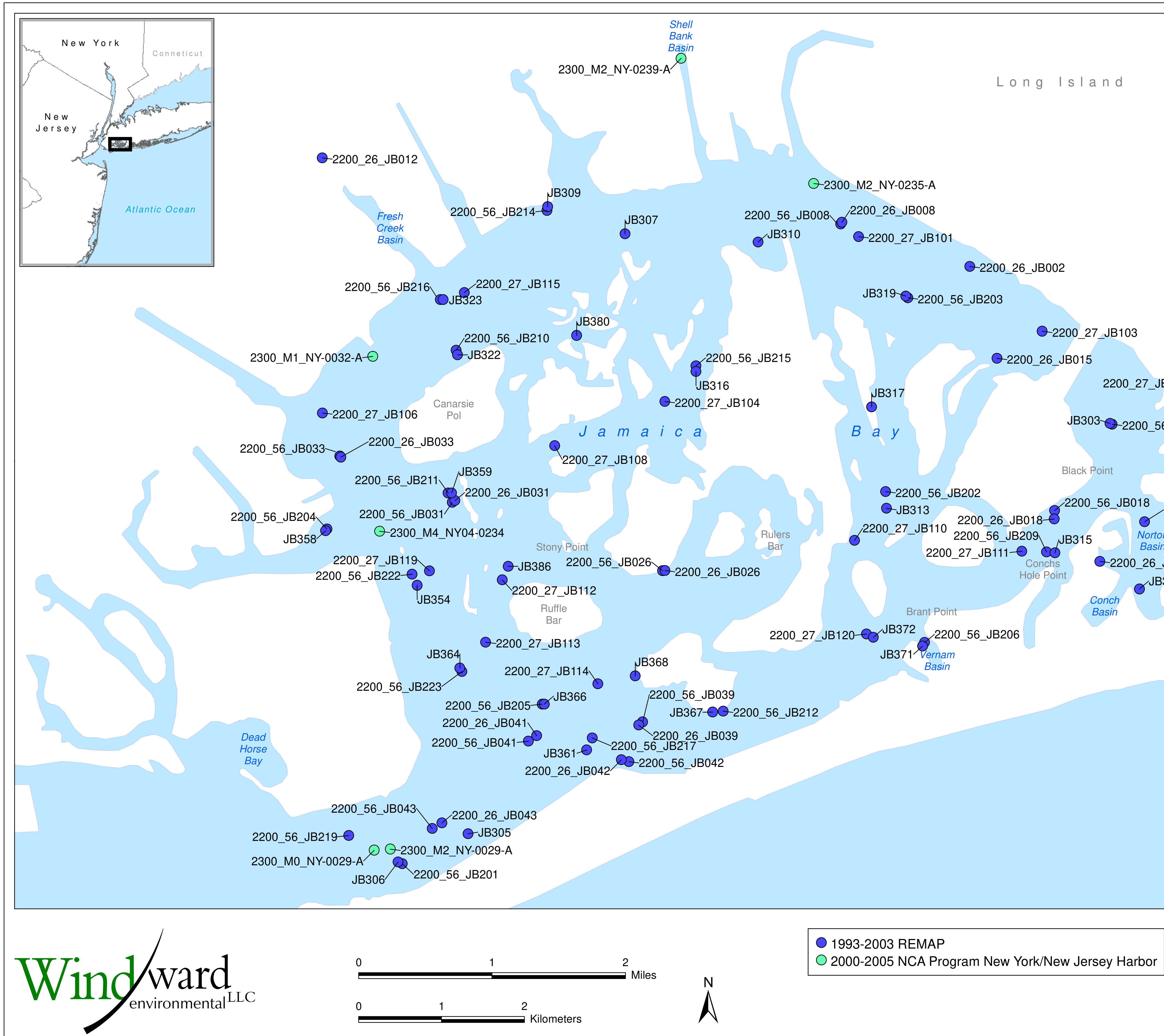

Figure 4-37. Jamaica Bay locations for benthic invertebrate community survey data

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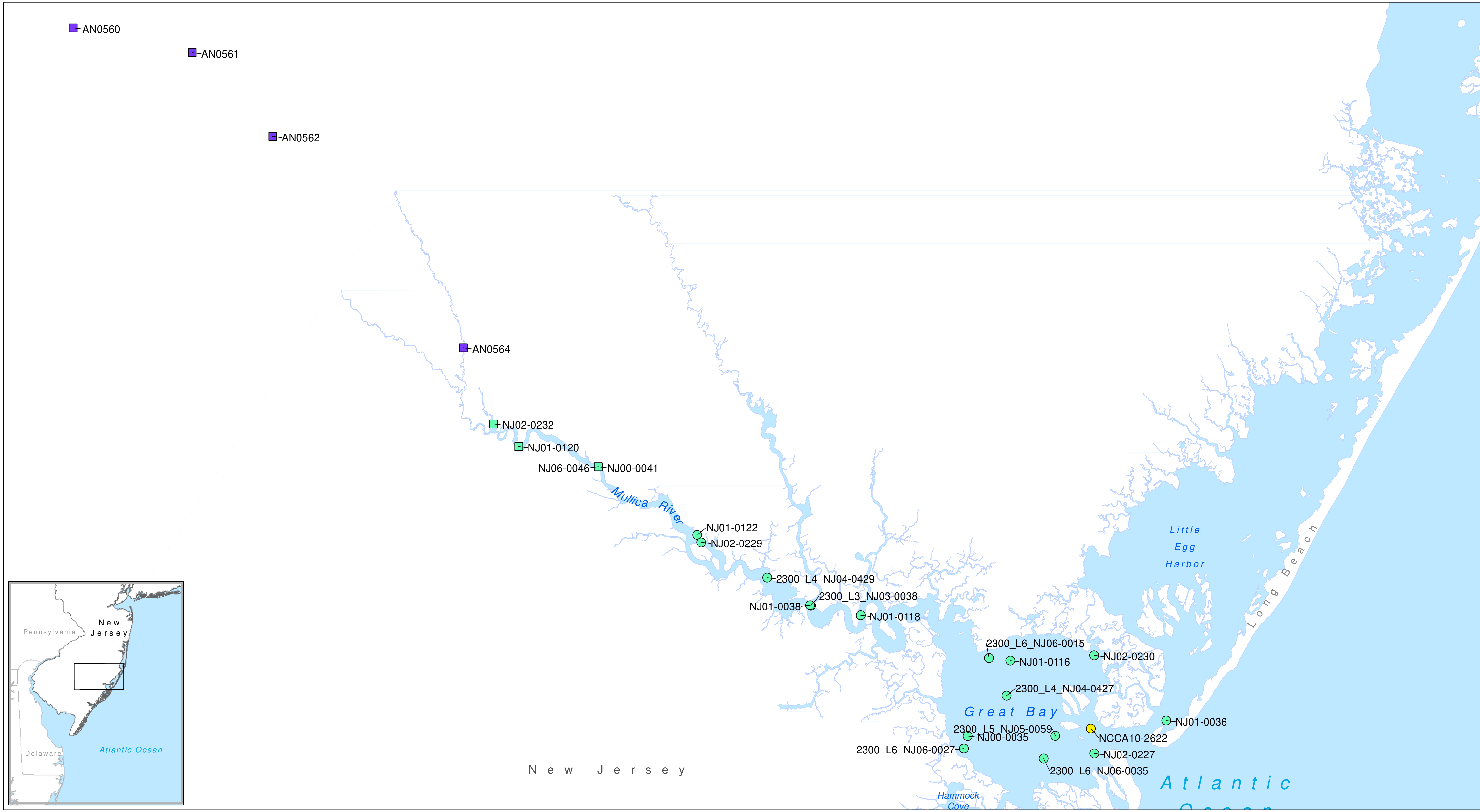

Freshwater
- 1995-2006 NJDEP
- 2000-2006 NCA Program New Jersey Atlantic Coast

Estuarine
- 2000-2006 NCA Program New Jersey Atlantic Coast
- 2010 NCCA Program

Figure 4-38. Mullica River and Great Bay locations for benthic invertebrate community survey data

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4.3 DATA REDUCTION RULES

Data reduction refers to computational methods used to aggregate data. This section presents data reduction methods for the following:

- Calculation of total concentrations (Section 4.3.1)
- Toxic equivalent (TEQ) derivation methods (Section 4.3.2)
- Selection of single results when multiple results were reported (Section 4.3.3)
- Calculation of whole-body concentrations from individual tissue types (Section 4.3.4)
- Normalization of data (Section 4.3.5)
- Determination of the number of signification figures for reporting (Section 4.3.6)
- Calculation of upper confidence limits on the mean (UCLs) (Section 4.3.7)
- Treatment of non-detects in risk calculations (Section 4.3.8)

These methods are consistent with the Data Usability Plan (Windward and AECOM 2015).

4.3.1 Calculated totals

Calculated total concentrations were derived based on the following rules:

- **Rule 1: Non-toxicity-weighted totals (e.g., total polychlorinated biphenyls [PCBs], total polycyclic aromatic hydrocarbons [PAHs])** – The total concentration was calculated based on the sum of the detected chemical constituents (non-detected chemical constituents were treated as zero).[62] If none of the chemical constituents were detected for a given sample, the total concentration was flagged as non-detected (U-qualified), and represented as the highest reporting limit (RL). If any one of the chemical constituents was not reported, partial totals were calculated and flagged. The use of zero for non-detected chemical constituents and the use of partial totals are addressed in the applicable uncertainty analysis sections of this document.
- **Rule 2: Toxicity-weighted totals (i.e., PCB TEQ and polychlorinated dibenzo-*p*-dioxin/polychlorinated dibenzofuran [PCDD/PCDF] toxic TEQ)** – The toxicity-weighted totals for PCBs and PCDDs/PCDFs were calculated by summing each of the detected chemical constituents multiplied by its respective toxic equivalency factor (TEF). TEQs were calculated using USEPA's TEQ calculator (USEPA 2014) using the Kaplan-Meier method to derive TEQs.

[62] The treatment of non-detected results as zero is discussed in the uncertainty section.

Table 4-6 presents the individual chemical constituents of each chemical group and the summation rules. The chemical constituents included in totals were applied to all data that met the acceptability criteria for use in developing risk estimates.

Table 4-6. Chemical groups and summation rules

Chemical Group	Chemical Constituents	Rule[a]
PCBs		
Total PCB congeners[b]	209 PCB congeners	Rule 1
PAHs		
Total HPAHs	benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene,[c] benzo(g,h,i)perylene, benzo(k)fluoranthene, chrysene, dibenzo(a,h)anthracene, fluoranthene, indeno(1,2,3,-c,d)pyrene, and pyrene	Rule 1
Total LPAHs	acenaphthene, acenaphthylene, anthracene, fluorene, naphthalene, and phenanthrene	Rule 1
Total PAHs	acenaphthene, acenaphthylene, anthracene, benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene,[c] benzo(g,h,i)perylene, benzo(k)fluoranthene, chrysene, dibenzo(a,h)anthracene, fluoranthene, fluorene, indeno(1,2,3,-c,d)pyrene, naphthalene, phenanthrene, and pyrene	Rule 1
Total benzofluoranthenes	benzo(b)fluoranthene,[c] benzo(k)fluoranthene	Rule 1
Pesticides		
Total chlordanes	alpha-chlordane, gamma-chlordane, oxychlordane, cis-nonachlor, and trans-nonachlor	Rule 1
Total endosulfan	alpha-endosulfan (Endosulfan I), beta-endosulfan (Endosulfan II), and endosulfan sulfate	Rule 1
Total 4,4′-DDx	4,4′-DDD; 4,4′-DDE; 4,4′-DDT	Rule 1
Total 2,4′- and 4,4′-DDD	2,4′-DDD; 4,4′-DDD	Rule 1
Total 2,4′- and 4,4′-DDE	2,4′-DDE; 4,4′-DDE	Rule 1
Total 2,4′- and 4,4′-DDT	2,4′-DDT; 4,4′-DDT	Rule 1
Total DDx	2,4′-DDD; 2,4′-DDE; 2,4′-DDT; 4,4′-DDD; 4,4′-DDE; 4,4′-DDT	Rule 1
TEQ		
PCDD/PCDF TEQ[d]	all 17 2,3,7,8-substituted PCDD and PCDF congeners	Rule 2
PCB TEQ[d]	12 dioxin-like PCB congeners[e]	Rule 2
Total TEQ[d]	all seventeen 2,3,7,8-substituted PCDD and PCDF congeners and 12 dioxin-like PCB congeners[e]	Rule 2

[a] Rule 1: Only detected chemical constituents were used in the sum; non-detects were treated as zero.
Rule 2: The TEQ was calculated by summing the concentration of each congener multiplied by its corresponding TEF value. TEQs were calculated using USEPA's TEQ calculator (USEPA 2014) using the Kaplan-Meier method to derive TEQs.

[b] Total PCBs were based on total PCB congeners (if available). When calculating a PCB congener sum, the concentration associated with a given co-elution was included in the sum only once.

[c] Benzo(j)fluoranthene, benzo(b/j)fluoranthene, and benzo(j/k)fluoranthene were also included in the HPAH, total PAH, and total benzofluoranthene totals when reported.

[d] TEQs were calculated for mammals, birds, and fish for each TEQ type (PCDDs/PCDFs, PCBs, and total).

[e] The 12 dioxin-like congeners were PCB 77, PCB 81, PCB 105, PCB 114, PCB 118, PCB 123, PCB 126, PCB 156, PCB 157, PCB 167, PCB 169, and PCB 189.

DDD – dichlorodiphenyldichloroethane

PCB – polychlorinated biphenyl

DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
LPAH – low-molecular-weight polycyclic aromatic hydrocarbon
PAH – polycyclic aromatic hydrocarbon
PCDD – polychlorinated dibenzo-*p*-dioxin
PCDF – polychlorinated dibenzofuran
TEF – toxic equivalency factor
TEQ – toxic equivalent
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4-DDD, 2,4′-DDE, 4,4-DDE, 2,4′-DDT, and 4,4′-DDT)

4.3.2 TEQ methodology

As described in Section 4.3.1, the TEQ is a toxicity-weighted value based on the estimated toxicity of various dioxin-like compounds relative to TCDD. Each compound is associated with a TEF of less than one, which represents its relative toxicity relative to TCDD. The TEQ is the sum of the concentrations of the dioxin-like compounds multiplied by their TEFs.[63] There are a number of uncertainties associated with this methodology, as discussed in USEPA (2008); these uncertainties are discussed in the context of receptor group-specific risk characterization in Sections 6 through 9:

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

In addition, recent studies have found that other congeners are more toxic than 2,3,7,8-TCDD, and that the current TEF of 1.0 for two PCDFs (2,3,4,7,8-pentachlorodibenzofuran and 2,3,7,8-tetrachlorodibenzofuran) may underestimate avian toxicity TEFs (Farmahin et al. 2012; Cohen-Barnhouse et al. 2011; Yang et al. 2010). Despite some inherent uncertainties, the TEQ methodology provides

[63] TEQs were calculated using USEPA's TEQ calculator (USEPA 2014) using the Kaplan-Meier method to derive TEQs.

a reasonable, scientifically justifiable, and widely accepted method for estimating risks to ecological receptors in Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) risk assessments (USEPA 2008).

4.3.3 Selection of single result when multiple results were reported

When multiple results were reported for a given sample, only one value was used so that every sample was associated with one result per analyte. The rules for selecting the most appropriate result were applied to all data for use in this BERA. The following subsections present the two types of situations in which a single sample result was selected: when multiple analytical methods were used for the analysis of the same chemical in a single sample, and when multiple results were available as a result of quality control (QC) analyses.

4.3.3.1 Multiple analytical results for a single sample

Multiple validated results for a single sample were sometimes reported for specific analytes. When multiple results were reported for a single parameter, the most appropriate result was selected according to the best result rules, as follows: [64]

- Analyte overlap occurred in the semivolatile organic compound (SVOC) and PAH groups, and the high-resolution results took precedence over the low-resolution results (i.e., high-resolution gas chromatography (HRGC)/high-resolution mass spectrometry [HRMS], HRGC/low-resolution mass spectrometry [MS]-selective ion monitoring [SIM], and gas chromatography [GC]/MS-SIM results took precedence over the low-resolution results [i.e., GC/MS]).
- Analyte overlap occurred in the SVOC and organochlorine pesticide groups (i.e., hexachlorobenzene). The HRGC/HRMS organochlorine results took precedence over the SVOC results.

4.3.3.2 Field duplicates and laboratory replicates

Field duplicates and/or laboratory QC analytical samples might have resulted in more than one analytical result for field-collected samples. QC samples were evaluated as part of the data validation process to ensure that quality assurance (QA)/QC criteria

[64] High-resolution methods offered the benefit of lower detection limits (DLs) than low-resolution methods. For example, benzo(a)pyrene was detected in 35% of the tissue samples using high-resolution methods and not detected in any of the tissue samples using low-resolution methods. In cases where benzo(a)pyrene was detected in tissue by high-resolution methods, the low-resolution DL was, on average, 2,000 times greater than the high-resolution detected results. Benzo(a)pyrene was detected by both high- and low-resolution methods for the majority of the sediment samples. In cases where benzo(a)pyrene was detected in sediment by both methods, the low-resolution DLs were up to 380 times greater than those from the high-resolution method. In cases where the high-resolution method reported a detected result for sediment and the low-resolution method did not, the high-resolution result was, on average, 60% of the low-resolution DL.

were met. If QC samples were analyzed for a given field sample, only the value of the parent sample was used.

Field duplicate results were averaged with the parent sample result using the following rules:[65]

- If both values were detected, the results were averaged to determine a single result.
- If a constituent was detected in only one sample, the detected value was used.
- If a constituent was not detected in either sample, the result was flagged as a non-detect (U-qualified), and the average of the two RLs was used.

Laboratory replicate results were not used; only the value reported with the parent field sample was used.

4.3.4 Calculation of whole-body tissue concentrations

Results for crab and fish tissue that were analyzed as individual tissue types (i.e., fish fillet, fish carcass, crab muscle and hepatopancreas, and crab carcass) were used to calculate whole-body fish and crab concentrations based on the fraction of the whole-body mass represented by each tissue type.

Whole-body fish tissue concentrations were calculated using the following equation:

$$C_{WB} = (C_{fillet} \times f_{fillet}) + (C_{carcass} \times f_{carcass}) \qquad \textbf{Equation 4-1}$$

Where:

C_{WB}	=	estimated whole-body tissue concentration (mg/kg ww)
C_{fillet}	=	fillet tissue concentration (mg/kg ww)
f_{fillet}	=	fraction of whole-body weight that is fillet
$C_{carcass}$	=	carcass tissue concentration (mg/kg ww)
$f_{carcass}$	=	fraction of whole-body weight that is carcass (non-fillet)

Whole-body (i.e., edible meat plus hepatopancreas and carcass) crab tissue concentrations were calculated using the following equation:

$$C_{WB} = (C_{muscle+HP} \times f_{muscle+HP}) + (C_{carcass} \times f_{carcass}) \qquad \textbf{Equation 4-2}$$

Where:

C_{WB}	=	estimated whole-body soft-tissue concentration (mg/kg ww)

[65] Field duplicates were collected for 10% of the sediment and surface water samples collected. Field duplicates exceeded the QAPP-approved limit of 50% relative percent difference in only 0.8% of the sediment and surface water samples. Of these samples, approximately one-half of the field duplicates had higher concentrations than the parent sample (and approximately one-half had concentrations less than the field duplicate). Therefore, risk is assumed to be an accurate representation based on the treatment of field duplicates.

$C_{muscle+HP}$ = muscle (edible meat) and hepatopancreas tissue concentration (mg/kg ww)
$f_{muscle+HP}$ = fraction of whole-body weight that is muscle (edible meat) and hepatopancreas
$C_{carcass}$ = carcass tissue concentration (mg/kg ww)
$f_{carcass}$ = fraction of whole-body weight that is carcass (non-muscle, non-hepatopancreas tissue)

For calculated whole-body fish or crab concentrations that included a non-detected value for at least one tissue type, the non-detected value(s) were represented in the calculation by one-half the RL. In cases where both tissue types were non-detected values, the final calculated whole-body result was flagged as a non-detected result (U-qualified). The uncertainties associated with the treatment of non-detected concentrations in calculating whole-body tissue concentrations were evaluated and are presented in the uncertainty analyses, as appropriate.

4.3.5 Normalization

Both normalized and non-normalized data were considered in the evaluation of sediment data. When applicable (e.g., when sediment criteria were based on OC-normalized values), OC-normalized sediment concentrations were calculated.

Sediment concentrations that were OC-normalized were calculated on a sample-specific basis using the following equation and the TOC data:

$$C_{sed,OC} = \frac{C_{sed,dw}}{f_{oc}}$$ **Equation 4-3**

Where:

$C_{sed,OC}$ = OC-normalized sediment chemical concentration (mg/kg OC)
$C_{sed,dw}$ = dry weight (dw) sediment chemical concentration (mg/kg)
f_{OC} = fraction OC, dry weight basis (%TOC/100)

Chemical concentrations in bivalve mollusk (mussel) tissue were normalized by subtracting the control (i.e., Day 0) from the final field-exposed mussel concentrations to account for non-LPRSA accumulation already present in mussels before they were placed in the LPRSA.[66] Day 0 and final field exposure mussel concentrations, as well as the calculated normalized concentrations (i.e., the difference between Day 0 and final field exposure concentrations), can be found in Attachment K.

[66] Field-exposed mussel concentrations were normalized to the RL when COI concentrations were below RLs in the control (Day 0) mussels.

When applicable, lipid-normalized tissue concentrations were calculated on a sample-specific basis using the following equation:

$$C_{tis,lipid} = \frac{C_{tis,ww}}{f_{lipid}} \qquad \textbf{Equation 4-4}$$

Where:

$C_{tiss,lipid}$	=	lipid-normalized tissue chemical concentration (mg/kg-lipid)
$C_{tiss,ww}$	=	wet weight tissue chemical concentration (mg/kg ww)
f_{lipid}	=	fraction lipid, wet weight basis (% lipid/100)

4.3.6 Significant figures

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

4.3.7 Calculating UCLs

UCL concentrations used to represent EPCs were calculated using USEPA's ProUCL® statistical package (Version 5.0.00) (USEPA 2013c) and were derived following USEPA guidance for calculating UCLs for EPCs at hazardous waste sites (2002a). USEPA's ProUCL® software can both test the goodness of fit for a given dataset and can calculate central tendency and UCLs of the dataset. The UCL recommended by USEPA's ProUCL® software (typically the 95% UCL, but in some cases the 97.5% or even the 99% UCL) was used. The UCL is a statistic that estimates the mean concentration with a specified degree of confidence, and accounts for variability among the sampling data. For datasets with fewer than six detected samples, a UCL was not calculated, and instead the maximum concentration was used to represent an EPC. In cases where statistically derived UCLs were greater than the maximum detected concentration, the maximum detected concentration was used in place of the UCL concentration to represent the upper-bound value.[67] UCLs used to represent EPCs in this BERA are summarized in Appendix C.

4.3.8 Treatment of non-detects in risk calculations

ProUCL® has an option for handling non-detect data (USEPA 2013c). All data (detected and non-detected) were used in UCL calculations. The sensitivity of the treatment of non-detects was evaluated in the uncertainty sections of the risk assessments. TEQs were calculated using USEPA's TEQ calculator (USEPA 2014) using the Kaplan-Meier method to derive TEQs.

[67] Cases where the maximum concentration was used instead of a UCL are identified in Appendix C, as appropriate.

5 SLERA Summary

This section summarizes the results of the SLERA, which are presented in detail in Appendix A. The SLERA was conducted to identify COPECs using a risk-based screening process for each exposure medium. The SLERA for the entire 17.4 mi of the LPRSA was conducted and prepared in accordance with Section IX.37.d of the May 2007 Settlement Agreement (USEPA 2007a). The SLERA is consistent with comments, responses, and directives received from USEPA on June 30, 2017 (USEPA 2017b), September 18, 2017 (USEPA 2017e), July 10, 2018 (USEPA 2018), January 2, 2019 (CDM 2019), and March 5, 2019 (USEPA 2019); during face-to-face meetings or conference calls on July 24, September 27, October 3, November 6, 2017, July 24, 2018, and August 16, 2018; and via additional deliverables and communications between the CPG and USEPA from August through December 2017, July through September 2018, and January through June 2019.

Conservative assumptions were used in the SLERA to provide a quantitative comparison between conservative exposure and effects levels in order to: 1) identify substances that can be eliminated from further consideration because they are unlikely to pose risk to ecological receptors, 2) identify COPECs that warrant further consideration in this BERA, and 3) identify chemicals that will be addressed in this BERA uncertainty section. The SLERA provides information that will allow risk assessors and risk managers to decide the level of evaluation necessary for the next step in the ERA process, referred to as a scientific/management decision point.

Per USEPA (1997a), the primary objective of the SLERA was to provide information to the risk manager to confirm one of three scenarios: 1) there is adequate information to conclude that ecological risks are negligible and therefore, there is no need for remediation on the basis of ecological risk; 2) the information is not adequate to make a decision at this point, and the ERA process will continue; or 3) the information indicates the potential for adverse ecological effects, and a more thorough assessment is warranted.

5.1 SLERA Approach

The SLERA was conducted for the assessment and measurement endpoints as summarized in Table 5-1. COPECs were identified for each receptor group as the chemicals measured in the exposure media at a concentration equal to or exceeding a toxicity screening value (TSV). To ensure that potential ecological risks were not overlooked in the identification of COPECs, the SLERA used conservative assumptions, as follows:

- Receptors are exposed to the maximum detected concentration or maximum calculated dose from the LPRSA media.
- Receptors are exposed 100% of the time.
- Receptors obtain 100% of their diet from their exposure area within the LPRSA.
- Chemicals are 100% bioavailable to receptors.

Table 5-1. Summary of ecological assessment endpoints, receptor groups, species, and data types used for COPEC identification

Assessment Endpoint	Ecological Receptor Group	Feeding Guild	Species	Type of Data				
				Whole-body Tissue Chemistry	Dietary Dose[a]	Surface Water Chemistry	Sediment Chemistry	Egg Tissue Chemistry
Protection and maintenance (i.e., survival, growth, and reproduction) of the benthic invertebrate community, both as an environmental resource in itself and as one that serves as a forage base for fish and wildlife populations	benthic invertebrate community	na	multiple infaunal species, including *Hyalella azteca*, *Chironomus dilutus*, *Ampelisca abdita,* polychaetes (i.e., *Nereis virens*), and oligochaetes (i.e., *Lumbriculus variegatus*)	X[b]		X	X	
Protection and maintenance (i.e., survival, growth, and reproduction) of healthy populations of blue crab and crayfish that serve as a forage base for fish and wildlife populations and as a base for sports fisheries	macroinvertebrate populations	na	blue crab[c]	X		X	X	
Protection and maintenance (i.e., survival, growth, and reproduction) of healthy mollusk populations	bivalve mollusk populations	na	estuarine (ribbed) mussel (*Geukensia demissa*) and freshwater mussel (*Elliptio complanata*)	X		X	X	

Table 5-1. Summary of ecological assessment endpoints, receptor groups, species, and data types used for COPEC identification

Assessment Endpoint	Ecological Receptor Group	Feeding Guild	Species	Type of Data				
				Whole-body Tissue Chemistry	Dietary Dose[a]	Surface Water Chemistry	Sediment Chemistry	Egg Tissue Chemistry
Protection and maintenance (i.e., survival, growth, and reproduction) of omnivorous, invertivorous, and piscivorous fish populations that serve as a forage base for fish and wildlife populations and of fish populations that serve as a base for sports fishery	fish populations	benthic omnivorous	mummichog, other SFF (i.e., gizzard shad, mixed forage fish, pumpkinseed, silver shiner, and spottail shiner), and common carp	X	X	X	X[a]	X
		invertivorous	white perch, channel catfish, brown bullhead, white catfish, and white sucker	X	X	X	X[a]	
		piscivorous	American eel, largemouth bass, smallmouth bass, and northern pike	X	X	X	X[a]	
Protection and maintenance (i.e., survival, growth, and reproduction) of herbivorous, omnivorous, sediment-probing, and piscivorous bird populations	bird populations	sediment-probing invertivorous	spotted sandpiper		X		X[a]	
		piscivorous	great blue heron, belted kingfisher		X		X[a]	X
Protection and maintenance (i.e., survival, growth, and reproduction) of aquatic mammal populations	mammal populations	piscivorous	river otter[d]		X		X[a]	
		omnivorous	mink[d]		X		X[a]	
Maintenance of zooplankton communities that serve as a food base for juvenile fish	zooplankton community	na	multiple species			X		

Table 5-1. Summary of ecological assessment endpoints, receptor groups, species, and data types used for COPEC identification

Assessment Endpoint	Ecological Receptor Group	Feeding Guild	Species	Type of Data				
				Whole-body Tissue Chemistry	Dietary Dose[a]	Surface Water Chemistry	Sediment Chemistry	Egg Tissue Chemistry
Protection and maintenance (i.e., survival, growth, and reproduction) of healthy amphibian and reptile populations	amphibian and reptile populations	na	multiple species[e]		X[f]	X	X[f]	
Maintenance of healthy aquatic plant populations as a food resource and habitat for fish and wildlife populations	aquatic plant community	na	multiple species, including submerged macrophytes[g]			X[h]		

[a] Dietary LOEs include whole-body tissue chemistry data (for prey ingestion) and sediment chemistry data (for incidental SI); bird and mammal dietary LOEs also include surface water chemistry data (for drinking water ingestion).

[b] Laboratory-exposed freshwater and estuarine infaunal invertebrates (i.e., *Nereis virens* and *Lumbriculus variegatus*) are termed estuarine and freshwater worms, respectively.

[c] Crayfish were identified in the PFD as representing freshwater macroinvertebrate populations. However, per agreement between CPG and USEPA (Windward 2010g), blue crab will be the only species used to represent the macroinvertebrate population, for both the estuarine and freshwater portions of the LPRSA, because of the limited number of crayfish collected in the freshwater portion.

[d] The selected semi-piscivorous mammal (i.e., river otter) is expected to be protective of herbivorous mammals (e.g., muskrat) because piscivorous mammals feed on organisms that are higher on the food chain. Mink will also be assessed because possible mink tracks were observed near Dundee Dam during the CPG LPRSA FSP2 biological surveys.

[e] Amphibians and reptiles have a limited presence in the estuarine portion of the LPRSA.

[f] A quantitative screening of dietary doses and/or surface sediment concentrations was not conducted for amphibians and reptiles because TSVs were not available for diet and sediment for this receptor group. The potential risks to amphibians and reptiles from dietary and direct sediment contact exposure are unknown.

[g] The aquatic plant community in the LPRSA is limited by the physical development of the shoreline and poor light penetration of the water.

[h] Limited sediment toxicity data are available for the development of effects thresholds for sediment and aquatic plants; aquatic plant screening-level effects thresholds for sediment are based on soil-based thresholds.

COPEC – chemical of potential ecological concern
CPG – Cooperating Parties Group
FSP – field sampling plan
LOE – line of evidence
LPRSA – Lower Passaic River Study Area
na – not applicable
PFD – problem formulation document
SFF – small forage fish
SI – sediment ingestion
TSV – toxicity screening value
USEPA – US Environmental Protection Agency

5.2 COPEC SCREENING METHODS

A step-wise screening process was conducted to identify a list of COPECs for each receptor group (see flowcharts in Figures 3-1 and 3-2 of Appendix A). The steps specified in the screening process are described below.

- **Step 1 –** Any chemical detected in the exposure media applicable to the receptor group (i.e., sediment, surface water, tissue, or prey) was identified as a chemical of interest (COI) for that group.
- **Step 2 –** TSVs were identified for screening COIs. COIs with no TSVs were retained for further discussion in this BERA.
- **Step 3 –** If the maximum detected COI or dose (calculated using the maximum detected concentration of the COI) was greater than or equal to the TSV, then the COI was identified as a COPEC.

In addition, DLs of both COIs with detected concentrations less than TSVs and of chemicals never detected (non-COIs) were evaluated to determine if DLs exceeded TSVs. Any chemicals for which the DLs did exceed TSVs were retained for further discussion in this BERA.

5.3 EXPOSURE ASSESSMENT

The exposure assessment consisted of the selection of chemical concentrations to represent the exposure of ecological receptor groups to COIs that were identified in the first step of the screening process.

5.3.1 Tissue, sediment, and surface water LOEs

The maximum detected tissue, surface sediment, or surface water COI concentrations from within the selected species' exposure areas were used as the screening-level concentrations. Exposure areas included the entire site for all species evaluated for tissue, sediment, or surface water LOEs, with the exception of amphibians and reptiles, which had a freshwater exposure area from RM 4 to RM 17.4. For the purposes of screening, sediment and surface water data from RM 0 to RM 13 were compared to marine/estuarine criteria and sediment, and surface water data from RM 4 to RM 17.4 were compared to freshwater criteria.

5.3.2 Dietary dose LOE

The screening-level dietary doses for fish, birds, and mammals were estimated based on ingestion of prey, incidental ingestion of sediment, and ingestion of surface water (as applicable). Dietary doses were estimated as the amount of each COPEC ingested per day on a body weight-normalized basis. Exposure in the diet for each species selected for evaluation was calculated using the maximum detected concentrations in tissue of any prey type consumed by the species, sediment from the exposure area of the species (i.e., the entire LPRSA or mudflat areas only), and surface water from RM 4 to RM 17.4 (i.e., freshwater areas). The equation and exposure assumptions used for the dietary dose calculations in the LPRSA are presented in Appendix A. Species-specific body weights and ingestions rates for food (prey), sediment, and water used in the dietary dose estimations were obtained from the literature. Potential prey types for species evaluated using the dietary dose LOE included only those for which tissue chemistry data from the LPRSA were available.

5.3.3 Egg tissue LOE

Fish and bird egg tissue concentrations were estimated for selected species for the egg tissue LOE. Equations and their sources are detailed in Appendix A. Screening-level fish egg tissue concentrations for mummichog were estimated using a chemical-specific adult-to-egg conversion factor (CF). Screening-level bird egg tissue concentrations for belted kingfisher and great blue heron were estimated using biomagnification factors (BMFs).

5.4 Effects Assessment

The receptor group pathway-specific TSVs used in the SLERA are presented in detail in Appendix A. TSVs were identified for tissue, sediment, surface water, and dietary doses. The maximum concentrations or dietary doses were compared to these TSV to derive a hazard quotient (HQ) using the following equation:

$$HQ = \frac{MDC \ \text{ or } \ Dose}{TSV} \qquad \textbf{Equation 5-1}$$

Where:

HQ = hazard quotient (unitless)
MDC = maximum detected concentration
Dose = calculated exposure dose (based on maximum detected concentrations)
TSV = toxicity screening value

COIs with HQs ≥ 1.0 were identified as COPECs. Calculated HQs for all LOEs evaluated are presented in Attachment A1 of the SLERA (Appendix A).

Tissue and dietary dose TSVs used in the SLERA are based on previous documents developed by USEPA Region 2 for the LPRSA, if available (i.e., USEPA's first draft of the LPR restoration project focused feasibility study [FFS] (Malcolm Pirnie 2007b), USEPA's revised draft of the FFS (Louis Berger et al. 2014), or the USEPA's LPR pathways analysis report [PAR] (Battelle 2005)). These TSVs are consistent with comments, responses, and directives received from USEPA on June 30, 2017 (USEPA 2017b), September 18, 2017 (USEPA 2017e), July 10, 2018 (USEPA 2018), January 2, 2019 (CDM 2019), and March 5, 2019 (USEPA 2019); during face-to-face meetings or conference calls on July 24, September 27, October 3, November 6, 2017, July 24, 2018, and August 16, 2018; and via additional deliverables and communications between the CPG and USEPA from August through December 2017, July through September 2018, and January through June 2019. TSVs were also identified from a comprehensive search and review of the toxicological literature (AECOM 2019b).

5.5 SLERA Results

The following summarizes the COPECs identified from the SLERA for further evaluation in this BERA and discusses additional COIs and chemicals that were also identified in the SLERA as needing further discussion.

5.5.1 COPECs

COPECs across all receptor groups include metals, PAHs, organochlorine pesticides, PCDDs/PCDFs, PCBs, SVOCs, volatile organic compounds (VOCs), and cyanide. These COPECs are presented in Table 5-2, and their bases for selection are presented in Appendix A. It was concluded in the SLERA that a BERA was warranted to provide a more site-specific and detailed assessment of chemicals that pose potential risk to ecological receptor groups.

Table 5-2. Summary of COPECs

COPEC	Sediment		Surface Water		Tissue				Diet		
	Benthic Invertebrates	Aquatic Plants	Invertebrates, Fish, Aquatic Plants, and Zooplankton	Amphibians/ Reptiles	Benthic Invertebrates	Fish	Fish Egg	Bird Egg	Fish	Birds	Mammals
Metals											
Antimony	X	X									
Arsenic	X	X			X	X					X
Cadmium	X	X	X		X	X			X	X	X
Chromium	X	X	X	X	X	X			X	X	
Cobalt	X	X			X				X		
Copper	X	X	X	X	X	X			X	X	X
Lead	X	X	X	X	X	X				X	X
Mercury	X	X	X	X	X	X	X	X	X	X	X
Methylmercury[a]	X				X	X	X	X	X	X	X
Nickel	X	X		X	X				X	X	X
Selenium	X	X	X		X	X			X	X	X
Silver	X		X	X	X	X					
Vanadium	X	X			X				X	X	X
Zinc	X	X	X	X	X	X			X	X	X
Butyltins											
TBT	X		X						X		
PAHs											
1-Methylnaphthalene	X										
1-Methylphenanthrene	X										
2,6-Dimethylnaphthalene	X										
2-Methylnaphthalene	X										
Acenaphthene	X	X									
Acenaphthylene	X										

Table 5-2. Summary of COPECs

COPEC	Sediment		Surface Water		Tissue				Diet		
	Benthic Invertebrates	Aquatic Plants	Invertebrates, Fish, Aquatic Plants, and Zooplankton	Amphibians/ Reptiles	Benthic Invertebrates	Fish	Fish Egg	Bird Egg	Fish	Birds	Mammals
Anthracene	X		X								
Benzo(a)anthracene	X		X								
Benzo(a)pyrene	X		X						X		
Benzo(b/j)fluoranthene	X										
Benzo(e)pyrene	X										
Benzo(g,h,i)perylene	X										
Benzo(k)fluoranthene	X										
Chrysene	X										
Dibenzo(a,h)anthracene	X										
Fluoranthene	X		X								
Fluorene	X										
Indeno(1,2,3-cd)pyrene	X										
Naphthalene	X										
Perylene	X										
Phenanthrene	X										
Pyrene	X		X								
Total benzofluoranthenes	X										
Total HPAHs	X				X	X				X	X
Total LPAHs	X				X	X				X	
Total PAHs	X								X		
SVOCs											
2,4-Dinitrotoluene	X										
2,6-Dinitrotoluene	X										
4-Methylphenol	X										

Table 5-2. Summary of COPECs

COPEC	Sediment		Surface Water		Tissue				Diet		
	Benthic Invertebrates	Aquatic Plants	Invertebrates, Fish, Aquatic Plants, and Zooplankton	Amphibians/ Reptiles	Benthic Invertebrates	Fish	Fish Egg	Bird Egg	Fish	Birds	Mammals
BEHP	X		X								
Butylbenzylphthalate	X		X								
Dibenzofuran	X										
Diethylphthalate	X										
Dimethylphthalate	X										
Di-n-butylphthalate	X										
Di-n-octylphthalate	X										
Isophorone	X										
n-Nitrosodiphenylamine	X										
Pentachlorophenol	X										
Phenol	X										
VOCs											
1,2,3-Trichlorobenzene	X										
1,2,4-Trichlorobenzene	X										
1,4-Dichlorobenzene	X										
1,4-Dioxane	X										
Acetone	X										
m, p-Xylene	X										
Toluene	X										
Trichloroethene				X							
PCBs											
Aroclor 1254	X										
Aroclor 1260	X										
Total PCBs	X		X		X	X	X	X	X	X	X

Table 5-2. Summary of COPECs

COPEC	Sediment		Surface Water		Tissue				Diet		
	Benthic Invertebrates	Aquatic Plants	Invertebrates, Fish, Aquatic Plants, and Zooplankton	Amphibians/ Reptiles	Benthic Invertebrates	Fish	Fish Egg	Bird Egg	Fish	Birds	Mammals
PCB TEQ[b]	X		X		X	X		X	X	X	X
PCDDs/PCDFs											
2,3,7,8-TCDD	X		X		X	X					
PCDD/PCDF TEQ[b]	X		X		X	X	X	X	X	X	X
Total TEQ[b]	X		X		X	X	X	X	X	X	X
Organochlorine Pesticides											
4,4'-DDD	X										
4,4'-DDE	X		X								
4,4'-DDT	X		X								
Aldrin	X										
alpha-BHC	X										
alpha-chlordane	X										
beta-BHC	X										
gamma-BHC (Lindane)	X										
Dieldrin	X		X		X	X		X			X
Endrin	X										
Endosulfan I	X					X					
Endosulfan II	X										
gamma-chlordane	X										
Heptachlor	X										
Heptachlor epoxide	X				X						
Hexachlorobenzene	X		X								
Methoxychlor	X										
Total chlordane	X		X								

Table 5-2. Summary of COPECs

COPEC	Sediment		Surface Water		Tissue				Diet		
	Benthic Invertebrates	Aquatic Plants	Invertebrates, Fish, Aquatic Plants, and Zooplankton	Amphibians/ Reptiles	Benthic Invertebrates	Fish	Fish Egg	Bird Egg	Fish	Birds	Mammals
Total DDx	X		X		X	X		X	X	X	
Herbicides											
2,4,5-TP (Silvex)	X										
Other											
Cyanide	X		X								

[a] Total mercury is included as well as methylmercury for tissue and diet COPECs because some of the tissue and dietary TSVs were based on total mercury in tissue. Typically, more than 50% of total mercury in lower-trophic-level fish and invertebrate tissue is in the form of methylmercury. Methylmercury made up 87% of the mercury in LPRSA fish collected in 2009, 84% in blue crab collected in 2009, and 76% in mummichog collected in 2010, but only 14% in bioaccumulation worms.

[b] TEQs are based on fish TEFs for sediment exposure for benthic invertebrates, and for surface water exposure for invertebrates, fish, aquatic plants, and zooplankton. TEQs for fish, birds, and mammals are based on TEFs for their respective receptor groups.

BEHP – bis(2-ethylhexyl) phthalate
BHC – benzene hexachloride
COPEC – chemical of potential ecological concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
LPAH – low-molecular-weight polycyclic aromatic hydrocarbon
LPRSA – Lower Passaic River Study Area
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
PCDD – polychlorinated dibenzo-*p*-dioxin
PCDF – polychlorinated dibenzofuran
SVOC – semivolatile organic compound
TBT – tributyltin
TCDD – tetrachlorodibenzo-*p*-dioxin
TEF – toxic equivalency factor
TEQ – toxic equivalent
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT and 4,4′-DDT)
TSV – toxicity screening value
VOC – volatile organic compound

5.5.2 COIs with no TSVs

In addition to the COPECs identified, COIs with no TSVs were retained for discussion in this BERA. These COIs are presented in Table 5-3 and discussed in the following subsections.

Table 5-3. Summary of COIs with no TSVs

COI	Sediment	Surface Water	Tissue		Diet		
	Benthic Invertebrates	Invertebrates, Fish, Aquatic Plants, and Zooplankton	Benthic Invertebrates	Fish	Fish	Birds	Mammals
Metals							
Antimony			X	X	X	X	
Beryllium	X		X	X	X	X	X
Cobalt				X			
Nickel				X			
Silver						X	X
Thallium	X			X	X		
Vanadium				X			
Butyltins							
Monobutyltin	X	X	X	X	X		
Dibutyltin	X	X	X		X		
Tetrabutyltin	X		X	X	X		
PAHs							
1-Methylnaphthalene			X	X	X	X	X
1-Methylphenanthrene		X	X	X	X	X	X
2,3,5-Trimethylnaphthalene	X	X	X	X	X	X	X
2,6-Dimethylnaphthalene		X	X	X	X	X	X
2-Methylnaphthalene			X	X	X	X	X
Benzo(e)pyrene		X	X	X	X	X	X
Benzo(k)fluoranthene		X					
Chrysene		X					
Dibenzo(a,h)anthracene		X					
Dibenzothiophene	X	X	X	X	X	X	X
Perylene		X	X	X	X	X	X
SVOCs							
1,1′-Biphenyl	X						

COI	Sediment	Surface Water	Tissue		Diet		
	Benthic Invertebrates	Invertebrates, Fish, Aquatic Plants, and Zooplankton	Benthic Invertebrates	Fish	Fish	Birds	Mammals
2,6-Dinitrotoluene		X					
4-Chloroaniline		X					
4-Methylphenol		X	X		X	X	
Acetophenone	X	X	X		X	X	
Benzaldehyde	X	X	X	X	X	X	
Bis(2-chloroethoxy)methane		X					
BEHP					X		
Caprolactam	X	X					
Carbazole	X	X					
n-Nitroso-di-n-propylamine	X	X					
VOCs							
1,4-Dioxane,		X					
4-Methyl-2-pentanone	X						
Bromodichloromethane		X					
Chloromethane	X	X					
cis-1,2-Dichloroethylene	X	X					
Isopropylbenzene		X					
Methylcyclohexane	X						
Methyl acetate	X						
PCDDs/PCDFs							
Individual PCDDs/PCDFs other than 2,3,7,8-TCDD	X	X					
Organochlorine Pesticides							
Aldrin				X	X		
alpha-BHC			X	X	X	X	X
beta-BHC			X	X	X	X	
delta-BHC			X	X	X	X	X
gamma-BHC (Lindane)					X		
alpha-Chlordane					X		
gamma-Chlordane					X		
Endosulfan sulfate			X	X	X	X	X
Endrin aldehyde			X	X	X	X	X
Endrin ketone	X	X		X	X	X	X

COI	Sediment	Surface Water	Tissue		Diet		
	Benthic Invertebrates	Invertebrates, Fish, Aquatic Plants, and Zooplankton	Benthic Invertebrates	Fish	Fish	Birds	Mammals
Heptachlor					X		
Heptachlor epoxide					X	X	X
Hexachlorobenzene					X		
Methoxychlor					X		
cis-Nonachlor					X		
trans-nonachlor					X		
Octachlorostyrene			X				
Oxychlordane					X		
Total Chlordane					X		
Total endosulfan					X		
Herbicides							
2,4,5-T	X						
2,4-DB							
TPHs							
TPH – alkanes	X						
TPH – purgeable	X						
TPH – extractable	X						

BEHP – bis(2-ethylhexyl) phthalate
BHC – benzene hexachloride
COI – chemical of interest
PAH – polycyclic aromatic hydrocarbon
PCDD – polychlorinated dibenzo-*p*-dioxin
PCDF – polychlorinated dibenzofuran
SVOC – semivolatile organic compound
TCDD – tetrachlorodibenzo-*p*-dioxin
TEF – toxic equivalency factor
TPH – total petroleum hydrocarbon
TSV – toxicity screening value
VOC – volatile organic compound

5.5.2.1 Sediment

A number of sediment COIs could not be evaluated in the SLERA because no freshwater or estuarine TSVs were available. A total of 40 COIs, including 16 individual PCDDs/PCDFs other than 2,3,7,8-TCDD, had no sediment TSVs (Table 5-3). The potential risk to benthic invertebrates from these COIs is evaluated using the SQT approach in this BERA, namely by conducting site-specific toxicity tests and community surveys and evaluating similar chemicals.

5.5.2.2 Surface water

A number of surface water COIs could not be evaluated in the SLERA because no freshwater or estuarine TSVs were available. A total of 42 COIs for aquatic organisms (i.e., benthic invertebrates, fish, zooplankton, and aquatic plants), including 16 individual PCDDs/PCDFs other than 2,3,7,8-TCDD, had no surface water TSVs; these COIs are presented in Table 5-3 and are discussed below:

- **Butyltins –** Tributyltin (TBT) is the most toxicologically significant of the butyltins and was identified as a COPEC; potential risks from other butyltins are assumed to be less than risks from TBT.
- **PAHs –** Of the nine COIs with no TSVs, three (benzo[k]fluoranthene, chrysene, and dibenzo[a,h]anthracene) are USEPA-classified priority pollutants. These PAHs were frequently detected (in 84 to 99.5% of LPRSA samples). TSVs were available for 15 other individual PAHs, 5 of which were identified as COPECs.
- **SVOCs** – Of the nine SVOCs with no TSVs, seven were detected in < 5% of samples, while benzaldehyde and carbazole were detected in 14 and 8% of samples, respectively. Based on the low detection frequency, it is unlikely that these SVOCs in surface water pose an unacceptable risk to aquatic organisms in the LPRSA.
- **PCDDs/PCDFs –** Many individual PCDD/PCDF congeners were frequently detected; however, surface water thresholds were unavailable for individual dioxins and furans other than 2,3,7,8-TCDD. 2,3,7,8-TCDD is the most toxicologically significant PCDD/PCDF and was evaluated as a COPEC.
- **Pesticides –** Endrin ketone was a COI detected in 66% of LPRSA surface water samples. However, endrin was not identified as a COPEC in surface water based on the endrin TSV.
- **VOCs** – Of these five COIs, four were infrequently detected: 1-4-dioxane and isopropylbenzene in < 10% of samples, bromodichloromethane in 14% of samples, and chloromethane in 24% of samples. It is unlikely that concentrations of these VOCs in LPRSA surface water would pose an unacceptable risk to aquatic invertebrates because of the low detection frequency. The remaining VOC, cis-1,2-dichloroethylene, was detected in 98% of samples. It is unlikely that these volatiles will pose unacceptable risk since they volatilize quickly.

5.5.2.3 Tissue

A number of tissue COIs could not be evaluated in the SLERA because no invertebrate or fish tissue TSVs were available. A total of 22 COIs for benthic invertebrates and 23 COIs for fish had no tissue TSVs and are discussed below:

- **Butyltin –** Of the butyltins, TBT is the most toxic to aquatic organisms because of its chemical properties as a triorganotin (USEPA 2003b). Although frequently detected in LPRSA aquatic organisms, risks from dibutyltin, monobutyltin, or tetrabutyltin (degradation products of TBT) are not expected to be any greater than those from TBT, which was not identified as a COPEC for invertebrates or fish tissue.

- **Pesticides**
  - The compounds alpha-, beta-, and delta-benzene hexachloride (BHC) are less toxic than gamma-BHC (ATSDR 2005b), indicating that the TSV for gamma-BHC can be used as a surrogate TSV for other BHC compounds. Alpha-, beta-, and delta-BHC were detected at concentrations below the gamma-BHC TSV for fish; these COIs were detected in 30% or less of all benthic invertebrate tissue samples, and in all but one sample, concentrations of these isomers were below those of gamma-BHC, which was detected in 10% of tissue samples. Thus, unacceptable risk to invertebrate or fish species from these compounds is unlikely.
  - Endosulfan sulfate is a breakdown product of the pesticide endosulfan; there is little difference in the toxicity of the two compounds (ATSDR 2013), so the TSV for endosulfan can be used as a surrogate for that of endosulfan sulfate. All detected concentrations of endosulfan sulfate were below the endosulfan TSVs, so unacceptable risk is unlikely.
  - There is very little information on the toxicity of aldrin, endrin aldehyde, and endrin ketone. Aldrin was detected in 52% of all fish tissue samples. Due to the lack of toxicity data, the risks to fish associated with aldrin are not known. Endrin aldehyde and endrin ketone were detected in 7 and 5% of all fish tissue samples, respectively. Endrin aldehyde was detected in only 8% of benthic invertebrate tissue samples. Risks to aquatic organisms from endrin aldehyde and endrin ketone are not likely given the low detection frequency of these COIs.
- **Metals –** Because of the variety of species-specific strategies used by fish and invertebrates to store, detoxify, and excrete bioaccumulated metals, and because fish tissue burdens tend to be time- and exposure route-dependent, metals tissue residues are poorly predictive of adverse effects (Adams et al. 2011; USEPA 2007e). Thus, lack of TSVs for metals does not substantially affect overall risk estimates for fish or invertebrates. For invertebrates, the potential risks from metal COIs with no TSVs (antimony and beryllium) is unknown.
- **PAHs –** Although tissue TSVs were not available for a number of individual PAHs, TSVs were available for high-molecular-weight polycyclic aromatic hydrocarbons (HPAHs) and low-molecular-weight polycyclic aromatic hydrocarbons (LPAHs). PAH sums were evaluated as COPECs for benthic invertebrates. Because fish rapidly metabolize PAHs, PAH tissue residues are poorly predictive of adverse effects. Thus, lack of TSVs for PAHs does not substantially affect overall risk estimates for fish.
- **SVOCs –** SVOC COIs with no TSVs were infrequently detected in tissue. The three SVOCs with no benthic invertebrate TSVs were detected in approximately 2 to 11% of all benthic invertebrate tissue samples, and the one SVOC with no

fish TSV (benzaldehyde) was detected in only 1% of all fish tissue samples. Unacceptable risks to aquatic organisms from these COIs are not expected, given their low detection frequency in tissue.

5.5.2.4 Diet

A number of dietary COIs could not be evaluated in the SLERA because no dietary TSVs were available; these are discussed in the following subsections.

Fish

A total of 37 COIs had no fish diet TSVs:

- **Metals –** Metals COIs with no diet fish TSVs were detected in prey tissue. There is very little information on the fish dietary pathway for these metals.
- **PAHs –** Fish diet TSVs were available for benzo(a)pyrene, which is generally the most toxic PAH, and for total PAHs; both were identified as COPECs. Thus, risks to fish from PAH exposure is likely accounted for by the available TSVs.
- **SVOCs and pesticides –** There is very little information on the dietary toxicity to fish of four SVOCs and numerous pesticides. Therefore, risks to fish resulting from dietary exposure to these COIs are unknown.

Wildlife

A total of 21 COIs had no bird diet TSVs, and a total of 15 COIs had no mammal diet TSVs:

- **Metals** – Metal COIs with no bird or mammal diet TSVs were detected in wildlife prey tissue. Very little information is available on the toxicity of these metals to wildlife from exposure via ingestion.
- **PAHs –** Bird and mammal screening-level TSVs were available for HPAHs and LPAHs; these TSVs are based on the toxicity of PAH mixtures that include most PAHs, including benzo(a)pyrene, which is generally the most toxic PAH. Thus, risks to wildlife from PAH exposure is likely accounted for by the available TSVs.
- **SVOCs** – There is very little information on the avian toxicity of the three SVOC COIs with no TSVs. These SVOCs were detected very infrequently in fish and crab tissue (detection frequency range of 0 to 4% of samples), indicating that unacceptable risks are unlikely for belted kingfisher and great blue heron, which prey on fish and crabs. The detection frequencies of 4-methylphenol, acetophenone, and benzadahyde in worm tissue were 29, 43, and 80%, respectively; risk to spotted sandpiper from these COIs is not known because of a lack of toxicity data for these compounds.

- **Pesticides**
  - The maximum doses of alpha-BHC, beta-BHC, delta-BHC, endosulfan sulfate, and heptachlor epoxide do not exceed their surrogate TSVs, indicating a low likelihood of posing an unacceptable risk to wildlife. The compounds alpha-BHC, beta-BHC, and delta-BHC are less toxic than gamma-BHC (ATSDR 2005b), indicating that the TSV for gamma-BHC can be used as a conservative surrogate TSV for other BHC compounds. Heptachlor breaks down rapidly (i.e., within hours) into heptachlor epoxide in the environment, so the toxicity of these two compounds is generally considered similar (ATSDR 2005a), and the TSV for heptachlor can be used as a surrogate for that of heptachlor epoxide. Similarly, endosulfan sulfate is a breakdown product of the pesticide endosulfan, and there is little difference in the toxicity of the two compounds (ATSDR 2013), so the TSV for endosulfan can be used as a surrogate TSV for endosulfan sulfate.
  - There is very little information on the toxicity of endrin aldehyde and endrin ketone; however, these compounds had low detection frequencies in prey tissue (0 to 11% in fish, crab, and worm tissue), indicating that pesticides without TSVs are unlikely to pose unacceptable risks to wildlife.

5.5.3 Analytes identified in the SLERA with DL exceedances of TSVs

In addition to COIs with no TSVs, the SLERA identified COIs and analytes for which DLs exceeded TSVs. These COIs and analytes are discussed in the following subsections.

5.5.3.1 Sediment

A total of 12 sediment COIs had maximum detected concentrations less than TSVs, but DLs greater than TSVs:

- **Herbicides** – 2,4-D
- **VOCs** – 1,2-dichlorobenzene, 1,3-dichlorobenzene, benzene, carbon disulfide, chloroform, cyclohexane, ethylbenzene, isopropylbenzene, methylene chloride, o-xylene, and trichloroethene

The potential toxicity of VOCs to benthic invertebrates is considered low given that these chemicals are volatile. The toxicity of 2,4-D to benthic invertebrates in sediment is unknown.

In addition, a total of 15 sediment analytes that were never detected in sediment had DLs greater than TSVs:

- **VOCs** – 2,4,5-trichlorophenol, 2,4,6-trichlorophenol, 2,4-dichlorophenol, 2,4-dimethylphenol, 2,4-dinitrophenol, 2-chloronaphthalene, 2-chlorophenol, 3,3'-dichlorobenzidine, 4-nitrophenol, bis(2-chloroethyl) ether, hexachlorobutadiene, hexachlorocyclopentadiene, hexachloroethane, and nitrobenzene

- **Pesticides** – toxaphene

The potential toxicity of VOCs to benthic invertebrates is considered low given that these chemicals are volatile. The potential toxicity of toxaphene to benthic invertebrates is unknown, since this chemical was never detected in LPRSA sediment based on the reported DLs.

5.5.3.2 Surface water

No surface water COIs had maximum detected concentrations less than TSVs and DLs greater than TSVs. A total of three sediment analytes that were never detected in surface water had DLs greater than TSVs:

- **SVOCs –** hexachlorobutadiene, pentachlorophenol, and trans-1,3-dichloropropene

The potential toxicity of these COIs to aquatic organisms is unlikely given that these chemicals were never detected in LPRSA surface water based on the reported DLs.

5.5.3.3 Tissue

One fish tissue COI had maximum detected concentrations less than TSVs and DLs greater than TSVs:

- **Pesticides –** endrin

DLs for three fish species (i.e., common carp, white catfish, and American eel) were greater than the endrin TSV. HQs were greater than 1.0 based on a comparison of DLs with a no-observed-adverse-effect level (NOAEL)-based TSV for the following: 4 out of 12 common carp samples (maximum HQ of 3.5), 1 out of 19 white catfish samples (maximum HQ of 1.7), and 5 out of 21 American eel samples (maximum HQ of 2.6) (see Appendix A). In addition, endrin was not identified as a COPEC for fish based on any of the detected concentrations. Therefore, endrin is not likely to pose an unacceptable risk to fish.

In addition, a total of 14 tissue analytes for invertebrates and/or fish that were never detected had DLs greater than TSVs:

- **SVOCs –** 2,4-dimethylphenol, 2,4-dinitrophenol, 4-methylphenol, 4-nitrophenol, atrazine, bis(2-ethylhexyl) phthalate (BEHP), bis-(2-chloroethyl)ether, butyl benzyl phthalate (BBP), diethylphthalate, dimethylphthalate, di-n-butylphthalate, hexachlorobutadiene, isophorone, and n-nitrosodiphenylamine

The potential toxicity of these COIs to aquatic organisms is unlikely given that these chemicals were never detected in LPRSA tissue based on the reported DLs.

6 Benthic Invertebrate Assessment

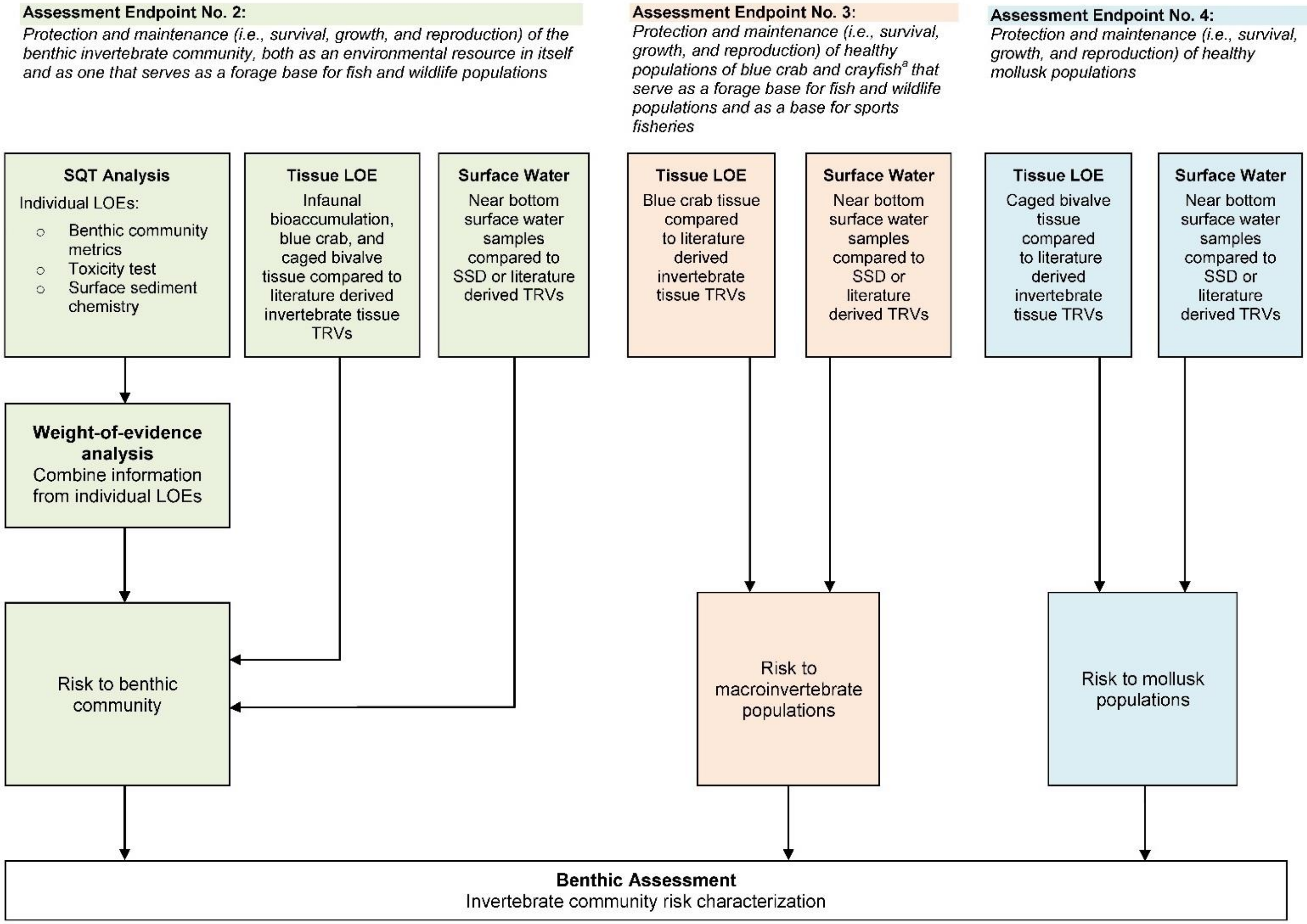
Benthic invertebrates represent a highly diverse group of taxa that play a key role in estuarine and riverine food webs (Thorp and Covich 2010b). Benthic invertebrates are an integral member of a fully functioning aquatic system and have a marked influence on ecosystems because they sort, rework, and oxygenate sediment (Bolam et al. 2002), and alter biogeochemical fluxes (e.g., nutrient cycling through the processing of detritus) (Covich et al. 1999). In the LPRSA, the benthic invertebrate community functions as a valuable environmental resource that provides important ecological services and serves as a forage base for fish and wildlife. Benthic invertebrate community structure (typically described using summary metrics or indices) and sediment toxicity tests conducted using benthic invertebrate species are often relied upon to assess sediment quality, because the species evaluated are intimately associated with sediment and are relatively immobile (Iannuzzi et al. 2008; Long and Chapman 1985).

The benthic assessment for the LPRSA focused on three of the assessment endpoints presented in Table 3-2 that address the protection and maintenance of the benthic community:

- **Assessment Endpoint No. 2 --** Protection and maintenance (i.e., survival, growth, and reproduction) of the benthic invertebrate community both as an environmental resource in itself and as one that serves as a forage base for fish and wildlife populations
- **Assessment Endpoint No. 3 –** Protection and maintenance (i.e., survival, growth, and reproduction) of healthy populations of blue crab and crayfish[68] that serve as a forage base for fish and wildlife populations and as a base for sports fisheries
- **Assessment Endpoint No. 4 –** Protection and maintenance (i.e., survival, growth, and reproduction) of healthy mollusk populations

Figure 6-1 presents a flow chart that shows the LOEs used to measure the risk to the benthic community for the three assessment endpoints.

[68] Crayfish were identified in the PFD as representing freshwater macroinvertebrate populations. However, per agreement between CPG and USEPA (Windward 2010g), blue crab will be the only species used to represent the macroinvertebrate population, for both the estuarine and freshwater portions of the LPRSA, because of the limited number of crayfish collected in the freshwater portion.



[a] Crayfish were identified in the PFD as representing freshwater macroinvertebrate populations. However, per agreement between CPG and USEPA (Windward 2010) and as discussed in the RARC Plan (Windward and AECOM [in prep]), blue crab will be the only species used to represent the macroinvertebrate population, for both the estuarine and freshwater portions of the LPRSA, because of the limited number of crayfish collected in the freshwater portion.

Figure 6-1. Benthic community risk characterization flowchart

The protection and maintenance of the benthic invertebrate community (Assessment Endpoint No. 2) was assessed by combining three LOEs (i.e., benthic invertebrate community metrics, sediment toxicity tests, and surface sediment chemistry data), referred to collectively as an SQT assessment. The SQT approach is well established in the literature (Bay and Weisberg 2012; Chapman 1990, 2000; Long and Wilson 1997) as a method for assessing ecological risks associated with contaminated sediment, and this approach has been previously applied to the lower portion of the LPRSA (Iannuzzi et al. 2008). The SQT approach to assessing risk to the benthic invertebrate community is consistent with NJDEP ecological evaluation guidance (NJDEP 2012a). USEPA and CPG agreed to use the SQT approach in the LPRSA, as documented in the USEPA-approved PFD (Windward and AECOM 2009) and the USEPA-approved benthic QAPP (Windward 2009b). Consistent with USEPA guidance (USEPA 1997a, 1998, 2002d, 2005a), the SQT approach uses site-specific data, which include empirically derived benthic response data (i.e., sediment toxicity and benthic invertebrate community data) to assess risks to the benthic community compared with reference area conditions (Bay and Weisberg 2012; Chapman 1990, 2000; Long and Wilson 1997). USEPA identified Jamaica Bay as the estuarine reference area representing an urban habitat; SQT data for this reference area was collected and analyzed by others. Similarly, USEPA identified the area upstream of Dundee Dam (Windward 2012a) as a freshwater reference area representing urban habitat; the reference dataset for this area was collected by CPG in fall 2012. Mullica River and Great Bay were also identified by USEPA as non-urban reference areas. However, acceptable SQT data were available from only estuarine portions of Mullica River and Great Bay; these data was collected and analyzed by others.

For the purpose of this BERA, the three SQT LOEs were evaluated individually and then combined using a WOE framework. The three individual SQTs LOEs are described in detail in Appendix P. The weighting of the three LOEs was initially developed by CPG, but was then modified by USEPA Region 2 (USEPA 2015d). The analyses of LOEs and the WOE are based on the weights assigned to LOEs and comparisons to urban and non-urban reference conditions, which form the basis for risk characterizations. The WOE analysis is provided and discussed in Section 6.1 (and presented in Appendix B, Tables B8 and B9). In addition, a quantitative analysis conducted to assess uncertainty is presented in the uncertainty section (Section 6.1.3). The quantitative analysis utilizes scientifically relevant approaches to each of the three SQT LOEs; these approaches were conducted as part of uncertainty analyses in Appendix P (Sections 2.3.4, 3.2.5, 4.1.4, and 4.3), and were intended to address key points of uncertainty associated with the SQT approach. By addressing these uncertainties, the quantitative analysis provides a reasonable bounding prediction of ecological risks to benthic invertebrates in the LPRSA.

In Appendix P, it is demonstrated that LPRSA benthic invertebrate community structure and sediment toxicity data are negatively associated with a mixture of chemicals and/or habitat variables in LPRSA sediment. Through multivariate analysis, it was determined that 10 of 11 measurement endpoints were negatively associated with sediment chemical concentrations, and that habitat conditions had negative associations with several measurement endpoints. For example, benthic invertebrate diversity, survival, and growth were negatively associated with a mixture of chemicals, including metals and pesticides (e.g., total DDx), along with the percent of total fine-grained sediment (a habitat variable). Some measurement endpoints were related to only chemistry (e.g., Shannon-Wiener H' and SDI) or only habitat (abundance). A number of the benthic measurement endpoints had chemical factors that were more important than habitat variables for predicting effects. The analysis showed that Factor scores, including Factor 2 (representing a mixture of metals, total DDx, and hexachlorobenzene), were negatively associated with survival and biomass in toxicity tests and benthic diversity in the field. Mixtures of chemicals were found to co-vary spatially (indicated by factor analysis), indicating that multiple COPECs, either singly or as a mixture, were likely responsible for benthic impairment in the LPRSA. Because many COPECs were correlated with a small number of factors, it was not possible to identify any single chemical driver of benthic invertebrate risk from the multivariate analysis. Additional LOEs related to fate, toxicity, and bioavailability of specific contaminants would be needed to reduce the uncertainty associated with identifying individual COPECs as risk drivers for benthic invertebrates. It is likely that the observed benthic invertebrate impacts were the result of exposure to multiple LPRSA-related COPECs, and these impacts were likely exacerbated by habitat conditions. Based on this conclusion, the sediment chemistry LOE is included in the overall WOE evaluation in the BERA risk characterization (Section 6.1.2).

Two additional LOEs were evaluated for the protection and maintenance of benthic invertebrates: the surface water LOE and the tissue LOE. In the surface water LOE, chemical concentrations in near-bottom (i.e., 3 ft above bottom) surface water samples were compared with surface water toxicity reference values (TRVs) expected to be protective of benthic invertebrates. In the tissue LOE, chemical concentrations in marine and freshwater worm tissue used in laboratory bioaccumulation testing, blue crab tissue collected from the LPRSA, and LPRSA-deployed *in situ* caged bivalve tissue were compared with tissue TRVs. These TRVs were considered relevant for assessing benthic invertebrate risk from surface water or bioaccumulation in tissues using a HQ approach.

An outline of the benthic assessment process is presented in Table 6-1. The three LOEs in the SQT analysis (i.e., benthic invertebrate community structure, sediment toxicity data, and sediment chemistry data) are presented in Appendix P. The SQT WOE analysis is discussed in Section 6.1. Uncertainties associated with various components of these assessments are summarized in Section 6.1.3. Surface water and tissue LOEs are presented in Sections 6.2 and 6.3, respectively. Overall conclusions for the benthic assessment for the LPRSA are presented in Section 6.4.

Table 6-1. Summary of the benthic risk assessment process and location in BERA

Step in Benthic Assessment	Location in BERA	Description
Benthic invertebrate community LOE	Appendix P, Section 2	one of three LOEs for the SQT analysis; presents methods and results of the benthic invertebrate community survey and a comparison of benthic invertebrate community data to reference area data; relevant data are also provided in Appendix B, Tables B3, B4, and B6; raw datasets are provided in Appendices K (Table K4) and L (Tables L6 and L7)
Sediment toxicity testing LOE	Appendix P, Section 3	one of three LOEs for the SQT analysis; presents methods and results of sediment toxicity tests and a comparison of sediment toxicity test data to negative control and reference area data; relevant data are also provided in Appendix B, Tables B3, B4, B5, and B6; raw datasets are provided in Appendices K (Table K5) and L (Table L8)
Sediment chemistry LOE	Appendix P, Section 4	one of three LOEs for the SQT analysis; presents methods and results of the comparison of LPRSA data to mean ERM and mean PEC quotients and T20 and T50 values; includes the methods and results of correlation and multivariate analyses comparing sediment chemistry with benthic community metric and sediment toxicity test data; includes uncertainty analysis of sediment chemistry LOE; relevant data are also provided in Appendix B, Tables B1, B2, and B7; raw datasets are provided in Appendices K (Table K1) and L (Tables L1 and L4)
SQT WOE assessment	Section 6.1	WOE assessment from the SQT analysis, as well as uncertainties and risk conclusions from the WOE analysis; relevant data are also provided in Appendix B, Tables B8, B9, and B10 (which are based on data in Tables B3, B4, and B7)
Surface water assessment	Section 6.2	for each secondary LOE, presents COPECs identified in the SLERA, exposure and effects data, HQs, uncertainty discussion, and summary of risk characterization
Benthic invertebrate tissue assessment	Section 6.3	
Risk conclusions	Section 6.4	summary of overall risk conclusions and proposed COCs

BERA – baseline ecological risk assessment
COC – chemical of concern
COPEC – chemical of potential ecological concern
ERM – effects range-median
HQ – hazard quotient
LOE – line of evidence
LPRSA – Lower Passaic River Study Area
PEC – probable effects concentration
SLERA – screening-level ecological risk assessment
SQT – sediment quality triad
T20 – 20% probability of observing toxicity
T50 – 50% probability of observing toxicity
WOE – weight of evidence

6.1 Sediment Quality Triad Weight of Evidence Assessment

The purpose of this section is to assess the three LOEs associated with the SQT data from the LPRSA (described in Appendix P) and combine them within a WOE risk framework. The SQT approach for establishing ecological risks associated with degraded sediment quality is well established in the literature (Long and Chapman

1985; Chapman 1990; Canfield et al. 1994; Carr 1997; Iannuzzi et al. 2008; McPherson et al. 2008; Bay and Weisberg 2012), and it has been applied previously to the lower portion of the LPRSA (Iannuzzi et al. 2008). A WOE framework using SQT data includes conceptual, qualitative, and quantitative measures used to arrive at an ultimate conclusion of risk posed by sediment quality (Chapman et al. 2002). Best professional judgment can be used to develop a meaningful WOE framework (Bay and Weisberg 2012; McPherson et al. 2008; Bay et al. 2007). Different weights are often given to each LOE based on the level of uncertainty involved with the application of a specific LOE (Bay and Weisberg 2012; McPherson et al. 2008). Because the comparison of sediment chemistry concentrations to prescriptive sediment quality guidelines often results in poor predictive accuracy in estimating measurable effects (i.e., laboratory-based sediment toxicity data, benthic community metrics) (Appendix P, Section 4.1.4), sediment chemistry tends to be considered a weaker LOE within the SQT paradigm than the other SQT LOEs (Bay and Weisberg 2012; McPherson et al. 2008). Benthic community and sediment toxicity data LOEs are generally given equal or greater weight than sediment chemistry, though there is discussion in the literature as to whether sediment toxicity or benthic community data are more important for determining impacted sediment quality (Bay et al. 2007). For the purposes of assessing benthic community risks in the LPRSA, the approach to evaluating the three SQT LOEs assumes that all three LOEs have equal weight (i.e., 1.0), consistent with USEPA guidance for the LPRSA (USEPA 2015d). A multivariate analysis of LPRSA SQT data (Appendix P) indicates that sediment chemical factors (in addition tohabitat factors) are negatively associated with benthic invertebrate community metrics and sediment toxicity test results. Therefore, it is reasonable to give sediment chemistry equal weight in the WOE analysis.

6.1.1 Methods

6.1.1.1 Assignment of weights to the individual LOEs

Equal weights were assigned to each of the three SQT LOEs (benthic invertebrate community, sediment toxicity, and sediment chemistry). Each LOE was given a maximum possible weight of 1.0; the combined maximum possible weight of all three LOEs in the WOE analysis was 3.0. Within the benthic invertebrate community and sediment toxicity LOEs, several metrics or sediment toxicity test endpoints were evaluated.[69] Each of these metrics or toxicity test endpoints was given an equal possible weight (Tables 6-2 and 6-3), and the sum of all the metrics or all the toxicity

[69] The following benthic invertebrate community metric and sediment toxicity test endpoint variables were included in the WOE analysis: **benthic community LOE**—abundance per m^2, taxa richness, Shannon-Wiener H′, Pielou's J′, Swartz's Dominance Index, and Hilsenhoff Biotic Index (tidal freshwater locations only); **sediment toxicity LOE**—*Ampelisca abdita* survival (estuarine only), *Hyalella azteca* survival and biomass, and *Chironomus dilutus* survival and biomass (freshwater only). Descriptions of these variables and their associated LOEs are provided in Appendix P.

test endpoint weights equaled 1.0 (within each LOE). A greater overall WOE weight provides greater certainty of sediment risks to benthic invertebrates; similarly, lesser weights across all three LOEs indicate greater certainty of low or no risk to benthic invertebrates from sediments. Moderate weights (i.e., "medium impacts") across all LOEs indicate disagreement within or among LOEs, which increases uncertainty when characterizing risks. The uncertainty analysis of LPRSA locations with moderate weights is discussed in Section 6.1.2.3, including a detailed evaluation of moderate weights.

Table 6-2. Weights used for the benthic invertebrate community LOE

Response Variable	Endpoint Weight	
	Upper Estuarine/Fluvial Estuary Zone Locations	Tidal Freshwater Zone Locations
Abundance (per m^2)	0.20	0.167
Taxa richness	0.20	0.167
Shannon-Wiener H′	0.20	0.167
Pielou's J′	0.20	0.167
Swartz's dominance index	0.20	0.167
Hilsenhoff Biotic Index	na	0.167
Total weight value	**1.0**	**1.0**

Note: Hilsenhoff Biotic Index was developed to describe freshwater communities, and so was not calculated for estuarine communities in the LPRSA.

LOE – line of evidence

LPRSA – Lower Passaic River Study Area

na – not applicable

WOE – weight of evidence

Table 6-3. Weights used for the sediment toxicity LOE

Response Variable	Endpoint Weight	
	Estuarine Toxicity Locations[a]	Freshwater Toxicity Locations[b]
Ampelisca abdita survival	0.333	na[c]
Chironomus dilutus survival	na[d]	0.25
Chironomus dilutus biomass	na[d]	0.25
Hyalella azteca survival	0.333	0.25
Hyalella azteca biomass	0.333	0.25
Total weight	**1.0**	**1.0**

a Estuarine toxicity locations are defined as having ≥ 5 ppth salinity in interstitial water at the time of collection for toxicity testing.

b Freshwater toxicity locations are defined as having < 5 ppth salinity in interstitial water at the time of collection for toxicity testing.

c *A. abdita* tests were conducted only at estuarine locations; no data are available for freshwater locations, so a weight value is not applicable.

d *C. dilutus* tests were conducted only at freshwater locations; no data available are available for estuarine locations, so a weight value is not applicable.

LOE – line of evidence

na – not applicable

ppth – parts per thousand

The actual benthic invertebrate community and sediment toxicity weights applied in the WOE analysis are based on the results from the analysis of each LOE presented in Appendix B, Tables B3 and B4. Benthic invertebrate community and sediment toxicity LOE weights were assigned based on comparisons of LPRSA SQT data to reference area conditions (Appendix P, Sections 2.3 and 3.2.2). For example, if at a given LPRSA SQT location a benthic invertebrate community metric exceeded the reference envelope for that metric, then the weight for that metric (as presented in Tables 6-2 and 6-3) was applied; similarly, this occurred for all other relevant metrics and sediment toxicity test endpoints at that location. The sediment chemistry LOE weight was applied in the WOE analysis, although the weighting of that LOE differs from the weighting of the benthic invertebrate community and sediment toxicity LOEs, as described below. Details about specific weights assigned to the benthic invertebrate community metrics and sediment toxicity test endpoints are provided in Tables 6-2 and 6-3, respectively. All variables within each LOE were given equal weighting.

For the sediment chemistry LOE (Appendix B, Table B7-1), logistic regression model-based T20 (20% probability of observing toxicity) and T50 (50% probability of observing toxicity) sediment quality guideline values (Field et al. 2002) were used to assign weights to LPRSA SQT locations (USEPA 2015b, c; 2016g; and other communications with USEPA throughout 2015 and 2016). LPRSA locations with at least one chemical exceeding a T20 value were assigned a weight of 0.5, and locations with at least one chemical exceeding a T50 value were assigned a weight of 1.0. If no T20 or T50 values were exceeded, a weight of 0 was assigned. Table 6-4 describes the weighting of the sediment chemistry LOE.

Table 6-4. Weights used for the sediment chemistry LOE

Any Sediment Quality Guideline Exceeded[a]	Sediment Chemistry LOE Weight	
	Upper Estuarine/Fluvial Estuary Zone Locations	Tidal Freshwater Zone Locations
T20	0.5	0.5
T50	1.0	1.0

Note: T20 and T50 values were applied to both estuarine and freshwater sediments.

a Weights are applied if indicated sediment criterion is exceeded by LPRSA sediment concentrations. If neither applicable criteria were exceeded, then a weight of 0 was assigned for the sediment chemistry LOE.

LOE – line of evidence

LPRSA – Lower Passaic River Study Area

T20 – 20% probability of observing toxicity

T50 – 50% probability of observing toxicity

Although several other analyses related to the sediment chemistry LOE are also presented in Appendix P, Section 4, these analyses were not used to assign weights in the WOE analysis. For example, simultaneously extracted metals-acid volatile sulfide (SEM-AVS) and the sum of 34 PAHs in LPRSA SQT samples were evaluated according to USEPA methods (USEPA 2003f, 2005c) in the analysis of uncertainty (Appendix P, Sections 4.3.2 and 4.3.3). In each case, there were inconsistent relationships or non-relationships between exceedances of literature-based toxic thresholds for bioavailable metals or PAHs and LPRSA sediment toxicity test results.[70]

[70] A possible exception is *H. azteca* biomass measured in sediments from estuarine LPRSA toxicity test locations (i.e., locations with interstitial salinity ≥ 5 ppth), which appears to be somewhat related to SEM-AVS (Appendix P, Figure 4-3). A similar relationship was not apparent for biomass measured in freshwater LPRSA toxicity tests with the same species.

6.1.1.2 WOE classification system

The classification system used to characterize potential risks to the benthic invertebrate community from the WOE analysis is provided in Table 6-5. WOE weights (i.e., the sum weight of all three LOEs) for all LPRSA SQT locations were compared with the ranges shown in Table 6-5, and the associated risk characterization category was assigned to those locations (Appendix B, Tables B8 and B9).

Table 6-5. Classification system for assigning benthic invertebrate risk based on WOE

Risk Characterization Based on WOE Analysis Result	Range of Weights	
	Low	High
No impact	≥ 0.0	≤ 0.75
Low impact	> 0.75	≤ 1.5
Medium impact	> 1.5	≤ 2.25
High impact	> 2.25	≤ 3.0

Note: Risk characterizations based on WOE analysis results are determined by the given ranges of sum WOE weights. For example, if the sum weight of all three LOEs at an LPRSA SQT location was between 1.5 and 2.25, then the location would be classified as having medium impact. Refinements to the medium-impact category are described in Section 6.1.2.2.

LOE – line of evidence
LPRSA – Lower Passaic River Study Area
SQT – sediment quality triad
WOE – weight of evidence

LPRSA locations with no or low impacts are considered to be of less concern than locations with high impacts. Risks associated with LPRSA locations with medium impacts have a higher degree of uncertainty, as the LOEs either disagree (e.g., benthic community metric data or sediment toxicity data are outside the reference conditions, but not both) or agree for only a small number of response variables (i.e., LOE components). This may be the result of moderate chemical impacts and/or stressful habitat characteristics. Therefore, medium-impact LPRSA locations require more detailed analysis using available site-specific information to reduce uncertainty in the location designation, are discussed in greater detail in Section 6.1.2.2. LPRSA locations with no, low, or high impacts are associated with a greater level of certainty than medium-impact locations because most or all LOEs are in agreement for locations with no, low, or high impacts (i.e., most or all LOEs indicate that the potential for risk is low or the potential for risk is high). The detailed analysis presented in Appendix B, Table B10, and described in Section 6.1.2.2 provides a greater level of certainty in WOE conclusions, to the extent practicable.

6.1.2 Risk characterization

6.1.2.1 Results of WOE analysis

Summaries of the WOE analyses of benthic impacts (based on comparisons of LPRSA SQT data to reference datasets representing either urban or non-urban habitats) are provided in Tables 6-6 and 6-7. The full WOE analyses are presented in Appendix B, Tables B8 and B9. Urban reference area data were available from Jamaica Bay and the area above Dundee Dam, and non-urban reference area data were available from Mullica River/Great Bay; these datasets are described in more detail in Appendix P, Sections 2.3.1 and 3.2.2, and Appendix B (Tables B3-1 [Jamaica Bay and Mullica River/Great Bay] and B4-1 [area above Dundee Dam]).[71] Results of the analyses are divided according to benthic salinity zones. Further information regarding the individual LOEs and the WOE analyses is provided in Appendix P and in Appendix B, Tables B3 (benthic invertebrate community and sediment toxicity LOEs, urban comparison), B4 (benthic invertebrate community and sediment toxicity LOEs, non-urban comparison), and B7 (sediment chemistry LOE).

Table 6-6. Summary of initial WOE analysis results, urban comparison

Benthic Salinity Zone	N	No Impact		Low Impact		Medium Impact		High Impact	
		n	%	n	%	N	%	n	%
Upper estuarine (RM 0 to RM 4)	25	0	0	13	52	10	40	2	8
Fluvial estuarine (RM 4 to RM 13)	54	0	0	14	26	35	65	5	9
Tidal freshwater (RM 13 to RM 17.4)	18[a]	0	0	1	6	6	33	11	61
Site wide	97	0	0	28	29	51	53	18	19

Note: Reference data representing urban habitats are from Jamaica Bay and the area above Dundee Dam. Medium-impact results are characterized in greater detail in Section 6.1.3.

[a] No benthic invertebrate community data were available at LPRT16B, so no WOE result was obtained.

% – percentage of locations

n – number of locations for each WOE conclusion

N – number of locations in each benthic salinity zone

RM – river mile

WOE – weight of evidence

Table 6-7. Summary of initial WOE analysis results, non-urban comparison

Benthic Salinity Zone	N	No Impact		Low Impact		Medium Impact		High Impact	
		n	%	n	%	n	%	n	%
Upper estuary (RM 0 to RM 4)	25	0	0	12	48	10	40	3	12
Fluvial estuary (RM 4 to RM 13)	54	0	0	8	15	37	69	9	17
Both estuarine zones (RM 0 to RM 13)	79	0	0	20	25	47	59	12	15

[71] As described in Appendix P, Sections 2.3.1 and 3.2.2, the full reference area datasets were subjected to screening steps to eliminate potentially contaminated and toxic samples.

Note: Reference data representing non-urban habitats are from Mullica River/Great Bay. Medium-impact results are characterized in greater detail in Section 6.1.2.2. Non-urban freshwater reference data were not available for comparison to LPRSA tidal freshwater data.

% – percentage of locations	N – number of locations in each benthic salinity zone
LPRSA – Lower Passaic River Study Area	RM – river mile
n – number of locations for each WOE conclusion	WOE – weight of evidence

Based on the summary presented in Table 6-6, the majority of locations site wide (51 of 97 locations, or 53%) received a WOE score that indicated a medium impact on the LPRSA benthic community compared to urban reference conditions. Throughout the LPRSA, no SQT locations had no impact, but 28 of 97 locations (29%) had low impacts. This leaves 18 LPRSA locations (19%) classified as high impact. These results suggest that benthic invertebrate communities in much of the LPRSA are impacted to some measurable degree, but that uncertainty remains (for medium-impact locations).

Table 6-7 indicates that LPRSA SQT locations (of the upper and fluvial estuarine zones) are marginally more impacted when compared to non-urban conditions than when compared to urban conditions.[72]

6.1.2.2 Post-hoc characterization of medium-impact locations

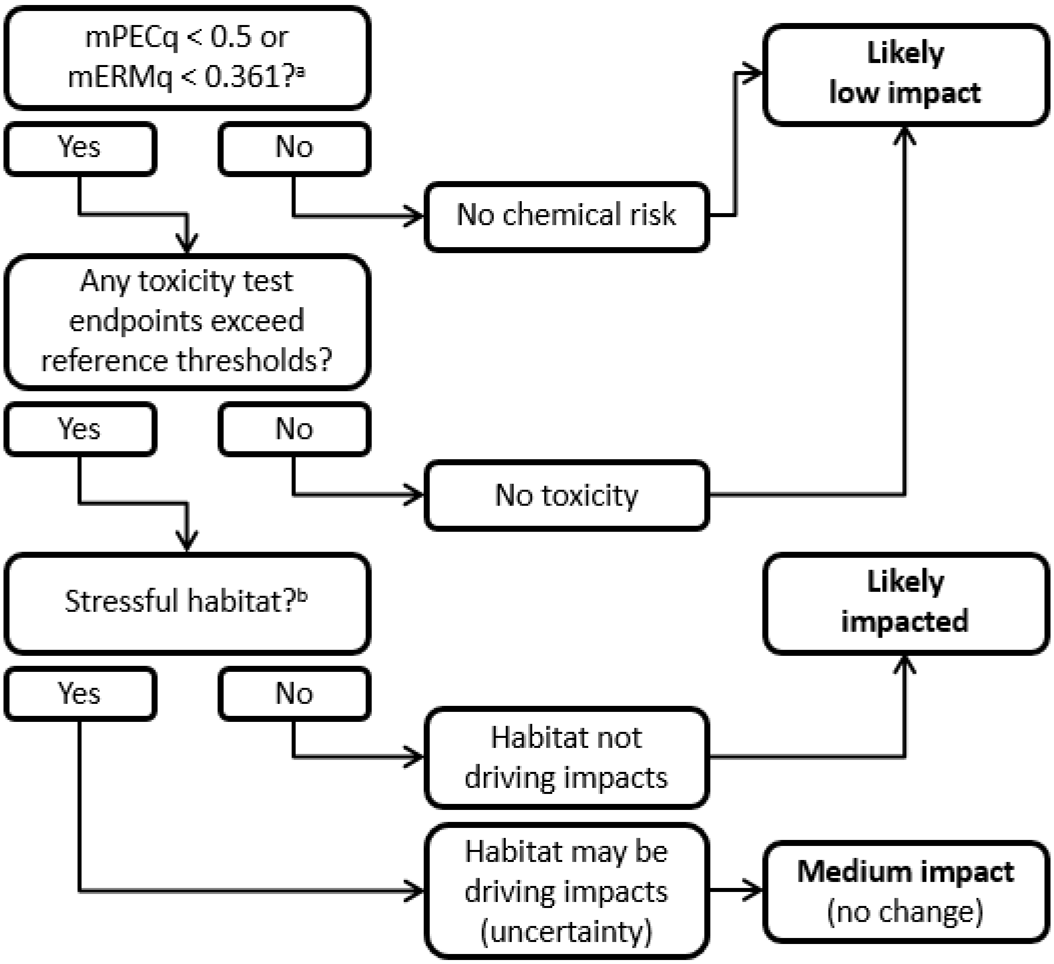
A key point of uncertainty that arises in evaluating the results of the WOE analysis is that 53% of LPRSA SQT locations are categorized as having a medium impact on the benthic community (relative to urban reference conditions); it is uncertain whether a medium impact implies that there is a moderate chemical effect, or that there is uncertainty associated with effects. This is because a medium-impact characterization is warranted when LOEs disagree, or when LOE components (i.e., benthic metrics or sediment toxicity test endpoints) are inconsistently different from reference conditions. Further site-specific analysis is warranted to qualify medium impacts because it is unclear to what extent impacts are attributable to sediment contamination or other confounding variables (e.g., habitat conditions). These uncertainties are evaluated in greater detail below. Also, Appendix B, Table B10, provides data for the post-hoc analysis of LPRSA SQT locations categorized as medium impact.

Methods

In an attempt to further characterize LPRSA locations classified as medium impact by the WOE analysis, available location-specific data were reviewed to determine if a location was more likely to be impacted or unimpacted by chemical concentrations in sediment (Appendix B, Table B10). The refinement process depicted in Figure 6-2 was used to recategorize impacts as likely low impact, likely impacted, or medium impact (unchanged) based on a detailed post-hoc assessment. Sediment chemistry, sediment

[72] The LPRSA location LPRT16B was not evaluated in the WOE analysis because benthic community data were not collected for that sample. Sediment toxicity and chemistry in the sample were relatively low (Appendix P), suggesting that overall, impacts associated with site-related releases of hazardous materials are also low at that location.

toxicity, benthic community metrics, and sediment habitat conditions were evaluated in sequence for each location characterized as medium impact in the WOE analysis. Locations with low sediment chemistry or negligible toxicity were recategorized as likely low impact. Sediment chemistry was considered low if mean probable effects concentration quotients (mPECqs) were less than 0.5, or if mean effects range-median quotients (mERMqs) were less than 0.361. These thresholds are consistent with the sediment chemistry screening step for freshwater reference area data (based on mPECqs) (Appendix P, Section 3.2.2) and with an elevated incidence of degraded benthic communities in the literature (based on mERMqs) (Hyland et al. 2003). Where sediment toxicity is negligible relative to reference conditions, it cannot be said that sediment chemistry is having a toxic impact on LPRSA benthic invertebrate communities. In that case, any observed community impacts at such medium-impact locations could be the result of some unknown factors other than sediment toxicity (e.g., sub-optimal habitat conditions).



a Mean quotient thresholds were based on input from USEPA (mPECq) and Hyland et al. (2003) (mERMq)

b Stressful habitat was defined as having total ammonia exceeding 30 mg/kg, TOC exceeding 3.5% (by mass), and/or total fines exceeding 95% (by mass). Rationales are provided in the text.

Figure 6-2. Flow chart describing post-hoc characterization of medium-impact locations

Habitat conditions (i.e., total ammonia, TOC, and total fines) were evaluated for locations at which sediment chemistry is elevated and toxicity is apparent (compared to reference conditions). Habitat was considered to be stressful if total ammonia exceeded 30 mg/kg, the upper bound of tolerance for *Ampelisca abdita* in sediment toxicity tests (USEPA 1994). Locations with TOC in excess of 3.5% were also considered to have stressful habitat because concentrations exceeding 3.5% TOC are associated with depressed benthic community richness in marine and estuarine habitats (Hyland et al. 2005). Very high levels of fine sediment (> 95% by mass) may reduce habitat suitability and influence sediment toxicity test results (Sibley et al. 1998; Vos et al. 2002); as a result, locations exceeding 95% fines were also said to have stressful habitat.

In cases where there was 1) elevated sediment chemistry (defined above), 2) apparent sediment toxicity (observed for any endpoint), and 3) sediment habitat that did not appear to be stressful, locations were recategorized as likely impacted (Figure 6-2; Appendix B, Table B10). When, even after the evaluation of location-specific data, uncertainty remained regarding the nature of locations characterized as medium impact, the category of medium impact was left unchanged (Figure 6-2). The degree and nature of impacts at those LPRSA SQT locations remains uncertain, although they may be the result of chemical exposures and/or some other confounding factors (e.g., stressful habitat conditions).

Results of Medium-impact Evaluation for WOE Results

LPRSA SQT locations categorized as medium impact in the WOE analysis were re-evaluated (according to Figure 6-2) using additional chemical concentration and habitat data, and taking into account the benthic invertebrate community and sediment toxicity test LOEs. Based on this re-evaluation, several medium-impact locations were recategorized as either likely low impact or likely impacted (Appendix B, Table B10). These new categories reduced the uncertainty associated with the initial WOE conclusions by clarifying whether or not medium impacts could be caused by chemical exposure and toxicity. Reclassified SQT locations remained less certain than locations initially categorized as low impact or high impact, for which the degrees of impacts were clearer. Stations recategorized as likely low impact had low sediment chemistry and/or negligible sediment toxicity (relative to the reference condition), and stations recategorized as likely impacted had elevated sediment chemistry and toxicity and suitable habitat conditions, suggesting that measured effects were more likely the result of chemical exposure than of some other confounding factor (e.g., habitat-related stress).

The results of the medium-impact evaluation (based on WOE results) are provided in Appendix B (Tables B10-2 and B10-4). Those results (in addition to the unchanged no-impact, low-impact, and high-impact conclusions) are summarized below and in Tables 6-8 and 6-9.

Table 6-8. Summary of WOE results after post-hoc medium-impact evaluation, urban comparison

Benthic Salinity Zone	N	No Impact		Low Impact		Medium Impact[a]						High Impact	
						Likely Low Impact		Medium Impact (Unchanged)		Likely Impacted			
		n	%	n	%	N	%	n	%	n	%	n	%
Upper estuarine (RM 0 to RM 4)	25	0	0%	13	52%	0	0%	5	20%	5	20%	2	8%
Fluvial estuarine (RM 4 to RM 13)	54	0	0%	14	26%	2	4%	26	48%	7	13%	5	9%
Tidal freshwater (RM 13 to RM 17.4)	18[b]	0	0%	1	6%	6	33%	0	0%	0	0%	11	61%
Site wide	97	0	0%	28	29%	8	8%	31	32%	12	12%	18	19%

a Medium-impact locations were re-evaluated using a post-hoc analysis; based on several factors, SQT locations were recategorized as likely low impact, likely impacted, or unchanged (medium impact) (Appendix B, Table B10)

b Of the 98 locations sampled in fall 2009 for sediment chemistry analyses and toxicity testing, benthic invertebrate communities were only analyzed at 97 locations. The WOE analysis was conducted at only the 97 locations for which all three types of SQT data were collected.

n – sample size (by category)
N – sample size (by benthic salinity zone or site-wide)
RM – river mile
WOE – weight of evidence

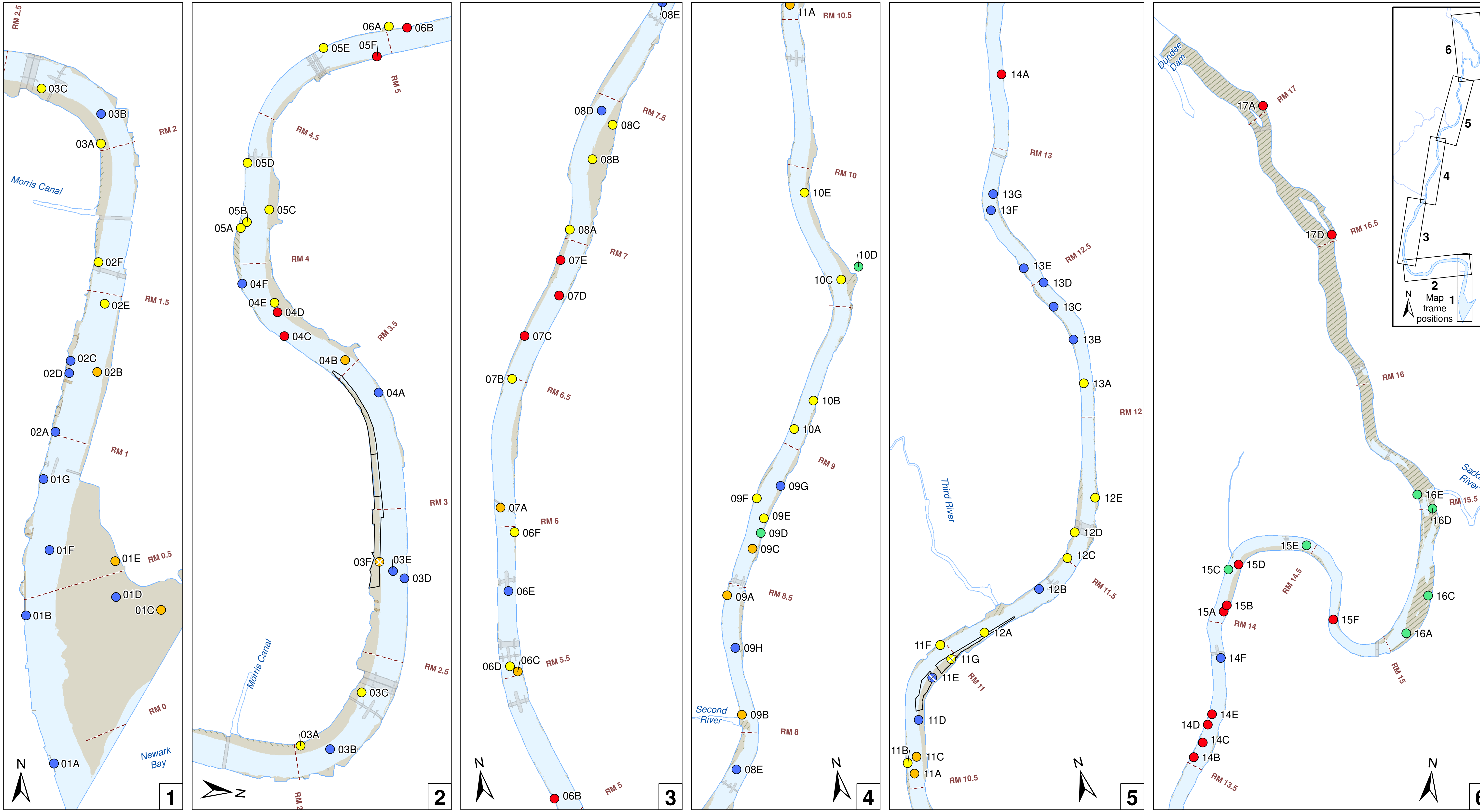
Table 6-9. Summary of WOE results after post-hoc medium-impact evaluation, non-urban comparison

Benthic Salinity Zone	N	No Impact		Low Impact		Medium Impact						High Impact	
						Likely Low Impact		Medium Impact (Unchanged)		Likely Impacted			
		n	%	n	%	n	%	n	%	n	%	n	%
Upper estuarine (RM 0 to RM 4)	25	0	0%	12	48%	0	0%	4	16%	6	24%	3	12%
Fluvial estuarine (RM 4 to RM 13)	54	0	0%	8	15%	5	9%	23	43%	9	17%	9	17%
Both estuarine zones (RM 0 to RM 13)	79	0	0%	20	25%	5	6%	27	34%	15	19%	12	15%

n – sample size (by category)
N – sample size (by benthic salinity zone)
RM – river mile
WOE – weight of evidence

The WOE analysis (based on comparison to urban reference conditions) initially categorized 51 (53%) of 97 LPRSA SQT locations as having a medium impact (Table 6-6). Based on the post-hoc analysis, 8 of the 52 locations were recategorized as having a likely low impact, and 12 were recategorized as likely impacted. The remaining 31 locations were not recategorized and remain uncertain; the impacts observed at those 31 locations may be moderate, but the cause of effects is unclear. The majority of those locations were in the fluvial estuarine zone. Most of the LPRSA locations that were recategorized as having likely low impact had low sediment chemistry (mPECq < 0.5 or mERMq < 0.361), and many of those locations were in the tidal freshwater zone. Most of the locations recategorized as being likely impacted were in the upper and fluvial estuarine zones, where sediment toxicity was observed (relative to the reference condition) but where habitat appeared to be suitable for benthic invertebrates.

The WOE analysis comparing LPRSA data to non-urban reference conditions resulted in 47 locations being categorized as having a medium impact (Table 6-7). Of those, 5 were recategorized as likely low impact and 15 were recategorized as likely impacted. Benthic invertebrate risk at the remaining 27 locations remains uncertain. The results of the WOE analyses (after post-hoc analysis of medium impacts) based on urban and non-urban reference area comparisons are provided in Figures 6-3 and 6-4, respectively (and Appendix B, Tables B10-2 and B10-4, respectively).



SQT weight of evidence
- High impact
- Likely impacted
- Medium impact
- Likely low impact
- Low impact

× Remediated location[a]
Dredge zone
River mile
LPRSA

Bridge
Abutment
Dock
Mudflat
Gravel with fines

0 0.25 0.5 Kilometers

0 0.25 0.5 Miles

[a]One sample was collected in the Lister Ave. dredge area at RM 2.8 and two were collected in the RM 10.9 dredge area.

Mudflat areas for ecological receptors are those areas where the river bottom slope is ≤ 6° and the depth is ≥ -2 ft MLLW, based on the 2007 CPG bathymetric survey.

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Figure 6-3. Conclusion of weight of evidence analysis of SQT data from the LPRSA

Lower Passaic River Study Area Baseline Ecological Risk Assessment

FINAL

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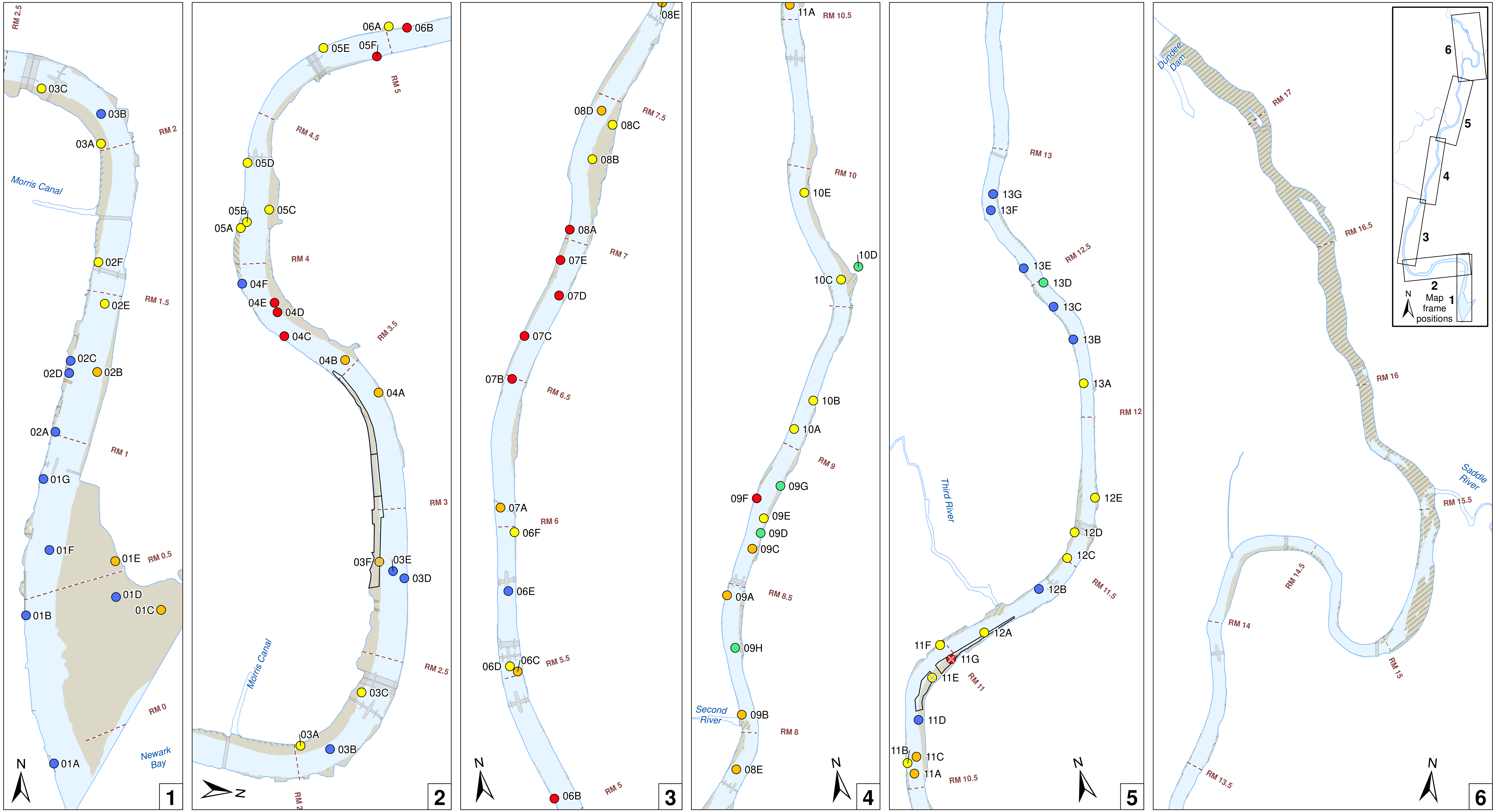

SQT weight of evidence
- High impact
- Likely impacted
- Medium impact
- Likely low impact
- Low impact

× Remediated location[a]
- Dredge zone
- Bridge
- Abutment
- Dock

- River mile
- Mudflat
- Gravel with fines
- LPRSA

[a]One sample was collected in the Lister Ave. dredge area at RM 2.8 and two were collected in the RM 10.9 dredge area.

Note: Likelihood of benthic impacts was determined by comparing LPRSA data with reference data representing non-urban habitats.

Mudflat areas for ecological receptors are those areas where the river bottom slope is ≤ 6° and the depth is ≥ -2 ft MLLW, based on the 2007 CPG bathymetric survey.

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Figure 6-4. Benthic impacts in the LPRSA based on WOE analysis, non-urban comparison

Lower Passaic River Study Area
Baseline Ecological Risk Assessment

FINAL

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6.1.3 Uncertainty analysis

This section outlines uncertainties associated with the WOE analysis (and component LOEs) that have an influence over the interpretation of ecological risks. A WOE analysis is presented herein to address several key uncertainties. More detailed discussions of uncertainties associated with each LOE are provided in Appendix P (Sections 2.3.4, 3.2.5, and 4.3). The following are key uncertainties:

- It is unclear whether the screened reference area datasets accurately reflect the reference condition. Specifically, the screening of reference area data using sediment chemistry or sediment toxicity criteria imposes a potentially unreasonable constraint on data acceptability. The resulting datasets may not capture the full range of possible benthic community metrics or sediment toxicity test results that should be expected under natural conditions (but for the LPRSA-specific release of hazardous materials). The screening approach in the quantitative analysis of uncertainty does not include a strict screening step for sediment toxicity test results; rather, extreme toxicity values (i.e., very low survival relative to the interquartile range of toxicity test data) are removed from the reference dataset.
- A comparison of LPRSA data to data from a non-urban reference area (Mullica River and Great Bay) is likely less relevant for characterizing risks in the LPRSA than a comparison of LPRSA data to urban conditions data. Comparison to non-urban conditions may not incorporate potential stressors that are generally observed in urban settings and are expected to influence the LPRSA benthic invertebrate community. Examples of these stressors include altered hydrology due to channelization and flood controls and increased organic and inorganic inputs from CSOs, SWOs, road waste, and permitted industrial discharges.
- Screening LPRSA data against T20 and T50 values is highly conservative for defining ecological risks. Field et al. (2002) note that the "LRM approach provides a useful framework for conducting screening-level assessments…" and that the model does not consider site-specific bioavailability or exposure. Furthermore, the T20 and T50 values are based on field-collected sediments rather than controlled sediments (Field et al. 2002), so they are likely to contain more hazardous substances than the one for which the criteria were developed. As a result, it is likely that T20 and T50 values overestimate the toxicity of the single contaminants for which they are reported. Mean-quotients are also based on sediment quality guidelines that are meant primarily for screening purposes, and that are based (at least in part) on data from field-collected sediment (with mixtures of sediment contaminants) (Long et al. 1995; MacDonald et al. 2000; Wenning et al. 2005), so the mean-quotient approach does not address those specific uncertainties.

- The logistic modeling approach used to derive T20 and T50 values does not address the magnitude of the relationship between concentration and "toxic" response. Therefore, it is not possible to determine what level of effect can be expected (i.e., what magnitude of risk to invertebrates) from exceedances of T20 and T50 values. The use of mean-quotient values as part of the approach discussed in this uncertainty section partly addressed the magnitude of possible effects by scaling sediment concentrations to concentrations correlated with toxic impacts in the literature. However, there are also substantial uncertainties associated with the ERM and probable effects concentration (PEC) guidelines (Wenning et al. 2005).
- T50 values are unreliable as predictors of toxicity in the LPRSA (as determined using reliability statistics in Appendix P, Section 4.3.1).
- The medium-impact classification for WOE analysis results suggests uncertainty associated with potential risks. Risk uncertainty at LPRSA SQT locations classified as medium impact has been minimized to the extent practicable by using a post-hoc analysis (Section 6.1.2.3).
- Given the degree of uncertainty in the sediment chemistry LOE (Appendix P, Section 4.3), it is unclear whether assigning that LOE a weight equivalent to that of the benthic community and sediment toxicity LOEs (up to 1.0) in the WOE analysis is appropriate. Appendix P (Sections 4.2.1, 4.2.2, and 4.3) provides several analyses showing that sediment toxicity test and benthic invertebrate community metric data from LPRSA SQT locations are weakly or inconsistently related to sediment chemistry, suggesting that sediment chemistry, alone, does not explain measured toxicity or impaired benthic communities. Results from multivariate analyses (Appendix P Section 4.2.2) indicate that sediment chemistry is generally negatively associated with benthic invertebrate community metrica and sediment toxicity test results, though these effects are often also related, at least in part, to habitat variables. Moreover, the relationships, though statistically significant, tend to be fairly weak. The sediment chemistry and sediment toxicity LOEs provide direct mesaurements of effects, making them far more certain for making risk conclusions than the sediment chemistry LOEs.Impacts at freshwater LPRSA SQT locations LPRT17A and LPRT17D are potentially influenced (at least in part) by differences in habitat conditions immediately below Dundee Dam compared with the area above Dundee Dam. The area above the dam has finer sediments than the area just below, which is predominately composed of coarse sand and cobble. In general, such sediments are not expected to have elevated sediment contamination. Changes to stream hydrology caused by the dam may also contribute to observed impacts at LPRT17A and LPRT17D.

6.1.3.1 Evaluation of uncertainty in WOE analysis

In order to quantitatively address key uncertainties associated with the WOE analysis, a quantitative analysis was applied to each SQT LOE; the results of the quantitative analysis of uncertainty are presented in Appendices B and P and are summarized below.

Methods

The two major uncertainties in the WOE approach described (Section 6.1.1) were quantitatively analyzed. First, reference area sediment samples with extremely low toxicity test survival were identified and removed from reference datasets, resulting in different reference envelope thresholds. Second, the sediment chemistry LOE was evaluated using reference area-specific data, rather than T20 and T50 values from the literature. Since the reference area dataset was fixed at a minimum toxicity value (i.e., 75 or 80% survival relative to the negative control for freshwater or estuarine toxicity test results, respectively), the 5th percentile of the acceptable reference area data—which is compared to LPRSA toxicity data in the sediment toxicity LOE (Appendix P and Appendix B, Tables B3 and B4)—may not be a well-suited expression of sediment toxicity associated with background conditions for the greater NY/NJ Harbor Estuary complex. The establishment of a reference condition is intended to address potential stress associated with natural conditions (without the influence of the hazardous substances), but this condition is not captured when an arbitrary bound is imposed on the reference data. The quantitative analysis for establishing reference conditions does not include the arbitrary bound on sediment toxicity, but rather uses an outlier test to eliminate extreme values that are inconsistent with the majority of reference data. Criticisms of the T20 and T50 criteria are provided in Section 6.1.3 and detailed in Appendix P (Section 4.1.4.1). The bounding analysis for establishing reference conditions for the LPRSA attempts to scientifically address uncertainties associated with the conservatism of the T20 and T50 screen in the sediment chemistry LOE (Appendix P) by using reference area-specific sediment chemistry thresholds.

For the WOE analysis approach, reference data were screened using both sediment chemistry and sediment toxicity test data (Appendix B, Tables B3 and B4, and Appendix P, Sections 2.3.1 and 3.2.2); however, it is unclear whether the consequent dataset effectively captures reference conditions expected in the greater NY/NJ Harbor Estuary complex that would be applicable to the LPRSA, particularly those influenced by urban stressors. Screening sediment samples based on chemistry data is reasonable because the reference condition is meant to be relatively free of contamination, insofar as contamination is associated with the site-specific release of hazardous substances (and not elevated ambient, urban pollutants). However, those locations with acceptably low sediment chemistry should not also be screened using a criterion for sediment toxicity. Any level of toxicity in sediments with low chemistry is consistent with a reference condition. Based on this reasoning, reference area datasets for the quantitative uncertainty analyses were screened using only sediment chemistry data (Appendix B, Tables B3 and B4). Sediment toxicity test results that appeared to be statistically extreme (and therefore inconsistent with the distribution of reference area data) were removed as outliers. This analysis generally resulted in larger reference area datasets and different reference envelope thresholds (Appendix B, Tables B3 and B4, and Appendix P, Sections 2.3.1 and 3.2.2).

For addressing uncertainties in the sediment chemistry LOE, LPRSA data from the upper and fluvial estuarine benthic salinity zones were used to calculate mERMq values, and LPRSA data from the tidal freshwater benthic salinity zone was used to calculate mPECq values. Similarly, these mean-quotient values were calculated for the reference areas of Jamaica Bay (i.e., mERMq) and above Dundee Dam (i.e., mPECq). Reference area mean-quotients were compared to reference area sediment toxicity data, and low and high mERMq and mPECq thresholds were set for each toxicity test endpoint (Appendix B, Table B7-2, and Appendix P, Section 4.1.4.4). These thresholds were used in the bounding analysis to assign sediment chemistry LOE weights for LPRSA SQT locations (Table 6-4; Appendix P, Section 4.3.5.1, and Appendix B, Table B7-2).

The LOEs were combined using the WOE analysis framework described in Sections 6.1.1 and 6.1.2; those results are presented in the following section as a bounding analysis for the WOE outcomes presented in risk characterization.

Results

Summary WOE analyses of benthic community risks based on comparisons to urban and non-urban reference conditions are provided in Tables 6-10 and 6-11, respectively. Results of the analyses are divided according to benthic salinity zones. Further information regarding these analyses is provided in Appendix B (Tables B8 and B9).

Table 6-10. Summary of bounding WOE analysis results, urban comparison

Benthic Salinity Zone	N	No Impact		Low Impact		Medium Impact		High Impact	
		n	%	n	%	n	%	n	%
Upper estuary (RM 0 to RM 4)	25	4	16	17	68	3	12	1	4
Fluvial estuary (RM 4 to RM 13)	54	10	19	32	59	12	22	0	0
Tidal freshwater (RM 13 to RM 17.4)	18[a]	2	11	8	44	8	44	0	0
Site wide	97	16	16	57	59	23	24	1	1

Note: Reference data representing urban habitats are from Jamaica Bay and the area above Dundee Dam. Medium-impact results are characterized in greater detail in Section 6.1.2.3.

[a] No benthic invertebrate community data were available at LPRT16B, so no WOE result was determined.

% – percentage of locations

n – number of locations for each WOE conclusion

N – number of locations in each benthic salinity zone

RM – river mile

WOE – weight of evidence

Table 6-11. Summary of bounding WOE analysis results, non-urban comparison

Benthic Salinity Zone	N	No Impact		Low Impact		Medium Impact		High Impact	
		n	%	n	%	n	%	n	%
Upper estuary (RM 0 to RM 4)	25	1	4	14	56	8	32	2	8
Fluvial estuary (RM 4 to RM 13)	54	6	11	32	59	14	26	2	4
Both estuarine zones (RM 0 to RM 13)	79	7	7%	46	47%	22	23%	4	4%

Note: Reference data representing urban habitats are from Mullica River/Great Bay. Medium-impact results are characterized in greater detail in Section 6.1.2.3. Freshwater reference area data were not available to compare to LPRSA tidal freshwater data.

% – percentage of locations
LPRSA – Lower Passaic River Study Area
n – number of locations for each WOE conclusion
N – number of locations in each benthic salinity zone
RM – river mile
WOE – weight of evidence

Based on the summary presented in Table 6-10, the majority of locations (73 of 97 locations, or 75%) received a WOE score that indicated no or low impact on the LPRSA benthic community from site-related releases of hazardous substances compared to an urban reference. Of the 97 LPRSA SQT locations, 23 (24%) had WOE scores that indicated medium impact. This left one LPRSA location classified as high impact. In comparison to the results of the WOE analysis presented in risk characterization (Section 6.1.2.1), the quantitative analysis of uncertainty suggests that LPRSA locations are bounded by conditions that indicate limited impacts on LPRSA benthic communities (relative to urban reference conditions) by site-related releases of hazardous materials. High impacts are observable at a very small portion (1%) of LPRSA locations, while moderate, more uncertain[73] impacts are observable at a minority (24%) of LPRSA locations.

Post-hoc analysis of the 23 locations categorized as medium impact resulted in the recategorization of 7 locations, 3 as likely low impact and 4 as likely impacted, leaving 16 locations unchanged (medium impact) (Appendix B Table B10-6).

Table 6-11 indicates that LPRSA SQT locations (in the upper and fluvial estuarine zones) are marginally more impacted when compared to non-urban conditions than when compared to urban conditions (Section 6.1.2.1).

When comparing the results presented in Table 6-6 to those presented in Table 6-10, or the results presented in Table 6-7 to those presented in Table 6-11, it can be seen that the primary approach resulted in a greater frequency of impacted LPRSA SQT

[73] Medium impacts determined in the quantitative analysis of uncertainty for the WOE analysis are further evaluated in Appendix B, Tables B10-3 and B10-5, and summarized in Table B10-6. The results of that analysis are analogous to those presented in Section 6.1.2.2, although they are not discussed in this BERA.

locations. The results of the quantitative analysis provides a bounding analysis that addresses two important uncertainties in the overall WOE analysis.

The following uncertainties are associated with the quantitative analysis presented above:

- A high mERMq threshold could not be developed for non-urban reference conditions because acceptable *A. abdita* survival was observed even at the highest calculated mERMq. Because of this, the sediment chemistry LOE comparing the LPRSA to non-urban reference conditions was based on mERMq thresholds developed using the urban reference condition. The use of the urban reference condition mERMq value may underestimate the sediment chemistry LOE for the comparison of LPRSA data to non-urban conditions data.
- The mERMq and mPECq thresholds used for predicting toxicity in the LPRSA may underpredict toxicity.

6.1.4 Conclusions and summary

As outlined in Section 2, the physical, hydrological, and habitat characteristics (e.g., TOC and sediment grain size) observed and modeled in the LPRSA are generally consistent with those of many other urban systems, and these non-chemical stressors can alter benthic community structure and function. Sediment contamination in the LPRSA also has the potential to cause toxicity to or alter benthic community structure and function. Statistical analysis in Appendix P (Section 4.2.2) indicates that benthic invertebrate community structure and sediment toxicity are negatively associated with a mixture of chemicals and/or habitat variables in LPRSA sediment. Through multivariate analysis, it was determined that 10 of 11 measurement endpoints are negatively associated with sediment chemical concentrations, with habitat conditions also having negative associations with several measurement endpoints. For example, benthic invertebrate diversity, survival, and growth are negatively associated with a mixture of chemicals including metals and pesticides (e.g., total DDx and hexachlorobenzene) along with the percent of total fine-grained sediment (a habitat variable). Some measurement endpoints were related to only chemistry (e.g., Shannon-Wiener H' and SDI) or only habitat (abundance). A number of the benthic measurement endpoints had chemical factors that were more important for predicting effects than habitat variables.

Survival and biomass in toxicity tests and benthic diversity in field measurements decrease with increasing Factor scores, including Factor 2 which represents a mixture of metals, total DDx, and hexachlorobenzene. Mixtures of chemicals were found to co-vary spatially (indicated by factor analysis) indicating that multiple COPECs, either singly or as a mixture are likely responsible for benthic impairment in the LPRSA. Because many COPECs were correlated with a small number of factors, it is not possible to identify any single chemical driver of benthic invertebrate risk from the multivariate analysis. Additional LOEs related to fate, toxicity, and bioavailability of specific contaminants would be needed to reduce the uncertainty associated with identifying individual COPECs as risk drivers for benthic invertebrates.

It is likely that the observed benthic invertebrate impacts are the result of exposure to multiple LPRSA-related COPECs, and these impacts are likely exacerbated by habitat conditions. Based on this conclusion, the sediment chemistry LOE is included in the overall WOE evaluation in the BERA risk characterization (Section 6.1.2).

Because there can be a mixed effect from sediment chemical factors and stressful, urban habitat conditions, the comparison of LPRSA data to an urban reference condition is the most relevant approach for characterizing benthic invertebrate risks. The focus of the following conclusions is on the results of urban reference comparison.

Based on the characterization of risk in Section 6.1.2 (and its subsections), the following conclusions can be made regarding the potential impacts on benthic invertebrates at LPRSA locations relative to reference conditions:

- Based on the WOE analysis, the number of LPRSA SQT locations with high impacts (compared to urban conditions) was 18 (of 97); the number of locations with no impact was 0; and the number of locations with low impacts was 28. A comparison to non-urban conditions resulted in a marginal increase in risk at upper and fluvial estuarine LPRSA locations; a similar comparison could not be made for tidal freshwater LPRSA locations.[74]

  Of the 97 SQT locations, 51 (53%) had medium impacts, suggesting that the results from the WOE analysis were relatively uncertain; LOEs either disagreed or had limited agreement, or impacts were moderate. Additional site-specific evaluations of these uncertain impacts resulted in 8 stations being recategorized as likely low impacts, and 12 stations being recategorized as likely impacted (with 31 remaining unchanged). That brought the fraction of LPRSA locations with no, low, or likely low impacts to 37%, and the fraction of locations with high impacts or likely impacts to 31%. The remaining 32% of LPRSA SQT stations stayed at a relatively unclear level of medium impact, possibly associated with moderate chemical risk (and exacerbated by other confounding factors such as habitat conditions). Impacts were, thus, observed at 63% of SQT locations.

- A quantitative analysis of two important uncertainties in the WOE—related to establishing reference conditions used to assess LPRSA toxicity response data and benthic community metrics, and how the sediment chemistry LOE is scored—provided a bounding estimate for the WOE analysis. The quantitative

[74] The comparison of LPRSA data to an acceptable, non-urban freshwater reference dataset was not possible.

analysis resulted in a shift in the WOE outcomes to more locations with scores that fell within the no- and low-impact categories (i.e., 75% rather than 29%).[75]

- Therefore, the percentage of benthic invertebrate communities with a WOE score classified as in the high-impact category ranged between 1 and 19%, and the percentage of benthic communities classified as having a limited impact (combination of no- and low-impact classifications) ranged between 29 and 75%.

6.2 Surface Water Assessment

The surface water assessment was conducted for benthic invertebrates (including the benthic invertebrate community, macroinvertebrates, and mollusks) to evaluate the effect of direct exposure to COPECs in surface water. Risk estimates are expressed as HQs, which were derived by comparing surface water EPCs with TRVs.

6.2.1 COPECs

Surface water COPECs for benthic invertebrates were identified in the SLERA as COIs with maximum concentrations equal to or exceeding their respective screening thresholds (Table 6-12).

Table 6-12. Surface water COPECs evaluated for invertebrates

COPECs	Estuarine (RM 0–RM 13)	Freshwater (RM 4–RM 17.4)
Metals[a]		
Cadmium	X	X
Chromium	X	X
Copper	X	X
Lead	X	X
Mercury	X	X
Selenium	X	X
Silver	X	X
Zinc	X	X
Butyltin		
TBT	X	
PAHs		
Anthracene	X	X
Benzo(a)anthracene	X	X

[75] Within the quantitative analysis of uncertainty, medium-impact locations were also reclassified (i.e., as likely low impact or likely impacted), but changes to medium-impact locations were not included in these percentages.

Table 6-12. Surface water COPECs evaluated for invertebrates

COPECs	Estuarine (RM 0–RM 13)	Freshwater (RM 4–RM 17.4)
Benzo(a)pyrene	X	X
Fluoranthene	X	X
Pyrene	X	X
SVOCs		
BEHP	X	X
BBP	X	X
PCBs		
Total PCBs	X	X
PCDDs/PCDFs		
2,3,7,8-TCDD	X	X
Pesticides		
4,4′-DDE	X	X
4,4′-DDT	X	X
Dieldrin	X	
Hexachlorobenzene	X	X
Total chlordane	X	X
Total DDx	X	X
Other		
Cyanide	X	X

Note: X indicates COPEC based on SLERA HQ ≥1.0.

[a] All metals were identified as COPECs based on the total concentrations.

BBP – butyl benzyl phthalate
BEHP – bis(2-ethylhexyl) phthalate
COPEC – chemical of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
HQ – hazard quotient
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
PCDD – polychlorinated dibenzo-*p*-dioxin
PCDF – polychlorinated dibenzofuran
RM – river mile
SLERA – screening-level ecological risk assessment
SVOC – semivolatile organic compound
TBT – tributyltin
TCDD – tetrachlorodibenzo-*p*-dioxin
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT and 4,4′-DDT)

A number of COIs could not be screened as part of the SLERA (Appendix A) because no freshwater or estuarine TSVs were available. These COIs are presented in Section 5.5.2, as are the implications of not being able to evaluate these COIs.

6.2.2 Exposure

The surface water EPCs for benthic invertebrates were calculated separately for two areas: between RM 0 and RM 13 for comparison to estuarine thresholds, and between

RM 4 and RM 17.4 for comparison to freshwater thresholds. Surface water data were limited to RM 10.2, as there were no sampling locations between RM 10.2 and Dundee Dam. As a result, the freshwater dataset is smaller than the estuarine surface water dataset. Benthic invertebrates are found throughout the LPRSA and are exposed to LPRSA surface water at the subsurface, or near-bottom, portion of the water column. Only near-bottom surface water (3 ft [0.9 m] above the bottom) data collected throughout the LPRSA during various flow events in 2011, 2012, and 2013 (see Table 4-4) were used in EPC calculations for the benthic invertebrate surface water assessment.

Surface water EPCs for benthic invertebrates were based on a conservative upper-bound estimate of the mean concentration (i.e., UCL). UCLs were calculated using USEPA's ProUCL® statistical package (Version 5.0.00) (USEPA 2013d), as described in Section 4.3.7.[76] UCL concentrations could not be derived for one COPEC (TBT) because there were no detected concentrations in near-bottom water samples; therefore, the maximum DL of 0.05 mg/L was used as the EPC. Summary concentrations in near-bottom surface water samples are presented in Table 6-13.

[76] The UCL recommended by USEPA's ProUCL® software (typically the 95% UCL, but in some cases the 97.5% or even the 99% UCL) was used. The selected UCL statistic is presented in Appendix C1.

Table 6-13. Summary statistics for near-bottom surface water concentrations

COPEC	Estuarine (RM 0–RM 13)						Freshwater (RM 4–RM 17.4)					
	No Detects/ No Samples	%	Concentration				No Detects/ No Samples	%	Concentration			
			Min.	Max.	Mean	UCL			Min.	Max.	Mean	UCL
Metals (µg/L)												
Cadmium (dissolved)	138/160	86.2	0.01	0.149	0.051	0.049	56/77	72.7	0.01	0.149	0.042	0.04
Chromium (dissolved)	98/100	98	0.21	5.46	0.81	0.92	49/49	100	0.28	5.46	1	1.2
Copper (dissolved)	160/160	100	1.11	9.26	2.54	2.7	77/77	100	1.36	9.26	2.97	3.26
Lead (dissolved)	160/160	100	0.07	9.97	0.89	1.4	77/77	100	0.098	9.97	1.3	2.2
Mercury (dissolved)[a]	159/160	99.4	0.28	91.5	7.9	12	77/77	100	0.45	91.5	9.6	18
Selenium (dissolved)	27/100	27	0.2	1.8	0.47	0.49	24/49	49	0.2	1.8	0.46	0.5
Silver (dissolved)	62/100	62	0.004	0.119	0.02	0.019	24/49	49	0.004	0.119	0.03	0.027
Zinc (dissolved)	100/100	100	1.54	18.5	7.2	7.8	49/49	100	2.1	18.5	7	8.2
Butyltin (µg/L)												
TBT	0/100	0	na	na	na	na	0/49	0	na	na	na	na
PAHs (ng/L)												
Anthracene	95/100	95	1.81	140	16.4	19.5	49/49	100	2.41	120	17.8	20.6
Benzo(a)anthracene	95/100	95	3.89	316	48.2	55.7	49/49	100	6.65	316	62.1	75.4
Benzo(a)pyrene	93/100	93	7.52	560	78.1	89.6	49/49	100	9.67	560	102	134
Fluoranthene	100/100	100	14.9	583	125	143	49/49	100	26.1	583	169	199
Pyrene	100/100	100	23.2	587	130	147	49/49	100	23.8	587	171	199
SVOCs (µg/L)												
BEHP	10/84	11.9	1.2	4.8	2.4	1.7	7/45	15.6	1.2	3.9	2.3	1.8
BBP	26/85	30.6	0.14	25	1.3	1.2	16/46	34.8	0.14	25	1.8	1.9

Table 6-13. Summary statistics for near-bottom surface water concentrations

COPEC	Estuarine (RM 0–RM 13) No Detects/ No Samples	%	Concentration Min.	Max.	Mean	UCL	Freshwater (RM 4–RM 17.4) No Detects/ No Samples	%	Concentration Min.	Max.	Mean	UCL
PCBs (ng/L)												
Total PCBs	160/160	100	0.0499	183	24.5	32.9	77/77	100	2.05	183	31.2	38.9
PCDDs/PCDFs (ng/L)												
2,3,7,8-TCDD	135/160	84.4	0.00099 6	1.83	0.0241	0.0704	71/77	92.2	0.00099 6	1.83	0.0402	0.141
Organochlorine Pesticides (ng/L)												
4,4′-DDE	91/100	91	0.24	8.26	1.4	1.5	46/49	93.9	0.29	8.26	1.9	2.9
4,4′-DDT	73/100	73	0.0509	3.82	0.56	0.54	44/49	89.8	0.0982	3.82	0.78	0.95
Dieldrin	88/100	88	0.16	3.18	1.1	1.1	49/49	100	0.412	3.18	1.5	1.6
Hexachlorobenzene	25/100	25	0.0836	2.57	0.402	0.21	11/49	22.4	0.154	1.04	0.437	0.23
Total Chlordane	100/100	100	0.0967	15.9	2.91	3.42	49/49	100	0.881	15.9	4.64	5.69
Total DDx	99/100	99	0.216	21.1	3.54	4.17	49/49	100	0.443	21.1	5.03	6.77
Other (mg/L)												
Cyanide	7/100	7	0.003	0.031	0.01	0.0078	6/49	12.2	0.003	0.014	0.007	0.0075

[a] Dissolved mercury concentrations are in ng/L.

BBP – butyl benzyl phthalate
BEHP – bis(2-ethylhexyl) phthalate
COPEC – chemical of potential ecological concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
na – not applicable (not detected)
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
PCDD – polychlorinated dibenzo-*p*-dioxin
PCDF – polychlorinated dibenzofuran
RM – river mile
SVOC – semivolatile organic compound
TBT – tributyltin
TCDD – tetrachlorodibenzo-*p*-dioxin
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT and 4,4′-DDT)
UCL – upper confidence limit on the mean

There was overlap between the datasets used to calculate the estuarine and freshwater EPCs, as they both included surface water data from between RM 4 and RM 10.2. Except for cadmium and cyanide, the freshwater EPCs (based on UCLs) for all other COPECs were slightly greater than the estuarine EPCs (Table 6-13).

In general, the EPCs for COPECs in near-bottom surface water were slightly greater than the site-wide EPCs used in the surface water assessment for fish (Section 7). For most COPECs, the maximum detected concentration was in the near-bottom surface water samples; the exceptions were TBT, BEHP, and 2,3,7,8-TCDD in the estuarine and freshwater portions of the LPRSA, and hexachlorobenzene in the freshwater portion (Table 6-13; Section 7).

Surface water chemistry results for organic chemicals were analyzed in whole-water samples (AECOM 2012c). EPCs for nonionic organic chemicals (e.g., PAHs, PCBs, and organochlorine pesticides) based on total concentrations in whole-water samples may overestimate the fraction of these chemicals that is bioavailable to aquatic organisms. The bioavailability of nonionic organic chemicals is influenced by dissolved and particulate OC present in the water column, concentrations of which determine the fraction of the chemical that is freely dissolved and, thus, bioavailable (Burkhard 2000).

6.2.3 Effects

TRVs were selected for each surface water COPEC. TRVs were determined using aquatic life-based criteria from existing USEPA ambient water quality criteria (AWQC), or from the published literature based on the USEPA AWQC derivation approach. If aquatic life-based criteria were not available, then chronic TRVs were derived to predict risk to benthic invertebrates and fish using up-to-date toxicological data relevant to aquatic species.[77] The TRV derivation method was dependent on the availability of sufficient, applicable toxicity data for a given COPEC. To the extent practicable, the TRV derivations were generally consistent with USEPA AWQC methodology (Stephan et al. 1985). Aquatic toxicity datasets were compiled for each COPEC using USEPA's ECOTOX database (USEPA 2016c). Datasets were limited to include only data relevant to aquatic species. TRVs were intended to be protective of aquatic organisms, including benthic invertebrates, zooplankton, and fish. The following sections present the methods used to derive the surface water TRVs and the selected TRVs for each of the COPECs.

[77] Some screening levels (i.e., for total PCBs, 2,3,7,8-TCDD, and other organic COIs) used in the SLERA were protective of wildlife or human health (i.e., a 304(a) aquatic life criterion using the final residue value [FRV] procedure issued in 1980 or 1986; this procedure is no longer used by USEPA to derive chronic criteria). Such screening levels were not used to evaluate the exposure of aquatic invertebrates and fish to surface water.

6.2.3.1 Methods for deriving surface water TRVs

This section describes the methods used to derive surface water TRVs using species sensitivity distributions (SSDs)[78] when an acceptable value based on the AWQC approach was not available; the section also describes the acceptability criteria for the use of toxicity data when deriving a TRV using AWQC methods. In addition, general uncertainties associated with the TRV derivation process are described. Further details on the surface water TRV derivation process is presented in Appendix D.

TRV Derivation

Acute and chronic surface water TRVs for the evaluation of risks to benthic invertebrates, zooplankton, and fish were selected using the following approach for each COPEC:

1. The USEPA AWQC were selected as TRVs, unless the available AWQC were based on the protection of wildlife or human health, in which case the original AWQC documents were reviewed to identify the criteria relevant to aquatic life (i.e., the final acute value [FAV] and final chronic value [FCV]). The FAV and FCV are typically based on SSDs. If an FAV and an FCV had been developed by USEPA, these values provided the basis for the selected acute and chronic TRVs. Consistent with Stephan et al. (1985), the FAV was divided by two to provide the acute TRV (to estimate a low-effect concentration, as the FAV is based on EC50 [concentration that causes a non-lethal effect in 50% of an exposed population] values). Chronic toxicity data were often insufficient (i.e., did not meet USEPA's "eight family rule") to derive an FCV directly, so an acute-to-chronic ratio (ACR) was applied to the FAV to provide the FCV; an ACR is the ratio of acute and chronic toxicity values, and in deriving FCVs, a chemical-specific ACR is used. The resulting FCV was selected as the chronic TRV.[79]

2. In cases where updated criteria have been developed following the USEPA AWQC approach using new data and improved methods, these values were selected as TRVs. Such updated criteria have been published for copper and saltwater (Chadwick et al. 2008), lead and saltwater (Church et al. 2017), lead and freshwater (DeForest et al. 2017), silver and freshwater (HydroQual et al. 2007), and zinc and freshwater (DeForest and Van Genderen 2012). The new data and methods used to develop these criteria were all based on SSDs, with

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[78] An SSD is a statistical model that can be used to calculate a chemical concentration protective of a predetermined percentage of a group of species. SSDs are intended to indicate both the total range and distribution of species sensitivities in natural communities, even when the actual range of sensitivities is unknown (Stephan 2002).

[79] In calculating an ACR, the acute value is the median LC50 (concentration that is lethal to 50% of an exposed population), and the chronic value is the no-observed-effect concentration (NOEC) or maximum allowable toxicant concentration (MATC) (i.e., the geometric mean of the NOEC and lowest-observed-effect-concentration [LOEC]) (Stephan et al. 1985).

the exception of copper and saltwater (Chadwick et al. 2008), which were based on the most sensitive endpoint evaluated.

3. If AWQC, or published values using the AWQC approach, were unavailable for a COPEC, and if data were sufficient, FAVs and FCVs were derived by developing SSDs in a manner consistent with the AWQC methodology outlined by Stephan et al. (1985), with some modifications.[80] The AWQC methodology and modifications used in this evaluation are described in Appendix D. Acute or chronic data for a minimum of five species were required to develop an acute or chronic SSD. When an SSD was developed using acute toxicity data, the FAV was the 5th percentile concentration of the best-fit distribution identified using @RISK, based on the Anderson-Darling (A-D) statistic. The FAV divided by two provided the acute TRV. When chronic toxicity data were sufficient (i.e., a minimum of five species) to develop an SSD, the FCV was the 5th percentile concentration for the best-fit distribution, which was selected as the chronic TRV. In most cases, insufficient chronic toxicity data were available, so the FCV was derived from the FAV using an appropriate ACR. As noted, USEPA uses a chemical-specific ACR to derive FCVs, but an ACR also may be identified based on chemical class or chemical mode of action. Consistent with AWQC methodology, if there was an acute or chronic toxicity value for a recreationally or commercially important species (e.g., rainbow trout [*Oncorhynchus mykiss*] or salmon), then the lowest toxicity value was selected as basis for the TRV.

4. When data were insufficient to develop a chronic or acute SSD (i.e., data were not available for at least five species), then the acute and chronic TRVs were based on the lowest toxicity values available from the literature. The acute TRV was selected as the lowest acute toxicity value divided by a factor of two. The chronic TRV was selected as the lowest chronic toxicity value (e.g., lowest LOEC). If chronic toxicity data were unavailable or unacceptable, then the chronic TRV was derived from the lowest acute toxicity value using an appropriate ACR.

For some COPECs, TRVs incorporated toxicity data using a biotic ligand model (BLM), a tool that can mechanistically predict the bioavailability of a variety of metals under the large range of water chemistry conditions that are observed in nature. The BLM approach considered the effect of water chemistry on metals toxicity, and TRVs were developed on a sample-specific basis. BLM-based models were used in the derivation of surface water TRVs for copper (freshwater and estuarine), lead (freshwater), silver (freshwater), and zinc (freshwater).

[80] The modifications to the AWQC methodology included the following: 1) use of @RISK software to model the distribution of available acute toxicity values and select the 5th percentile of the best-fit curve as the basis of the FCV, and 2) use of ACRs for organic chemicals obtained or derived from sources other than USEPA.

Toxicity Data Selection

The criteria for toxicity data selection were based on the AWQC methodology outlined by Stephan et al. (1985). For COPECs with freshwater and/or saltwater AWQC, aquatic toxicity data published at least two years prior to the published date of the AWQC document were exported from the ECOTOX database (USEPA 2016c). Data were limited to laboratory tests using saltwater and freshwater media and relevant test organisms (e.g., crustaceans, fish, daphnids, mollusks, worms). Toxicity endpoints were limited to growth, population, mortality, reproduction, and behavioral effect measurements.

Acute toxicity data were selected for further review for potential inclusion in TRV derivation if they met the following requirements:

- Measured effects were for growth, mortality, reproduction, and behavior.[81]
- Tests were conducted with any species except brine shrimp.[82]
- Data were from studies that used controls, and controls were not noted as unsatisfactory in the ECOTOX database.
- Results were reported as 96-hr LC50s for fish, bivalves, and other aquatic invertebrates. If LC50s were limited or not available, 96-hr EC50s were retained for review.
- Results were reported as 48- to 96-hr LC50s for daphnids/cladocerans.[83] If LC50s were limited or not available, 48- to 96-hr EC50s were retained for review.

Chronic toxicity data were reviewed using the following requirements:

- Measured effects were for growth, mortality, reproduction, and behavior.[84]
- Tests were conducted with any species except brine shrimp.
- Data were from studies that used controls, and controls were not noted as unsatisfactory in the ECOTOX database.

[81] Measured effect for behavior was not typically considered an acceptable standard endpoint for TRV development, but some toxicity data for behavioral effects that result in mortality (e.g., immobilization in fish) were considered on a case-by-case basis for inclusion in SSDs.

[82] The inclusion of all species was a modification from Stephan et al. (1985), which recommends only North American species; this modification was effected in order to augment the datasets for COPECs for which data were already limited.

[83] Exposures greater than 24 hrs with single-celled organisms were not considered acute exposures, and were included for further evaluation as chronic toxicity data.

[84] Measured effect for behavior was not typically considered an acceptable standard endpoint for TRV development, but some toxicity data for behavioral effects that result in mortality (e.g., immobilization in fish) were considered on a case by case basis for inclusion in SSDs.

- Data were from flow-through exposures for all test organisms; renewal exposures were considered acceptable for daphnids.
- Data were from studies with reported measured chemical analyses; data from studies for which target concentrations were reported based on nominal concentrations, rather than analytically measured, were retained for review if data were limited.
- Results represented NOEC/no-observed-effect level (NOEL) and LOEC/lowest-observed-effect level (LOEL) values; if data were limited, other effect levels, such as LC10/EC10 (concentration that is lethal to 10% of an exposed population/concentration that causes a non-lethal effect in 10% of an exposed population) and LC20/EC20 (concentration that is lethal to 20% of an exposed population/concentration that causes a non-lethal effect in 20% of an exposed population), were retained.
- Life cycle, partial life cycle, or early life stage tests were preferred for all test organisms, if test type was reported, with the following minimum exposure times: at least 24 days for fish species (90 days for salmonids) and 7 days for daphnids and mysids.

If data were extremely limited, then the alternative LC50s or EC50s for non-standard endpoints (e.g., behavior) or exposure types (e.g., sub-chronic) and studies with unmeasured chemistry or unsatisfactory controls were considered for inclusion.

TRV Uncertainty – TRVs Based on SSDs

Compared to a LOAEL based on a single study and test species, SSDs provide a measure of community sensitivity by incorporating not only toxicity data for many species, but also multiple toxicity values for the same species from different studies. The use of SSDs is also conservative, in that the SSD TRV tends to be selected from the lower tail of the SSD, most often the 5th percentile value. There are some uncertainties associated with TRVs derived from SSDs that should be considered in interpreting risk estimates based on SSD-derived TRVs, including the number of samples within the SSD, the suitability of the distribution used to fit the SSD (i.e., the best-fit model), and the application of ACRs.

The number of samples included in the SSD affects the reliability and stability of the derived TRV. As the number of samples increases, the stability of the TRV generally increases. This is particularly true of TRVs based on values selected from the lower tail of an SSD (e.g., 5th percentile); these TRVs are less than all but 5 or 10% of the effects data. Wheeler et al. (2002) and Newman et al. (2000) indicate that relatively sizable datasets (between 10 and 55 data points, depending on the distribution and spread of the data) are required for a low percentile TRV (e.g., 5th percentile) to be stable, regardless of the dataset from which the SSD was developed. Roman et al. (1999) conclude that when fewer than five data points are available to derive an SSD, TRVs based on the lowest value are more precise than those derived from the SSD approach, but that increasingly lower TRVs may be generated from the lowest value approach as the number of toxicity studies increases. Roman et al. (1999) also indicate that with five or more data points, the SSD approach yields a relatively stable value for the TRV, and as the amount of toxicity data used to develop the SSD increases, confidence in the reliability and protectiveness of the TRV also increases. Greater uncertainty exists in chronic TRVs estimated from SSDs based on fewer data points (i.e., acute SSDs for 4,4′-dichlorodiphenyldichloroethylene [DDE], anthracene, and BBP, each of which has fewer than 10 data points).

A common uncertainty associated with fitting a distribution to a concentration-effect dataset is the assumption that one specific distribution (e.g., lognormal) can be used to describe any dataset (Newman et al. 2000). Consideration of the various fit statistics provided by @RISK, along with visual examination of the curve and the values at the low end of the distribution, ensures that the most suitable best-fit model available is selected for estimating low-effects thresholds. The use of ACRs to estimate chronic TRVs from acute toxicity is also uncertain, although the uncertainty is reduced by the use of chemical- or mode of action-specific ACRs.

TRV Uncertainty – TRVs Based on Biotic Ligand Model

The TRVs for copper (freshwater and estuarine), lead (freshwater), silver (freshwater), and zinc (freshwater) are based on the BLM. Uncertainties associated with BLM-based freshwater TRVs for lead and zinc are expected to be minimal, because those BLMs have been shown to be highly capable of predicting effect concentrations for several fish and invertebrate species (DeForest and Van Genderen 2012; DeForest et al. 2017). In addition, methods consistent with USEPA guidelines for the derivation of AWQC (Stephan et al. 1985) were used to derive BLM-based benchmarks that are analogous to WQC (e.g., criterion maximum concentration [CMC] and criterion continuous concentration [CCC]) for lead and zinc (DeForest and Van Genderen 2012; DeForest et al. 2017).[85] As these benchmarks were derived with an up-to-date BLM, the TRVs used in this evaluation should be adequately protective of freshwater organisms.

The primary uncertainty related to using the BLM-based copper WQC (USEPA 2007b) for acute and chronic freshwater TRVs is that the chronic TRV is derived using an ACR. In the 2007 update to the copper WQC, there were insufficient chronic data to develop a chronic SSD, so the use of an ACR was necessary. Since 2007, additional chronic toxicity data have been collected for copper, so it has become possible to evaluate the protectiveness of the BLM-based chronic WQC (i.e., using an ACR of 3.22). To evaluate the protectiveness of the BLM-based chronic WQC, the BLM was applied to an updated toxicity dataset consisting of the most sensitive organisms included in the SSD used to derive the 2007 WQC. Among the 10 most sensitive genera, chronic data were available for 6 species (in order of acute sensitivity): *Daphnia*

[85] These BLM-based benchmarks are consistent with the level of protection intended by WQC, but they have not yet been adopted for use by USEPA.

ambigua (Harmon et al. 2003), *Daphnia pulex* (Winner 1985), *Daphnia magna* (Van Leeuwen et al. 1988; Muyssen and Janssen 2007; De Schamphelaere and Janssen 2004; Villavicencio et al. 2011), *Ceriodaphnia dubia* (Spehar and Fiandt 1986; Oris et al. 1991; Cerda and Olive 1993; Schwartz and Vigneault 2007; Cooper et al. 2009; Wang et al. 2011; Harmon et al. 2003), rainbow trout (Marr et al. 1996; Besser et al. 2001; Besser et al. 2005; McKim et al. 1978; Wang et al. 2014), and Chinook salmon (*Oncorhynchus tshawytscha*) (Chapman 1982, 1975). The BLM-based chronic WQC determined for each of the exposure conditions in the updated dataset was compared to the chronic effect concentrations reported for these studies. The comparison demonstrated that the approach is fully protective of the mean species values, making it consistent with USEPA's intended level of protection of WQC (Stephan et al. 1985). The ratios of reported effect concentration to predicted chronic WQC were > 1 for 115 of 118 observations.

The primary uncertainty related to using the BLM to derive copper TRVs in saltwater is that chronic toxicity data are limited. In the 2003 draft update to ambient copper WQC (USEPA 2003a), only one acceptable chronic value (368 µg/L for growth) and one ACR (1.48) for the relatively insensitive sheepshead minnow (*Cyprinodon variegatus*) were available. For comparison, the genus mean acute value for *Mytilus* was 11.53 µg/L. A recent chronic value reported for the rotifer *Brachionus plicatilis* was 7.9 µg/L (Arnold et al. 2011), and an ACR determined from the same study was 1.7. These results suggest that the ACR for saltwater copper toxicity is < 2. Given that the saltwater copper BLM was developed for *Mytilus galloprovincialis* (Chadwick et al. 2008), which represents the most sensitive genus in the acute SSD, and that the ACR is likely < 2, an acute TRV derived using *M. galloprovincialis* (i.e., BLM-predicted EC50 divided by two) is likely to be protective under both acute and chronic copper toxicity conditions.

6.2.3.2 Selected TRVs

This section discusses the surface water TRVs selected for surface water COPECs in the evaluation of risks to benthic invertebrates.

Table 6-14 presents the selected surface water TRVs and summarizes the general representativeness of the selected TRVs of invertebrate toxicity. Details on the selected TRVs are summarized in the subsections following Table 6-14. Additional details can be found in Appendix D.

Table 6-14. Surface water TRVs used in the evaluation of benthic invertebrates

COPEC	TRV Type	TRV (µg/L)[a] Acute	TRV (µg/L)[a] Chronic	TRV Derivation Method	Invertebrate Toxicity Relative to Selected TRV
Metals[b]					
Cadmium	estuarine	33	7.9	saltwater CMC and CCC USEPA (2015a); (USEPA 2016a)	TRVs are expected to be protective of invertebrates. Acute toxicity data included 16 fish and 78 invertebrate species, showing a wide range of sensitivity; the most sensitive species were invertebrates. The chronic TRV was derived from the FAV using an ACR (USEPA 2016a).
Cadmium	freshwater	1.4–6.5	0.59–2.0	freshwater CMC and CCC (USEPA 2016a); TRV ranges reflect range of mean sample-specific hardness values	Acute TRV may be conservative for invertebrates. Acute toxicity data included 66 invertebrate species, showing a wide range of sensitivities; the six most sensitive genera were fish. Chronic TRV is expected to be protective of invertebrates, as the two most sensitive genera included in the USEPA (2015a) chronic SSD dataset were invertebrates (USEPA 2016a).
Chromium	estuarine	1,100	50	saltwater CMC and CCC, as dissolved chromium(VI) (USEPA 2017c)	Documentation could not be found for the development of USEPA's current saltwater chromium AWQC, so its representativeness of invertebrate sensitivity cannot be confirmed. Based on results reported in standard methods, aquatic invertebrates appear to be more sensitive to chromium(VI) than fish (American Society for Testing and Materials et al. 1986).
Chromium	freshwater	16	11	freshwater CMC and CCC from USEPA (1996), converted to dissolved chromium using USEPA-recommended CF (USEPA 2017c)	TRVs are expected to be protective of invertebrates. USEPA (1996) included acute toxicity data for 17 fish and 17 invertebrate species; invertebrate species were generally the most sensitive to chromium (e.g., 10 most sensitive genera were invertebrates).

Table 6-14. Surface water TRVs used in the evaluation of benthic invertebrates

COPEC	TRV Type	TRV (µg/L)[a] Acute	TRV (µg/L)[a] Chronic	TRV Derivation Method	Invertebrate Toxicity Relative to Selected TRV
Copper	estuarine	0.98–11.2[c,d]	0.98–11.2[c,d,e]	near-bottom sample-specific CMC based on saltwater BLM developed for most sensitive species (Chadwick et al. 2008); CMC is assumed to be protective of chronic toxicity, so the acute and chronic TRVs are the same values	Acute TRV is expected to be protective of acute and chronic toxicity to invertebrates. The acute TRV for copper in saltwater is based upon the sensitivity of the invertebrate *Mytilus galloprovincialis*, which represents the most sensitive genus considered in USEPA (2003a). Chronic data for saltwater organisms are limited, and evaluation of potential ACRs indicate that acute criteria or TRVs based on early life stages of sensitive invertebrates would be protective of chronic toxicity.
	freshwater	14.3–76.1[c,d]	8.9–62.1[c,d]	near-bottom sample-specific CMC and CCC (using ACR) based on freshwater BLM from USEPA (2007f)	The acute TRV is expected to be protective of invertebrates. Acute toxicity data were considered for 38 species, with the 9 most sensitive genera represented by cladocerans, snails, amphipods, and freshwater mussels. The acute TRV is driven by the sensitivity of invertebrates, with the most sensitive fish being about 10-fold less sensitive than the most sensitive invertebrate. The chronic TRV was based on applying an ACR of 3.22 to the acute TRV. Given the relative acute sensitivity of fish and invertebrates, the chronic TRV is expected to be protective of fish (Appendix D).

Table 6-14. Surface water TRVs used in the evaluation of benthic invertebrates

COPEC	TRV Type	TRV (µg/L)[a] Acute	TRV (µg/L)[a] Chronic	TRV Derivation Method	Invertebrate Toxicity Relative to Selected TRV
Lead	estuarine	100	9.7	proposed acute and chronic saltwater criteria (Church et al. 2017) based on acute and chronic SSDs	TRVs are expected to be protective of invertebrates. Acute toxicity data are available for 54 species, and the 18 most acutely sensitive species are all invertebrates. As such, saltwater fish are relatively insensitive to lead. Chronic toxicity data are available for 21 species, 19 of which are invertebrates. The chronic TRV is driven by the sensitivity of an invertebrate (a mysid), which is about 5 times more sensitive than the most sensitive fish species tested to date (Appendix D).
Lead	freshwater	196–761[c,d]	7.5–35[c,d]	sample-specific CMC and CCC (using ACR) based on freshwater BLM (DeForest et al. 2017)	TRVs are expected to be protective of invertebrates. Acute toxicity data are available for 32 species, 21 of which are invertebrates. The 4 most acutely sensitive species are invertebrates and the most acutely sensitive fish species is about 1 order of magnitude less sensitive than the acute TRV. Chronic toxicity data are available for 15 species, 11 of which are invertebrates. TRV is driven by the sensitivity of an invertebrate (a snail) and the 7 most sensitive species tested to date are invertebrates (Appendix D).
Mercury	estuarine	1.8	0.94	saltwater CMC and CCC from USEPA (1984), converted to dissolved mercury using USEPA's metals-specific CF (USEPA 2017c)	TRVs are expected to be protective of invertebrates. Acute toxicity data from USEPA (1984) for fish and invertebrates showed a wide range of sensitivities to mercury, with the most sensitive species being invertebrates (Appendix D).
Mercury	freshwater	1.4	0.21	acute TRV is freshwater CMC from USEPA (1996), converted to dissolved mercury using USEPA's metals-specific CF (USEPA 2017c); chronic TRV is lowest LOEC from (USEPA 2016c)	Acute TRV is expected to be protective of invertebrates, as the CMC is based on the most sensitive invertebrate species. Chronic TRV may be conservative for invertebrates, as it is the lowest chronic toxicity value for a fish species (Appendix D).

Table 6-14. Surface water TRVs used in the evaluation of benthic invertebrates

COPEC	TRV Type	TRV (µg/L)[a] Acute	TRV (µg/L)[a] Chronic	TRV Derivation Method	Invertebrate Toxicity Relative to Selected TRV
Selenium	estuarine	290	71	saltwater AWQC (USEPA 2017c; U.S. Environmental Protection Agency 1999)	The representativeness of the estuarine TRVs cannot be evaluated because the source of the criteria does not indicate how the values were derived.
	freshwater	na	3.1	chronic TRV is dissolved selenium CCC in lotic waters (USEPA 2016b); no acute TRV selected for selenium	No invertebrate species were included in the SSD used to derive the chronic criterion. However, fish are the group most sensitive to selenium, so the chronic criterion is protective of invertebrates.
Silver	estuarine	5.54	2.0	5th percentile of saltwater SSD based on acute toxicity data divided by 2; chronic value derived using an ACR of 5.536	TRVs are expected to be protective of invertebrates. Toxicity data included 12 fish and 11 invertebrate species; invertebrate species were generally more sensitive than fish species (Appendix D).
	freshwater	1.8	0.69	acute and chronic values based on a proposed freshwater BLM from an unpublished report (HydroQual et al. 2007)	TRV is expected to be protective of invertebrates; BLM is based on both invertebrates and fish toxicity and accounts for influence of water quality characteristics (Appendix D).
Zinc	estuarine	75	19	acute TRV is the 5th percentile of an acute saltwater SSD divided by 2; chronic TRV is the 5th percentile of a chronic saltwater SSD	TRVs are expected to be protective of invertebrates. Toxicity data included 18 fish and 107 invertebrate species, showing a wide range of sensitivity among species (Appendix D).
	freshwater	210–1,660[c,d]	52–229[c,d]	sample-specific CCC and CMC based on freshwater BLM from DeForest and Van Genderen (2012)	TRVs are expected to be protective of invertebrates. Acute toxicity data were considered for 96 species, with the 10 most sensitive species representing cladocerans, fish, amphipods, and mussels. Chronic toxicity data were considered for 20 species, 10 of which were invertebrates. The most sensitive organism was an invertebrate (a water flea) (Appendix D).
Butyltins					
TBT	estuarine	0.42	0.066	USEPA-calculated saltwater FAV divided by 2 and FCV from USEPA (2003b)	TRVs are expected to be protective of invertebrates; toxicity data included in derivation of FCV from 26 invertebrate and 7 fish species indicate fish are less sensitive than some invertebrate species (USEPA 2003b).

Table 6-14. Surface water TRVs used in the evaluation of benthic invertebrates

COPEC	TRV Type	TRV (µg/L)[a]		TRV Derivation Method	Invertebrate Toxicity Relative to Selected TRV
		Acute	Chronic		
PCBs					
Total PCBs	estuarine	4.6	0.16	acute TRV is 5th percentile of saltwater SSD based on acute toxicity data, divided by 2; chronic TRV is lowest chronic LOEC (sheepshead minnow reproduction)[f]	Acute TRV is expected to be protective of invertebrates, as the acute toxicity data included only 1 fish and 10 invertebrate species. Chronic TRV may be overly conservative for invertebrates, as it is based on toxicity data from the most sensitive fish species (sheepshead minnow) (Appendix D),
	freshwater	1.2	0.27	acute TRV is 5th percentile of acute SSD based on toxicity data from USEPA (1980d) and USEPA (2016c); chronic TRV derived using an ACR of 8.4	TRVs may be conservative for invertebrates. Acute toxicity data included 15 fish and 10 invertebrate species. TRVs were based on the lowest SMAV, which was for a fish species (largemouth bass) (Appendix D).
PCDDs/PCDFs					
2,3,7,8-TCDD and TEQs - fish	estuarine	0.025	1.65×10^{-5}	chronic TRV is LOEC derived from Wintermyer and Cooper (2003) for reduced fertilization rates and increased egg mortality rates in eastern oyster (acute TRV selected for the benthic invertebrate assessment based on TRV derived from tests with a sensitive fish species (zebrafish)	Chronic TRV is expected to be protective of invertebrates (Appendix D). Acute TRV, which is based on fish toxicity, is likely conservative for invertebrates, which tend to be insensitive to dioxins (West et al. 1997).
	freshwater	0.0041	9.8×10^{-4}	acute TRV is lowest acute LC50 for a freshwater fish species (Japanese medaka); chronic TRV derived using an ACR of 8.3 from Raimondo et al. (2007)	TRVs likely conservative for invertebrates, as TRVs are based on toxicity data from the most sensitive fish species (Japanese medaka) (Appendix D). Invertebrates tend to be insensitive to dioxins (West et al. 1997).

Table 6-14. Surface water TRVs used in the evaluation of benthic invertebrates

COPEC	TRV Type	TRV (µg/L)[a]		TRV Derivation Method	Invertebrate Toxicity Relative to Selected TRV
		Acute	Chronic		
Organochlorine Pesticides					
4,4′-DDE	estuarine	1.25	0.30	acute TRV is lowest acute toxicity value for saltwater invertebrate species (*Nitocra spinipes*) divided by 2; chronic TRV is the lowest chronic toxicity value for the same species	TRVs are expected to be protective of invertebrates, as they are based on the lowest acute and chronic toxicity values available in USEPA (2016c), which were both for a copepod (*Nitocra spinipes*) (Appendix D).
	freshwater	2.40	1.40	acute TRV is 5th percentile of freshwater SSD based on acute toxicity data, divided by 2; chronic TRV derived using an ACR of 3.6 for DDT-type chemicals	TRVs are expected to be protective of invertebrates. Toxicity data included 3 fish and 2 invertebrate species. The chronic TRV is less than the lowest chronic toxicity value identified in USEPA (2016c) (Appendix D).
4,4′-DDT/total DDx	estuarine	0.034	0.019	acute TRV is 5th percentile of acute SSD based on saltwater data from USEPA (1980c) and USEPA (2016c), divided by 2; chronic TRV derived using an ACR for DDT-type chemicals	TRVs are expected to be protective of invertebrates. TRV represented by FCV that includes toxicity based on 14 fish and 18 invertebrate species, with invertebrates among the most sensitive species (USEPA 1980d).
	freshwater	0.45	0.25	acute TRV is 5th percentile of acute SSD based on freshwater data from USEPA (1980c) and USEPA (2016c), divided by 2; chronic TRV derived using an ACR for DDT-type chemicals	TRVs are expected to be protective of invertebrates. TRV based on FCV that incorporates toxicity data from 42 fish and 27 invertebrate species, with invertebrates among the most sensitive (USEPA 1980c).
Total chlordane	estuarine	0.045	0.0064	USEPA-calculated saltwater CMC and CCC from USEPA (1980b)	TRVs are expected to be protective of invertebrates. TRV is based on FCV that incorporates toxicity data for 4 invertebrate and 4 fish species. Chronic toxicity data indicate that fish are less sensitive than invertebrates (USEPA 1980b).
	freshwater	1.2	0.17	USEPA-calculated freshwater CMC and CCC from USEPA (1980b)	TRVs are expected to be protective of invertebrates. TRV is based on FCV that incorporates toxicity data for 4 invertebrate and 9 fish species. Chronic toxicity data indicate that freshwater invertebrates are more sensitive that fish (USEPA 1980b).

Table 6-14. Surface water TRVs used in the evaluation of benthic invertebrates

COPEC	TRV Type	TRV (µg/L)[a] Acute	Chronic	TRV Derivation Method	Invertebrate Toxicity Relative to Selected TRV
Dieldrin	estuarine	0.36	0.084	USEPA-calculated saltwater CMC and CCC from USEPA (1980a)	TRV may be overly conservative for fish; TRV is based on FCV that incorporates toxicity data from 8 invertebrate and 13 fish species. Chronic toxicity data indicate that freshwater invertebrates are more sensitive than fish (USEPA 1980a).
Hexachlorobenzene	saltwater	71	23	lowest acute LC50 for a saltwater species (common sole) divided by 2; chronic value derived using an ACR	Toxicity data are limited for saltwater species. TRVs are conservative for invertebrates, as both are based on the lowest acute toxicity value for a fish species (Appendix D).
	freshwater	180	57	5th percentile of freshwater SSD based on acute toxicity data; chronic value derived using an ACR	Toxicity data are limited for freshwater invertebrate species. TRVs may be conservative for invertebrates, as they are based on acute toxicity data for 9 fish species (Appendix D).
PAHs					
Anthracene	estuarine	34.5	13.5	acute TRV is lowest acute LC50 for a saltwater species (dwarf surf clam) divided by 2; chronic TRV derived using an ACR of 5.09 from DiToro et al. (2000)	TRVs are expected to be protective of invertebrates; no acceptable chronic toxicity data were available (Appendix D).
	freshwater	0.26	0.10	acute TRV is 5th percentile of freshwater SSD based on acute toxicity data divided by 2; chronic value derived using an ACR of 5.09	TRVs may be conservative for invertebrates. Acute toxicity data included 4 fish and 2 invertebrate species, with fish species being the most sensitive (Appendix D).
Benzo(a)anthracene	estuarine	0.48	0.19	same as freshwater TRVs [f]	same as freshwater TRVs
	freshwater	0.48	0.19	acute TRV is lowest acute LC50 for a freshwater species (*Daphnia magna*); chronic TRV derived using an ACR of 5.09	TRVs expected be protective of invertebrates (Appendix D).

Table 6-14. Surface water TRVs used in the evaluation of benthic invertebrates

COPEC	TRV Type	TRV (µg/L)[a] Acute	Chronic	TRV Derivation Method	Invertebrate Toxicity Relative to Selected TRV
Benzo(a)pyrene	estuarine	0.51	0.20	acute TRV is lowest acute LC50 for *Daphnia magna*, divided by 2; chronic TRV derived using an ACR of 5.09	TRVs expected be protective of invertebrates (Appendix D).
	freshwater	2.03	0.80	acute TRV is 5th percentile of freshwater SSD based on acute toxicity data divided by 2; chronic TRV derived using an ACR of 5.09	TRVs expected be protective of invertebrates. SSD based on acute toxicity data for 2 fish and 3 invertebrate species; data indicate that freshwater invertebrates are more sensitive than fish (Appendix D).
Fluoranthene	estuarine	3.02	1.19	acute TRV is 5th percentile of saltwater SSD based on acute toxicity data, divided by 2; chronic TRV derived using an ACR of 5.09	TRVs expected be protective of invertebrates. Acute toxicity data included only 3 fish and 16 invertebrate species, with invertebrate species being among the most sensitive (Appendix D).
	freshwater	13.2	5.20	acute TRV is 5th percentile of freshwater SSD based on acute toxicity data; chronic TRV derived using an ACR of 5.09	TRVs are expected to be protective of invertebrates. Acute toxicity data included 4 fish and 10 invertebrate species. TRVs are less than the lowest fish and invertebrate SMAVs (Appendix D).
Pyrene	estuarine	0.46	0.18	acute TRV is lowest acute EC50 for a saltwater species (dwarf surf clam) divided by 2; chronic TRV derived using an ACR of 5.09	TRVs expected be protective of invertebrates (Appendix D).
	freshwater	2.2	0.84	acute TRV is lowest acute EC50 for a freshwater species (*Daphnia magna*); chronic TRV derived using an ACR of 5.09	TRVs expected be protective of invertebrates (Appendix D).

Table 6-14. Surface water TRVs used in the evaluation of benthic invertebrates

COPEC	TRV Type	TRV (µg/L)[a] Acute	TRV (µg/L)[a] Chronic	TRV Derivation Method	Invertebrate Toxicity Relative to Selected TRV
SVOCs					
BEHP	estuarine	500	100	acute TRV is lowest LC50 divided by 2; chronic TRV derived using an ACR of 6.9 based on DeFoe et al. (1990)	Acute TRV is expected be protective of invertebrates (Appendix D); chronic TRV may be conservative for invertebrates because it is based on the lowest available chronic value for a fish species (Appendix D).
	freshwater	24.1	7.0	acute TRV is 5th percentile of freshwater SSD based on acute toxicity data; chronic TRV derived using an ACR of 6.9	TRVs may be conservative for invertebrates. Acute SSD includes toxicity based on 12 fish and 4 invertebrate species, with multiple fish species being among the most sensitive (Appendix D).
BBP	estuarine	245	71	acute TRV is lowest acute LC50 for a saltwater species (shiner perch); chronic TRV derived using an ACR of 6.9.	TRVs may be conservative for invertebrates, as both are derived from the lowest acute toxicity value for a fish species (shiner perch) (Appendix D).
	freshwater	107	30.9	acute TRV is 5th percentile of freshwater SSD based on acute toxicity data; chronic TRV derived using an ACR of 6.9.	TRVs may be conservative for invertebrates. SSD includes toxicity data based on 4 fish and 4 invertebrate species, with fish species being among the most sensitive (Appendix D).
Other					
Cyanide	estuarine	6.1	1.9	acute TRV is 5th percentile of SSD based on acute toxicity data of 13 invertebrate and 3 fish species; chronic TRV derived using an ACR of 8.6 from Gensemer et al. (2006)	TRVs are expected to be protective of invertebrates. Acute toxicity data included in SSD indicate invertebrates may be more sensitive than fish (Appendix D).
	freshwater	32.3	7.5	acute TRV is 5th percentile of SSD based on acute toxicity data of 24 invertebrate and 11 fish species; chronic TRV derived using an ACR of 6.5 from Gensemer et al. (2006)	TRVs may be conservative for invertebrates. Acute toxicity data show low range of values in SSD based on fish toxicity, with less invertebrate sensitivity (Appendix D).

a NOAEL TRVs were not developed for surface water; SSD-derived 5th percentile TRVs were based on effects levels from the literature.

b TRVs for metals are based on the dissolved chemical form.

[c] For COPECs with BLM-based TRVs, the distinction between freshwater and saltwater was based on 3.5 ppth salinity.

[d] As they are sample specific, the BLM-based TRVs are a range of values (i.e., each individual sample has a corresponding BLM-based TRV).

[e] Due to lack of chronic copper toxicity data for saltwater species, the sample-specific acute BLM-based TRVs were also used as the chronic TRVs.

[f] The freshwater TRVs for benzo(a)anthracene were selected as surrogate estuarine TRVs due to lack of saltwater toxicity data.

ACR – acute-to-chronic ratio
AWQC – ambient water quality criteria
BBP – butyl benzyl phthalate
BEHP – bis(2-ethylhexyl) phthalate
BLM – biotic ligand model
CCC – criterion continuous concentration
CMC – criterion maximum concentration
COPEC – chemical of potential ecological concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
EC50 – concentration that causes a non-lethal effect in 50% of an exposed population
FAV – final acute value
FCV – final chronic value
LC50 – concentration that is lethal to 50% of an exposed population
LOEC – lowest-observed-effect concentration
na – not applicable
NOAEL – no-observed-adverse-effect level
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
PCDD – polychlorinated dibenzo-*p*-dioxin
PCDF – polychlorinated dibenzofuran
ppth – parts per thousand
SMAV – species mean acute value
SSD – species sensitivity distribution
SVOC – semivolatile organic compound
TBT – tributyltin
TCDD – tetrachlorodibenzo-*p*-dioxin
TEQ – toxic equivalent
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT and 4,4′-DDT)
TRV – toxicity reference value
USEPA – US Environmental Protection Agency

Cadmium

The selected estuarine and freshwater TRVs were based on the draft CMC and CCC for dissolved cadmium, as presented in USEPA (2015a).

The draft hardness-dependent freshwater CMC of 2.1 µg/L and CCC of 0.73 µg/L were normalized to a hardness of 100 mg/L as calcium carbonate. The draft AWQC document presents the freshwater CMC and CCC at various hardnesses, ranging from 25 to 400 mg/L as calcium carbonate. The hardness in the LPRSA ranges from 76.9 to 3,510 mg/L as calcium carbonate, with an average of 490 mg/L. Because of the variable hardness concentrations in the LPRSA, location-specific acute and chronic freshwater TRVs were calculated (Appendix D). Cadmium toxicity decreases with increased hardness (USEPA 2016a), so for LPRSA locations with average hardness concentrations greater than 400 mg/L as calcium carbonate (Appendix D), the CMC and CCC values were adjusted to 400 mg/L as calcium carbonate to be sufficiently conservative. Acute and chronic TRVs for dissolved cadmium ranged from 1.4 to 6.5 µg/L and 0.59 to 2.0 µg/L, respectively.

The saltwater CMC was derived by dividing the FAV of 66.25 µg/L by 2. Given that the chronic toxicity dataset represented only two saltwater species in one genus, the CCC was derived by dividing the FAV by an ACR of 8.291, which was based on the geometric mean of seven genus-level ACRs for one saltwater invertebrate species, two freshwater invertebrate species, and four freshwater fish species. The criteria for dissolved cadmium were calculated using a CF 0.994. The resulting CMC of 33 µg/L and CCC of 7.9 µg/L were selected as the acute and chronic saltwater TRV, respectively.

Chromium

The freshwater AWQC for chromium(VI), which were last updated by USEPA in 1995, were selected as the basis for the acute and chronic TRVs (USEPA 1996). The CMC and CCC for dissolved chromium—16 and 11 µg/L, respectively—were selected as the acute and chronic TRVs, respectively. The CMC and CCC were derived from acute toxicity data for 28 freshwater genera. Invertebrate species are generally more sensitive to chromium than are fish species, and cladocerans are the most sensitive invertebrate group. The CCC was calculated from the CMC with an ACR of 2.917. USEPA determined the CMC and CCC to be sufficiently protective of commercially or recreationally important fish or invertebrate species.

The saltwater AWQC for chromium(VI) were selected as the basis for the acute and chronic BERA TRVs. The CMC and CCC derived for saltwater species were 1,100 µg/L and 50 µg/L, respectively (USEPA 2017c). The CMC was derived from acute toxicity data for 12 marine invertebrate species, and the CCC was based on life cycle or partial life cycle data for 3 marine worm species.

Copper

The freshwater AWQC for copper, based on the BLM, were selected as the basis for the acute and chronic TRVs (USEPA 2007b). The BLM approach considers the effect of water chemistry on copper toxicity (Di Toro et al. 2001; Santore et al. 2001), so TRVs were developed on a sample-specific basis.

Sample-specific water chemistry was used to calculate acute and chronic copper TRVs in the freshwater segment of the LPRSA. For samples with missing major ion data, BLM inputs were estimated from regressions with salinity; for samples with missing pH and temperature data, TRVs were calculated using the event-, station-, and depth-specific minimum, maximum, and mean values. For near-bottom surface water samples, the ranges of acute and chronic TRVs were 14.3 to 76.1 µg/L and 8.9 to 47.2 µg/L, respectively; these ranges were used to evaluate risks to benthic invertebrates from surface water exposure.

A saltwater BLM, described in Chadwick et al. (2008), was developed to predict the toxicity of copper to sensitive larval stages of several marine invertebrates. The most sensitive endpoint evaluated was the larval development of *M. galloprovincialis*. In addition to evaluating larval development EC50s (concentrations that causes a non-lethal effect in 50% of an exposed population), copper accumulation was also reported. The saltwater BLM was able to characterize the observed copper accumulation and predict effect concentrations. Because the saltwater BLM was capable of predicting effect concentrations for *M. galloprovincialis*, it was selected as the basis for the acute and chronic TRVs. Sample-specific water chemistry was used to calculate saltwater TRVs in the saltwater segment of the LPRSA. Inputs for the saltwater BLM included salinity, dissolved organic carbon (DOC), pH, and temperature, but if individual ion measurements were available, they could be used in place of salinity. For samples with missing pH and temperature data, TRVs were calculated using the event-, station-, and depth-specific minimum, maximum, and mean values. For near-bottom surface water samples, the range of acute/chronic TRVs was 1.0 to 11.2 µg/L, which is used to evaluate risks of surface water exposure to benthic invertebrates.

Lead

DeForest et al. (2017) recently developed a freshwater lead BLM to derive site-specific and time-variable criteria; this BLM was used to calculate sample-specific lead TRVs for LPRSA freshwater samples. The freshwater lead BLM works similarly to the freshwater copper BLM. For near-bottom surface water samples, acute and chronic TRVs ranged from 196 to 761 µg/L and from 7.5 to 35 µg/L, respectively. BLM-based freshwater TRVs are compared to only surface water lead concentrations at stations with salinity < 3.5 ppth; all other stations are compared to the saltwater lead criteria (described in the following section). Because BLM-based TRVs are heavily influenced by water quality parameters (i.e., model inputs), it was decided that it was more appropriate to apply sample-specific TRVs to "freshwater" LPRSA locations as defined by measured salinities, rather than as by samples within generalized salinity zones (as was done for non-BLM-based TRVs).

Church et al. (2017) recently developed updated saltwater lead criteria based on acute and chronic saltwater toxicity data from US Environmental Protection Agency (1985a) and USEPA (2013a), as well as data provided by the International Lead Association (formerly the International Lead Zinc Research Organization) and obtained through literature searches. Criteria were developed following USEPA methods (Stephan et al. 1985) (with minor deviations such as the inclusion of non-North American species). The acute (100 µg/L) and chronic (9.7 µg/L) criteria proposed by Church et al. (2017) were selected as the acute and chronic saltwater TRVs.

Mercury

The AWQC for total recoverable mercury, which were last updated by USEPA in 1995, were selected as the basis for the freshwater acute and chronic TRVs (USEPA 1996). The freshwater CMC and CCC for total recoverable mercury are 1.7 and 0.91 µg/L, respectively. The CMC was based on an FAV of 3.388 µg/L divided by two. The CCC was derived from the FAV using an ACR of 3.731, which was based on the geometric mean of ACRs for two invertebrate species. USEPA (1996) concluded that the freshwater CCC may not be adequately protective of important fish species due to estimated chronic values for rainbow trout, coho salmon (*Oncorhynchus kisutch*), and bluegill that were more than a factor of two less than the CCC. The lowest estimated chronic value in USEPA (1996) is 0.25 µg/L for bluegill (based on an acute value of 160 µg/L and an ACR of > 646.2 for fathead minnow [*Pimephles promelas*]), which is similar to the lowest acceptable chronic toxicity identified in USEPA (2016c), a 30-day growth LOEC of 0.23 µg/L for early life stage fathead minnow. As such, the LOEC of 0.23 µg/L was selected as the basis of the chronic freshwater TRV. Using USEPA's CF of 0.85 for mercury, the CMC and LOEC were converted to the dissolved mercury criteria of 1.4 and 0.21 µg/L, which were selected as the freshwater acute and chronic TRVs, respectively.

The selected saltwater TRVs were based on the saltwater CMC (2.1 µg/L) and CCC (1.1 µg/L) for mercury derived in USEPA (1984). The CMC was derived from an FAV of 4.125 µg/L, based on acute toxicity data for 29 genera of saltwater fish and invertebrates. The CCC was derived from the FAV using the ACR of 3.731. Using USEPA's CF of 0.85, the CMC and CCC were converted to dissolved mercury concentrations, providing the acute TRV of 1.8 µg/L and the chronic TRV of 0.94 µg/L. Both of these values are less than the lowest acceptable toxicity values identified in USEPA (2016c), and as such were determined to be appropriately conservative for this BERA.

Selenium

USEPA's revised freshwater aquatic life chronic AWQC of 3.1 µg/L was selected as the chronic TRV (USEPA 2016b). An acute criterion has not been selected by USEPA due the limited acute aquatic toxicity data available for selenium, and the protectiveness of the chronic criterion pertaining to bioaccumulation and reproductive toxicity (USEPA 2016b).

The acute and chronic saltwater AWQC for selenium—290 and 71 µg/L, respectively (USEPA 2017c; EPA 1987)—were selected as the acute and chronic BERA TRVs, respectively. USEPA derived a CMC (294 µg/L, later rounded to 290 µg/L) from acute toxicity data for 15 saltwater species, and the CCC was calculated using an ACR of 8.314.

Silver

The acute and chronic freshwater TRVs for silver (1.8 and 0.69 µg/L, respectively) are based on an unpublished proposed BLM developed by HydroQual et al. (2007) that uses an assumed DOC concentration of 2 mg/L, which is similar to the 10th percentile DOC concentration of the LPRSA. These BLM-based TRVs are better estimates of acute and chronic low-effect levels than a chronic value derived from an SSD based on acute toxicity data from the literature that are unadjusted for water quality characteristics (Appendix D). The use of the proposed BLM-based TRVs is more consistent with recent USEPA guidance, and more appropriately accounts for the influence of water quality characteristics on silver bioavailability to freshwater organisms and toxicity.

No BLM is available for silver in saltwater. As such, the saltwater acute and chronic TRVs of 5.5 and 2.0 µg/L, respectively, were derived from an SSD based on acute toxicity data for fish and invertebrate species (Figure 6-5; Appendix D). A saltwater ACR of 5.536 was selected from HydroQual et al. (2007) to derive 11.1 µg/L, the chronic TRV from the 5th percentile of the best-fit distribution.

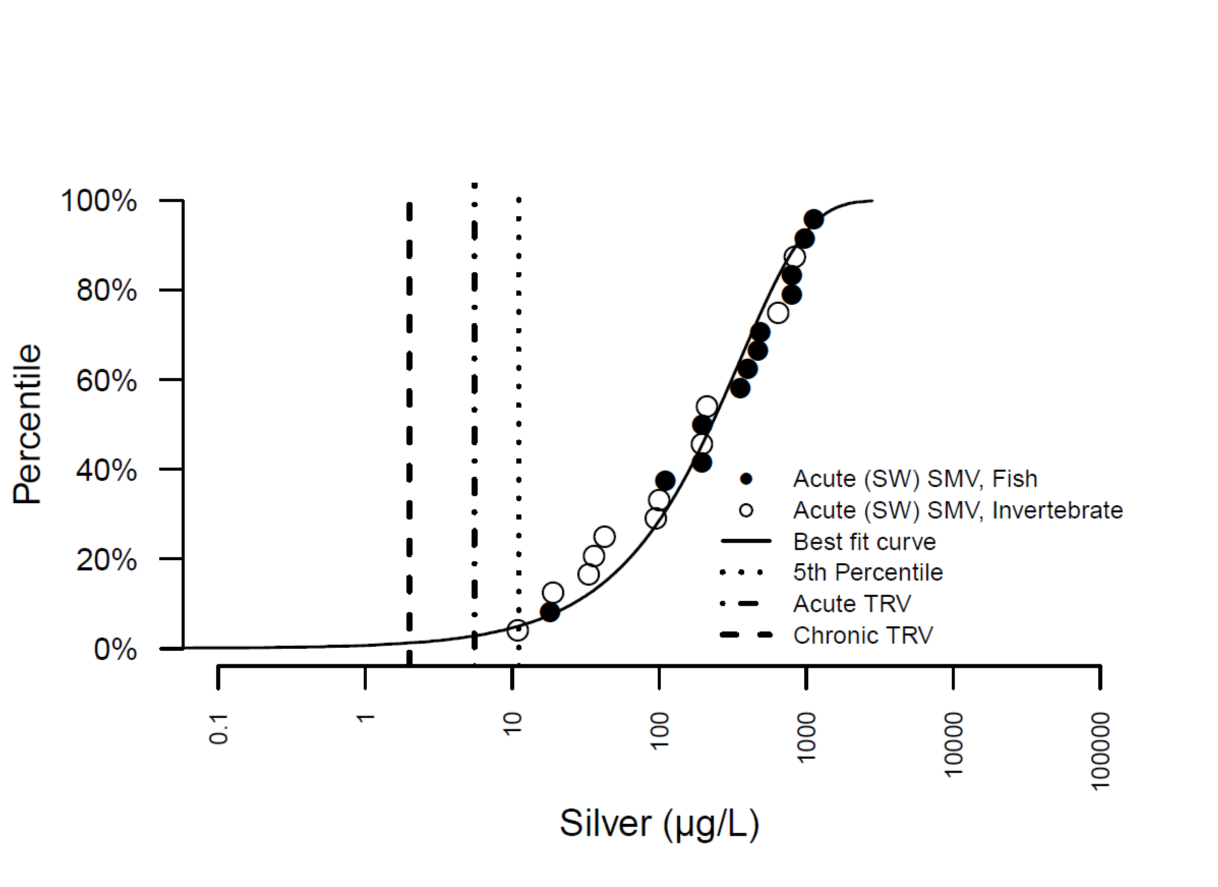


Figure 6-5. Acute saltwater SSD for silver

Zinc

The unified zinc BLM, described by DeForest and Van Genderen (2012), was selected as the basis for acute and chronic freshwater TRVs. This version of the zinc BLM uses a single set of biotic ligand parameters to characterize the effects of water chemistry on zinc toxicity to a variety of freshwater organisms (including invertebrates and fish) under both acute and chronic exposures. In addition to deriving a version of the zinc BLM that was highly predictive of acute and chronic zinc toxicity, DeForest and Van Genderen (2012) applied USEPA methods (US Environmental Protection Agency 1985b) to evaluate the 5th percentiles of BLM-normalized acute and chronic SSDs that had been updated with recent toxicity data. The BLM critical accumulations (i.e., sensitivity parameters) associated with the 5th percentiles of the BLM-normalized acute and chronic SSDs can be used to calculate BLM-predicted effect concentrations that are analogous to FAVs and FCVs. To derive sample-specific TRVs, the unified zinc BLM was applied to calculate values analogous to the CMC and CCC, wherein the CMC was the 5th percentile of the BLM-normalized acute SSD divided by two, and the CCC was the 5th percentile of the BLM-normalized chronic SSD.

Sample-specific zinc TRVs were calculated in the freshwater segment of the LPRSA. For samples with missing major ion data, BLM inputs were estimated from regressions with salinity, and for samples with missing pH and temperature data, TRVs were calculated using the event-, station-, and depth-specific minimum, maximum, and mean values. For near-bottom surface water samples, acute and chronic TRVs ranged from 210 to 1,660 µg/L and from 52 to 229 µg/L, respectively.

The acute and chronic saltwater TRVs of 75 and 19 µg/L, respectively, were based on acute and chronic SSDs (Appendix D). The acute SSD is shown in Figure 6-6. Since chronic toxicity data were sufficient for an SSD, the 5th percentile value of the distribution was selected as the chronic TRV, rather than using an ACR to derive it from the 5th percentile of an acute SSD (Figure 6-7).

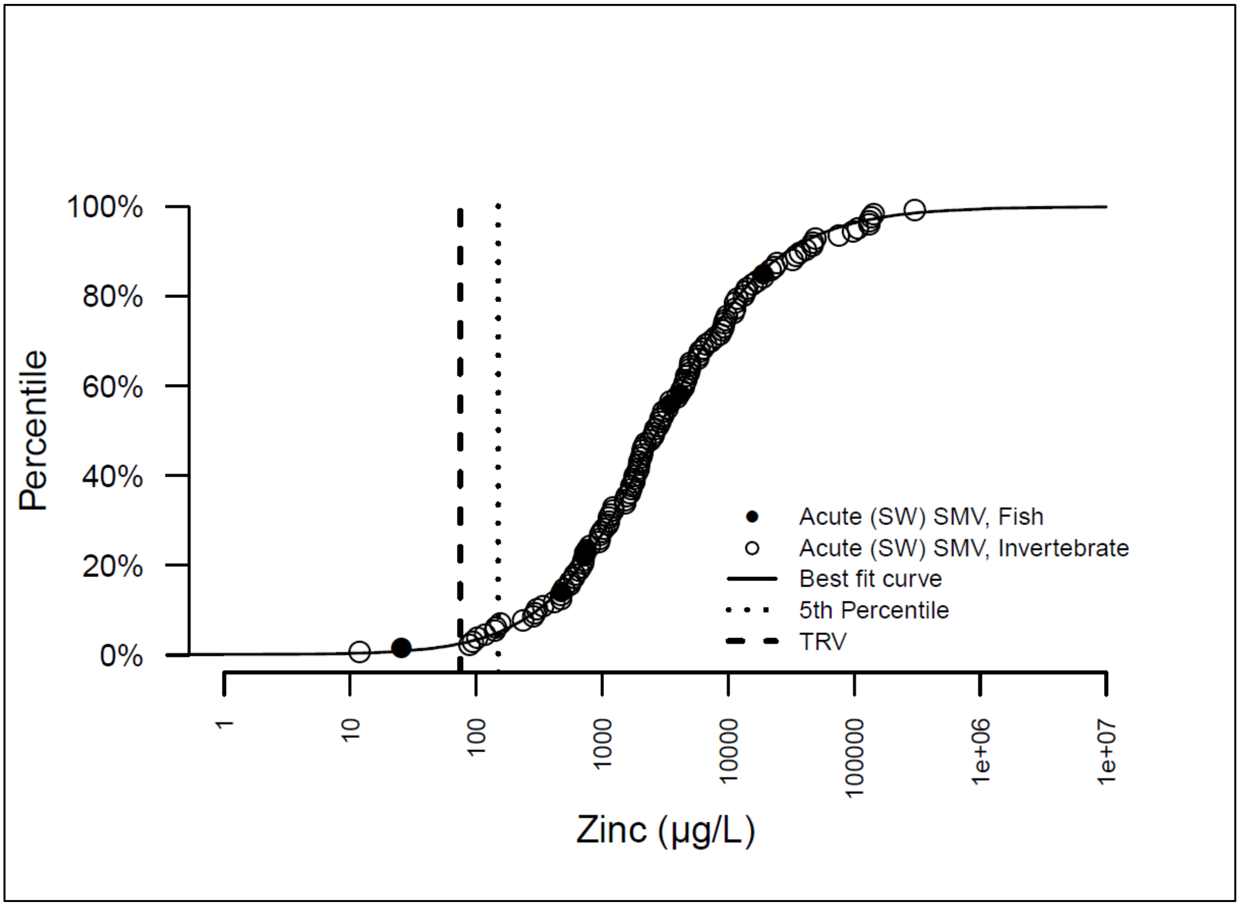


Figure 6-6. Acute saltwater SSD for zinc

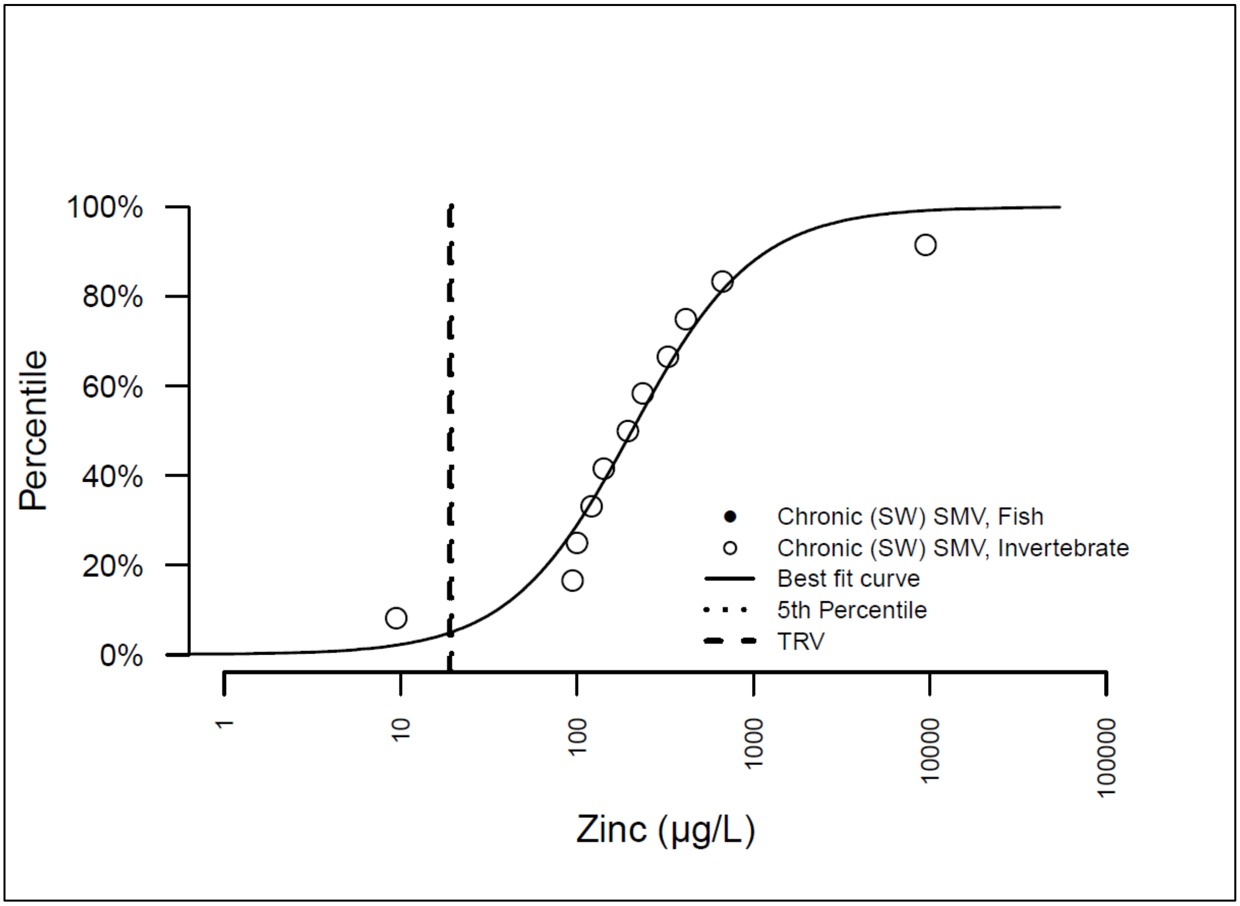


Figure 6-7.Chronic saltwater SSD for zinc

Tributyltin

The acute and chronic saltwater TRVs for TBT (0.42 and 0.066 µg/L, respectively) are based on the AWQC FAV (USEPA 2003b). The FAV is based on the direct, acute toxicity of the TBT cation to 26 invertebrate and 7 fish species; an ACR of 12.69 was used to derive the FCV. This ACR is the geometric mean of ACRs for three invertebrate (freshwater and saltwater) species and one freshwater fish species. The similarity of ACRs derived by USEPA (2003b) indicates that salinity has little influence on toxicity of TBT. The acute data used in the calculation of the FAV indicate that saltwater invertebrate species are more sensitive to TBT than are fish. The limited acceptable chronic data reported by USEPA (2003b) indicate that saltwater invertebrate species, such as the copepod *Eurytemora affinis* and the mysid *Acanthomysis sculpta*, are more sensitive to TBT than are freshwater invertebrates (e.g., *D. magna*), but the chronic values for these species are greater than 0.0658 µg/L. The only saltwater species with a LOEC less than 0.0658 µg/L is the Atlantic dogwinkle or dog whelk, *Nucella lapillus*. This species is known to be exceptionally sensitive to TBT at very low concentrations; the lowest chronic NOAEL reported for this species was the basis for lowering the published saltwater AWQC to 0.0074 µg/L. However, 0.0074 µg/L is an overly protective value for evaluating the effects of direct toxicity on benthic invertebrates, zooplankton, and fish in the estuarine portion of the LPRSA. Thus, the FAV and FCV were selected as the basis for the acute and chronic saltwater TRVs (Appendix D).

PCBs

Freshwater acute fish and invertebrate toxicity data for total PCBs were sufficient to develop an FCV following USEPA methods with additional recent toxicity data. The lowest species mean acute value (SMAV) (2.3 µg/L) was selected as the acute value used to derive a chronic TRV because it was less than the 5th percentile concentration of the acute SSD (4.5 µg/L) based on 10 invertebrate and 15 fish species (Figure 6-8; Appendix D). The acute freshwater TRV for total PCBs (1.2 µg/L) was based on the lowest SMAV for largemouth bass (2.3 µg/L). The chronic TRV (0.27 µg/L) was derived from the 5th percentile of the SSD using an ACR of 8.4 from USEPA (1980d). The ACR is based the geometric mean of ACRs for an amphipod and fathead minnow.

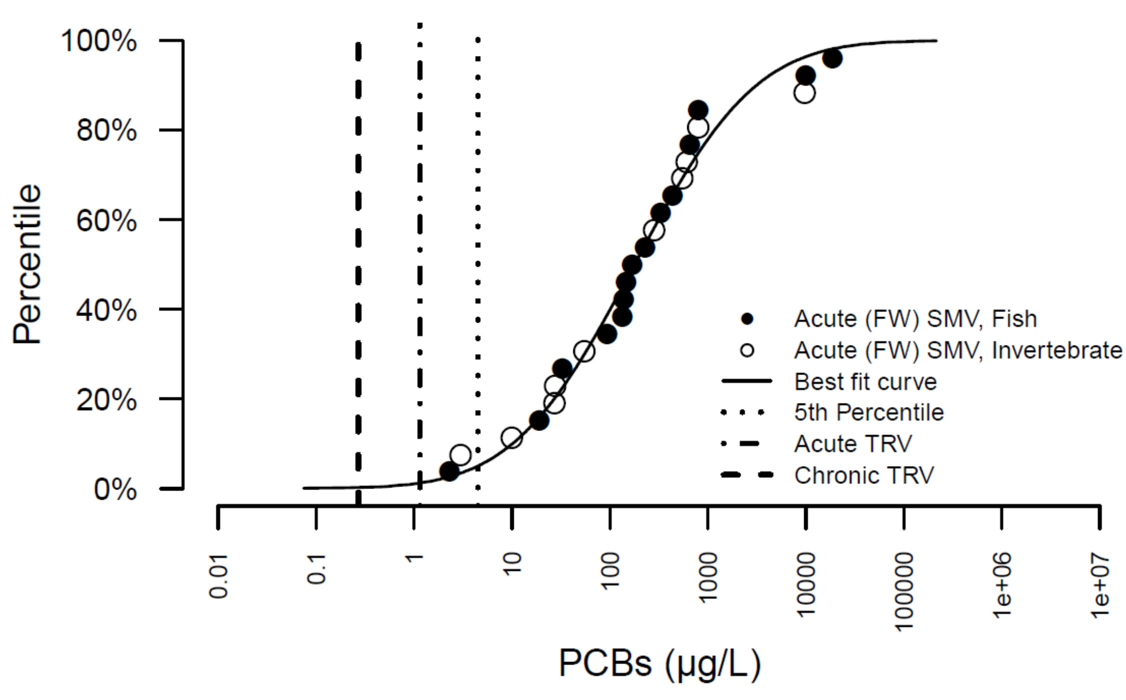


Figure 6-8. Acute freshwater SSD for PCBs

The acute saltwater TRV for total PCBs (4.6 µg/L) was derived from the 5th percentile of an SSD based on acute toxicity data for 10 invertebrate and 1 fish species (Figure 6-9). The chronic saltwater TRV (0.16 µg/L) was based on the lowest acceptable LOEC, which corresponds to reduced fry survival following flow-through exposure of early life stage sheepshead minnow to Aroclor 1254 for 28 days (Schimmel et al. 1974). This chronic value was selected because it was less than the calculated chronic value (1.1 µg/L) derived from a 5th percentile concentration (9.1 µg/L) of an acute SSD using an ACR of 8.4 (Appendix D).

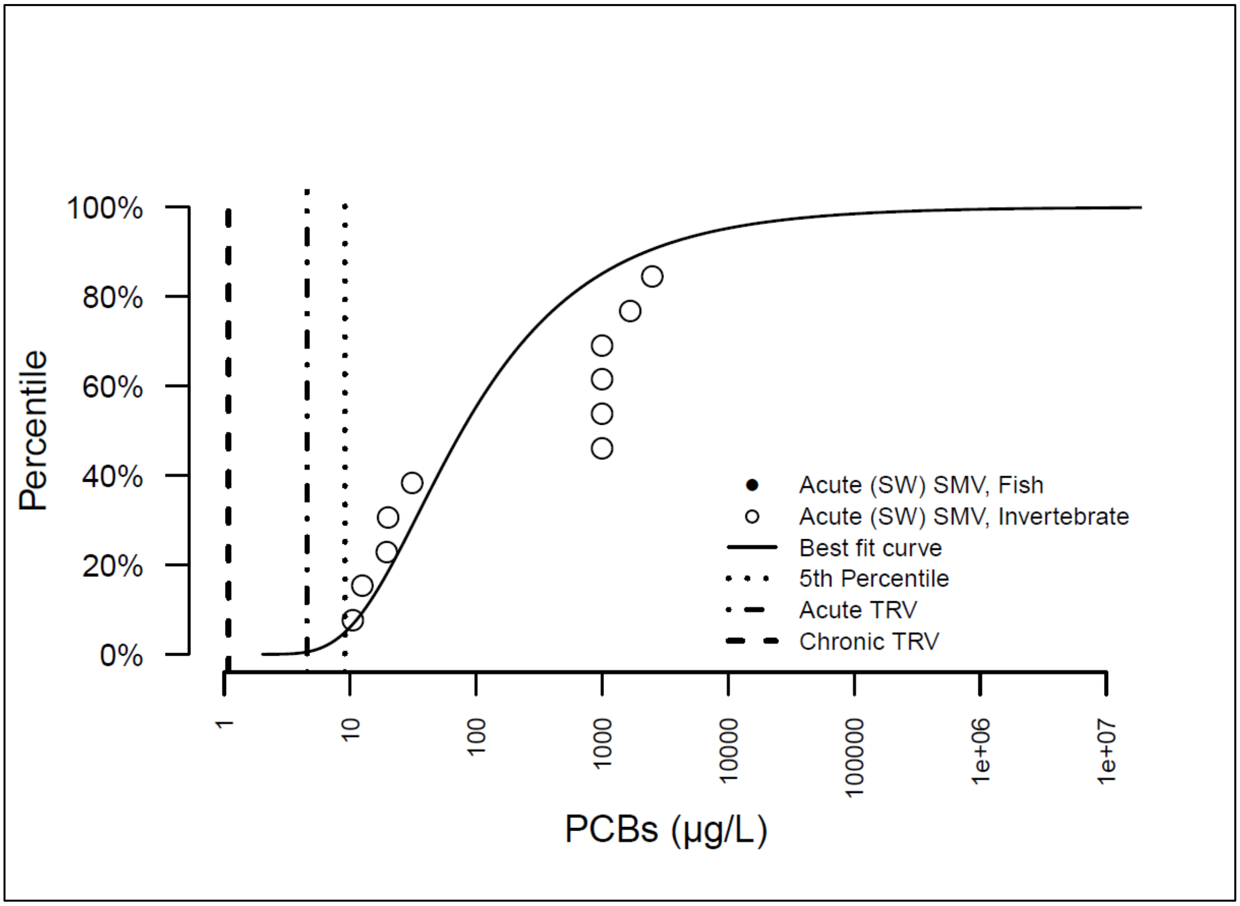


Figure 6-9. Acute saltwater SSD for PCBs

PCDDs/PCDFs

Toxicity data for 2,3,7,8-TCDD from the literature were limited to freshwater and saltwater fish species, and were insufficient for developing either acute or a chronic SSDs. As such, the acute and chronic TRVs were based on the lowest chronic toxicity values available from USEPA (2016c) for freshwater and saltwater species.

The lowest acute toxicity value for a freshwater species was a 72-hr LC50 of 0.0081 µg/L for early life stage Japanese medaka (*Oryzias latipes)* from Kim and Cooper (1999); this value was divided by 2 to provide the acute TRV of 0.0041 µg/L for freshwater. The lowest chronic toxicity value was a 28-day early life stage static renewal LC50 of 0.0017 µg/L for fathead minnow (Adams et al. 1986). Since applying an ACR of 8.3

derived by Raimondo et al. (2007) to the acute LC50 provided a lower chronic value of 9.8×10^{-4} µg/L, this value was selected as the chronic freshwater TRV.[86]

Since USEPA guidelines (Stephan et al. 1985) consider 48-hr toxicity data from embryo-larval growth and development tests with mollusks to be representative of chronic toxicity for this type of invertebrate, the LOEC of 1.65×10^{-5} µg/L from Wintermyer and Cooper (2003) was selected as the chronic estuarine TRV for the benthic assessment. An acute benthic TRV of 0.025 µg/L was selected from sensitive fish toxicity test data (14-day early life stage LC50 for zebrafish [*Danio rerio*] [Appendix D]). This value is expected to be conservative for characterizing invertebrate risk, because invertebrates lack an Ah receptor, effectively reducing their sensitivity to dioxins and dioxin-like chemicals (relative to fish, for example).

4,4′-DDE

The acute and chronic freshwater TRVs for 4,4′-DDE (2.4 and 1.4 µg/L, respectively) are based on an SSD for two invertebrate and three fish species (Figure 6-10). The chronic TRV was derived using an ACR of 3.6 for DDT-type chemicals from Raimondo et al. (2007). The ACR is the median ACR reported by Raimondo et al. (2007) in four studies of chemicals with a DDT-like mode of action (ACRs ranged from 3 to 5). The chronic TRV is less than the lowest chronic toxicity value from USEPA (2016c).

[86] The ACR of 8.3 is the overall median value of 456 aquatic invertebrate and fish ACRs analyzed by Raimondo et al. (2007).

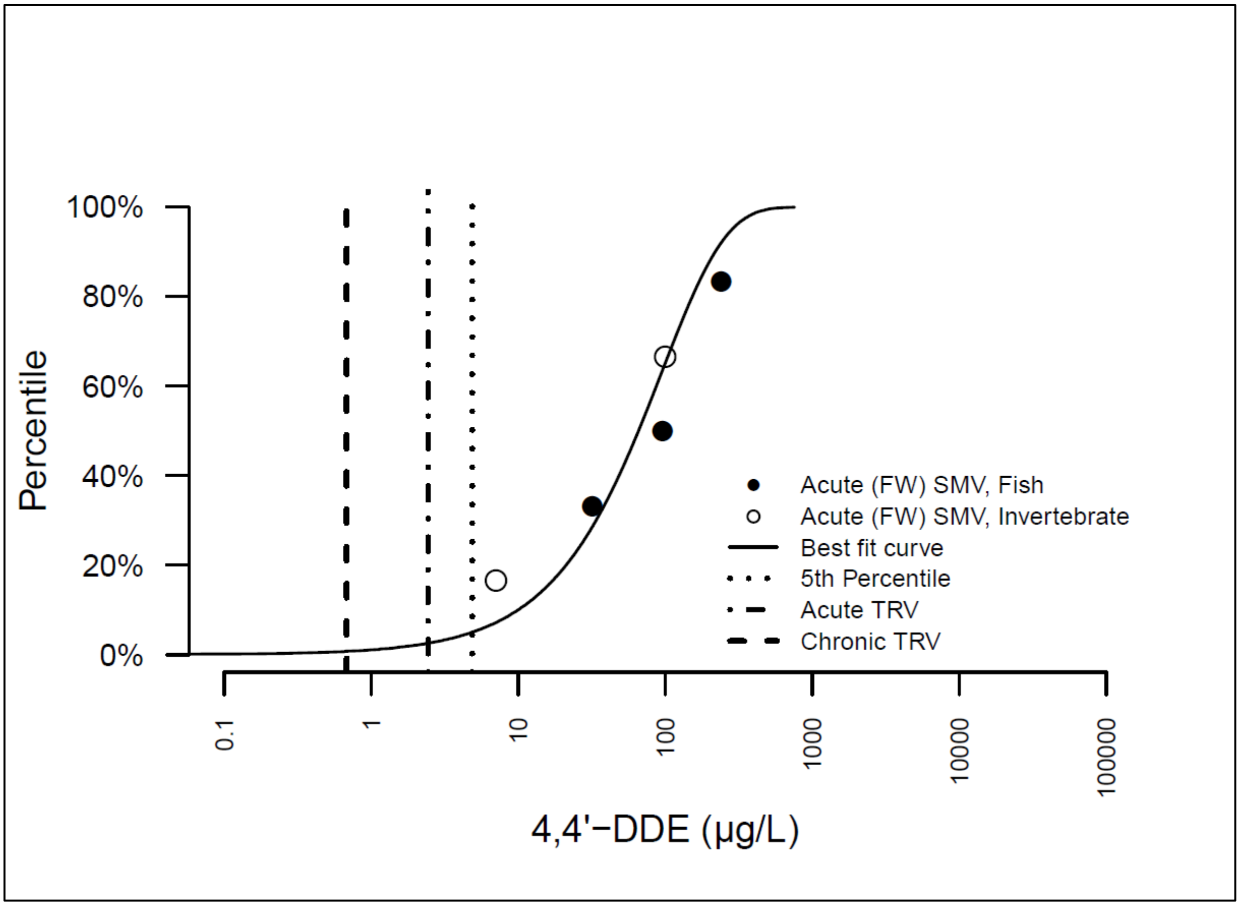


Figure 6-10. Acute freshwater 4,4′-DDE toxicity data SSD

Saltwater acute and chronic toxicity data for 4,4′-DDE were insufficient to develop SSDs. The lowest acceptable acute toxicity value identified in ECOTOX was an LC50 of 2.5 µg/L for *Nitocra spinipes*, a harpacticoid copepod, from Bengtsson (1978). Dividing the LC50 by two provided the acute TRV of 1.25 µg/L. The lowest chronic toxicity value was for the same species—a 14-day EC50 for reproduction of 0.3 µg/L, also from Bengtsson (1978)—is similar to the LC50 divided by the ACR of 3.6. As such, the 14-day EC50 was selected as the chronic saltwater TRV.

4,4′-DDT and Total DDx

Since acute toxicity data were sufficient to develop an SSD, the 5th percentile value of 0.84 µg/L provided the basis for the acute and chronic freshwater TRVs for 4,4′-DDT and total DDx (sum of all six DDT isomers [2,4′-dichlorodiphenyldichloroethane (DDD), 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT and 4,4′-DDT]). The 5th percentile of the SSD divided by two provided the acute TRV of 0.45 µg/L (Figure 6-11). A single ACR of 65 was identified in USEPA (1980c). Suter and Tsao (1996) recommend an ACR of 17.9 when fewer than three ACRs are available; however, Raimondo et al. (2007) report ACRs ranging from 3 to 5 (median 3.6) in four studies of chemicals with a DDT-like mode of action. Applying the median ACR from Raimondo et al. (2007) to the 5th percentile of the SSD provided the chronic TRV of 0.25 µg/L (Figure 6-12; Appendix D).

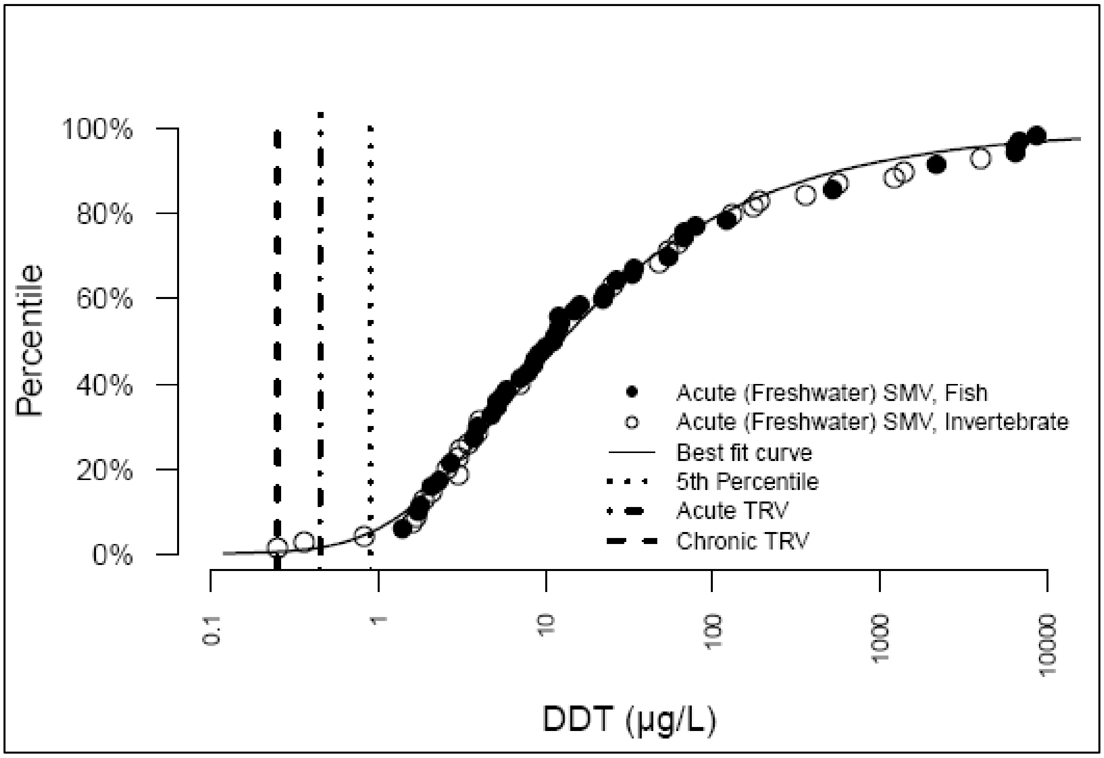


Figure 6-11. Acute freshwater SSD for 4,4′-DDT/total DDx

The acute and chronic saltwater water TRVs are based on an acute SSD for 18 invertebrate and 14 fish species (Figure 6-12). The 5th percentile of the SSD divided by two provides the acute TRV of 0.034 µg/L. Applying the ACR of 3.6 for DDT-type chemicals from Raimondo et al. (2007) provides the chronic TRV of 0.019 µg/L.

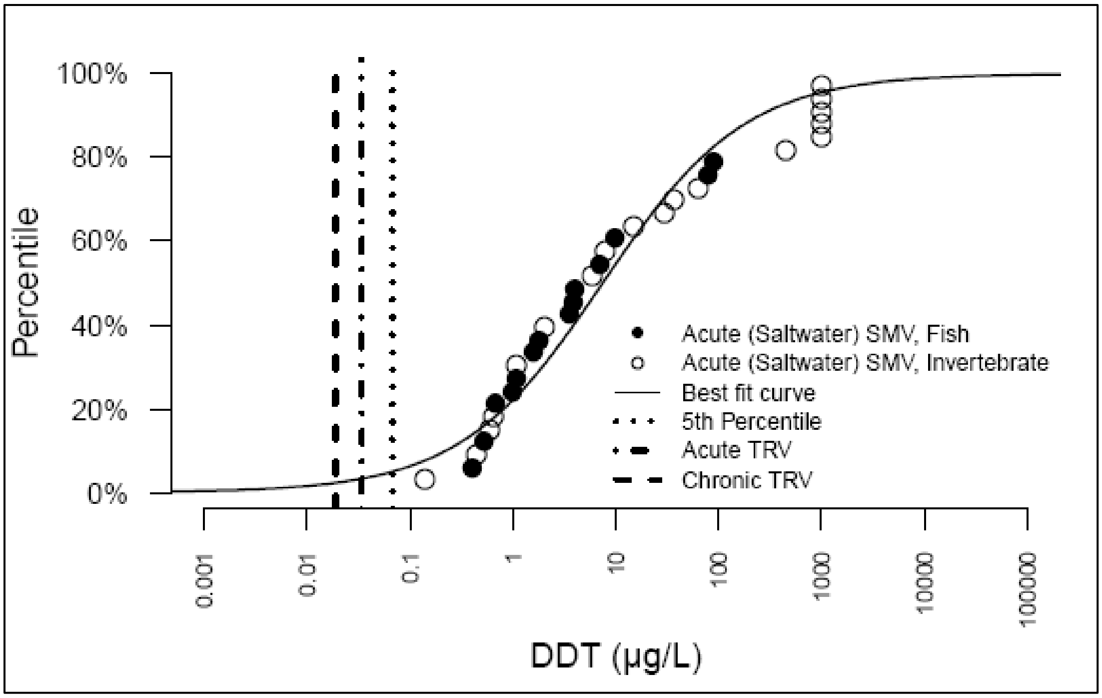


Figure 6-12. Acute saltwater SSD for 4,4′-DDT/total DDx

Total Chlordane

The acute and chronic freshwater and saltwater TRVs were derived from the AWQC FAVs from USEPA (1980b). The saltwater FAV was based on data for four invertebrate and four fish species. The freshwater FAV was based on data for five invertebrate and nine fish species. The FAVs divided by two provided the acute TRVs of 1.2 and 0.045 µg/L for freshwater and saltwater, respectively. The same ACR of 14, the geometric mean of three species-specific ACRs, was applied to the FAVs to provide the chronic TRVs of 0.17 and 0.0064 µg/L for freshwater and saltwater, respectively.

Dieldrin

Saltwater toxicity data for dieldrin is limited. In USEPA (1980a), a saltwater FAV of 0.71 µg/L was derived based on acute toxicity data for 8 invertebrate and 13 fish species. Dividing the FAV by two provided the acute TRV of 0.36 µg/L. The FCV of 0.084 µg/L, which was derived from the FAV using an ACR of 8.5, was selected as the chronic saltwater TRV.

Hexachlorobenzene

The acute and chronic freshwater TRVs for hexachlorobenzene (180 and 57 µg/L, respectively) are based on an SDD for nine fish species (Figure 6-13). The 5th percentile divided by two provides the acute TRV of 180 µg/L. An ACR of 6.2 for chemicals that have an uncoupling mode of action was selected from Raimondo et al. (2007); this median ACR is based on 22 invertebrate and fish ACRs. The chronic toxicity values identified in ECOTOX ranged from 1.8 to 63 µg/L for freshwater fish and invertebrate species.

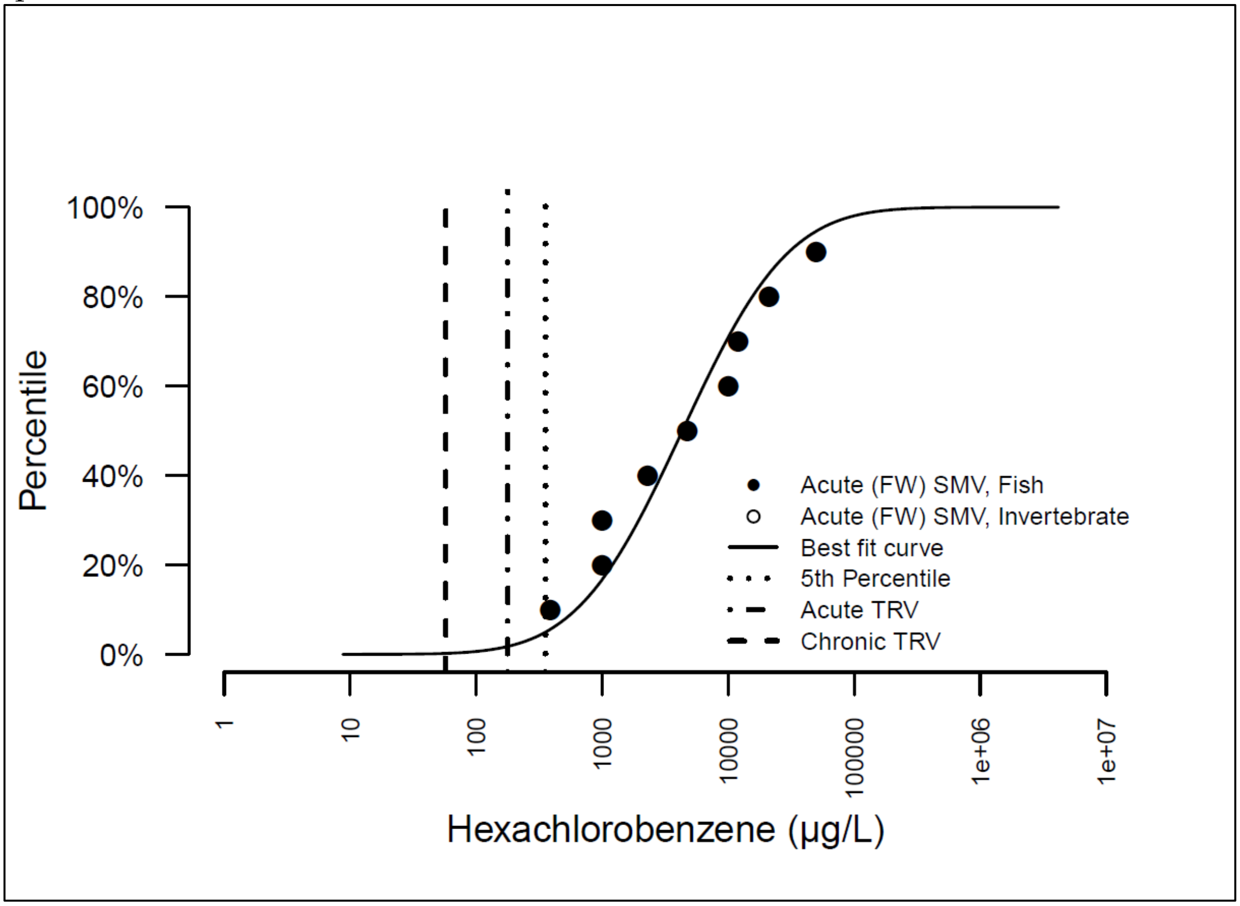


Figure 6-13. Acute freshwater SSD for hexachlorobenzene

Due to limited saltwater toxicity data, the acute and chronic saltwater TRVs were derived from the lowest acceptable acute toxicity value, which was a 96-hr LC50 of 142 µg/L for a fish species, Dover sole (*Solea solea*). The LC50 was divided by two to provide the acute saltwater TRV of 71 µg/L (Furay and Smith 1995). The chronic TRV of 23 µg/L was derived from the acute TRV using the ACR of 6.2 from Raimondo et al. (2007).

Anthracene

The acute and chronic freshwater TRVs for anthracene (0.26 and 0.10 µg/L, respectively) are based on an acute SSD for two invertebrate and four fish species (Figure 6-14). The invertebrate species represented in the SSD, *D. magna* and *D. pulex* (both zooplankton), had SMAVs of 3.7 and 10 µg/L, respectively. The 5th percentile of the SSD was divided by two to provide the acute TRV. No acceptable chronic toxicity data were available for freshwater invertebrates, so the chronic TRV was derived from the 5th percentile using an ACR of 5.09 from DiToro et al. (2000). The ACR is the geometric mean of 35 ACRs for 6 species (fish and invertebrate) and 20 individual

PAHs (DiToro et al. 2000). Using ACRs to extrapolate a chronic criterion from limited acute toxicity data reduces the certainty in the TRV.

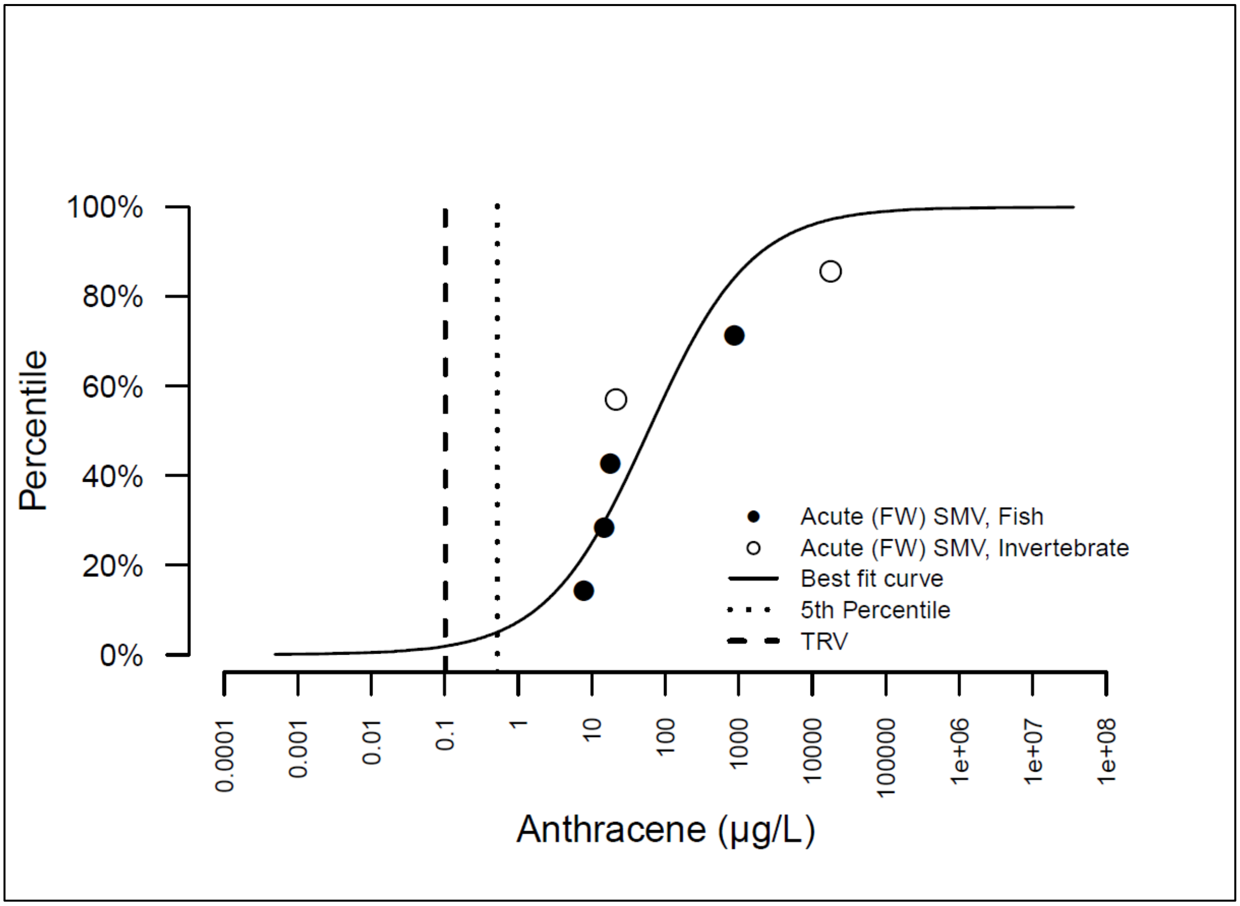


Figure 6-14. Acute freshwater SSD for anthracene

Acute and chronic toxicity data for saltwater species were insufficient to develop SSDs for anthracene. The lowest acute toxicity value for a saltwater species was a 96-hr LC50 of 68.9 µg/L for dwarf surf clam (*Mulina lateralis*) from Pelletier et al. (1997), as cited in USEPA (2016c). The lowest chronic toxicity value from USEPA (2016c) was a 162-day LOEC of 95 µg/L for population abundance of a polychaete worm, *Capitella capitata*. Dividing the lowest acute toxicity value by two provided the acute TRV of 34.5 µg/L. The ACR of 5.09 was applied to the acute toxicity value to provide a more conservative chronic TRV of 13.5 µg/L. The certainty of the representativeness of the estuarine TRV is low due to the limited toxicity data.

Benzo(a)anthracene

The acute and chronic TRVs (0.48 and 0.19 µg/L, respectively) are based on the lowest acute toxicity value for a freshwater species that met the data acceptability criteria, because acute freshwater toxicity data were insufficient for the development of an SSD. The lowest acute value was a 48-hr LC50 of 0.96 µg/L for *D. magna* from Lampi et al. (2005), as reported in USEPA (2016c). In this study, *D. magna* were exposed to simulated solar radiation, which can be assumed to be similar to potential ultra-violet radiation in the field (CCME 1999). The LC50 was divided by two to provide the acute freshwater TRV. No acceptable chronic toxicity values for fish or invertebrates were available, so the chronic TRV was calculated by dividing the acute LC50 by the ACR of 5.09 from DiToro et al. (2000). Due to the lack of saltwater toxicity data, the freshwater acute and chronic TRVs were selected as surrogate saltwater TRVs.

Benzo(a)pyrene

The acute and chronic freshwater TRVs for benzo(a)pyrene (2.0 and 0.80 µg/L, respectively) are based on an acute toxicity SSD (Figure 6-15). The 5th percentile of the SSD was divided by two to provide the acute TRV. The chronic TRV of 0.80 µg/L was derived by dividing the 5th percentile by the ACR of 5.09 from DiToro et al. (2000). The lowest acute toxicity value for a saltwater species was an LC50 of 1.02 µg/L for *D. magna* from Weinstein and Garner (2008), as reported in USEPA (2016c). The LC50 divided by 2 provided the acute saltwater TRV of 0.51 µg/L. No chronic data for saltwater species were available, so the chronic saltwater TRV of 0.20 µg/L was derived by dividing the LC50 by the ACR of 5.09 from DiToro et al. (2000).

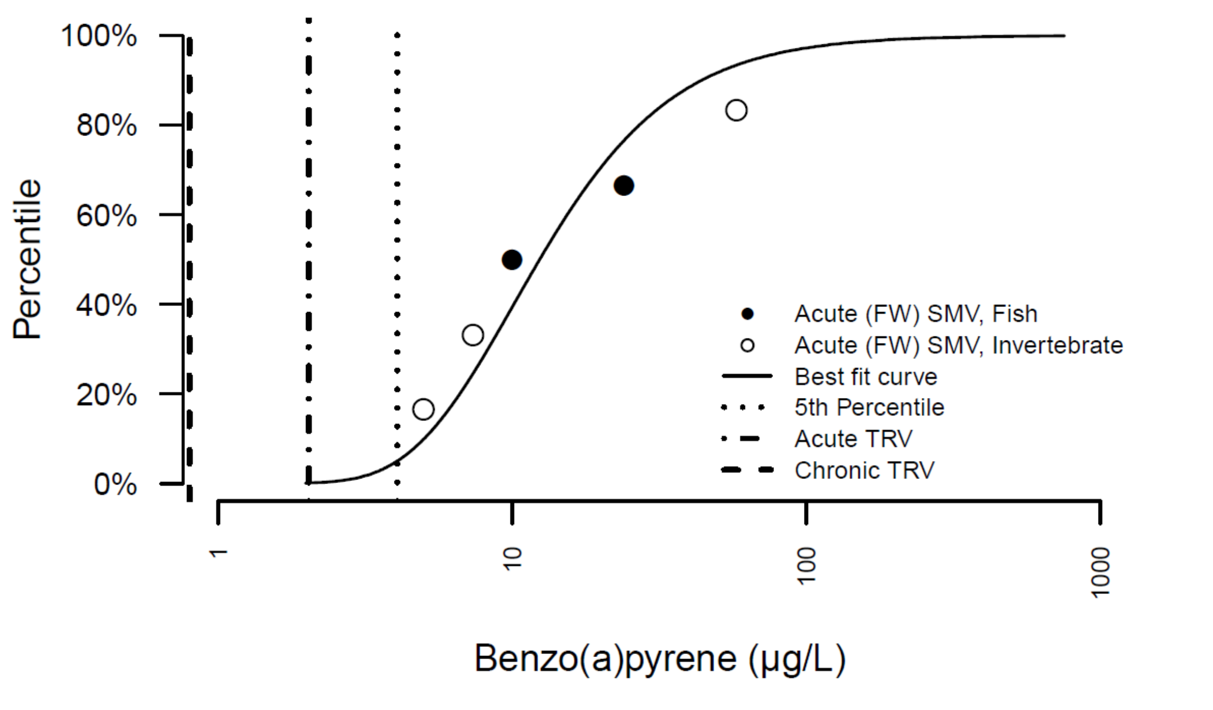


Figure 6-15. Acute freshwater SSD for benzo(a)pyrene

Fluoranthene

The freshwater and saltwater TRVs for fluoranthene are based on acute SSDs (Figures 6-16 and 6-17). Acute freshwater toxicity data were available for 10 invertebrate and 4 fish species. Acute saltwater toxicity data were available for 16 invertebrate and three fish species. Dividing the 5th percentiles by 2 provided the acute freshwater and saltwater TRVs of 13.2 µg/L and 3.02, respectively. Dividing the 5th percentiles by the ACR of 5.09 from DiToro et al. (2000) provided the chronic freshwater and saltwater TRVs of 5.20 µg/L and 1.19 µg/L, respectively.

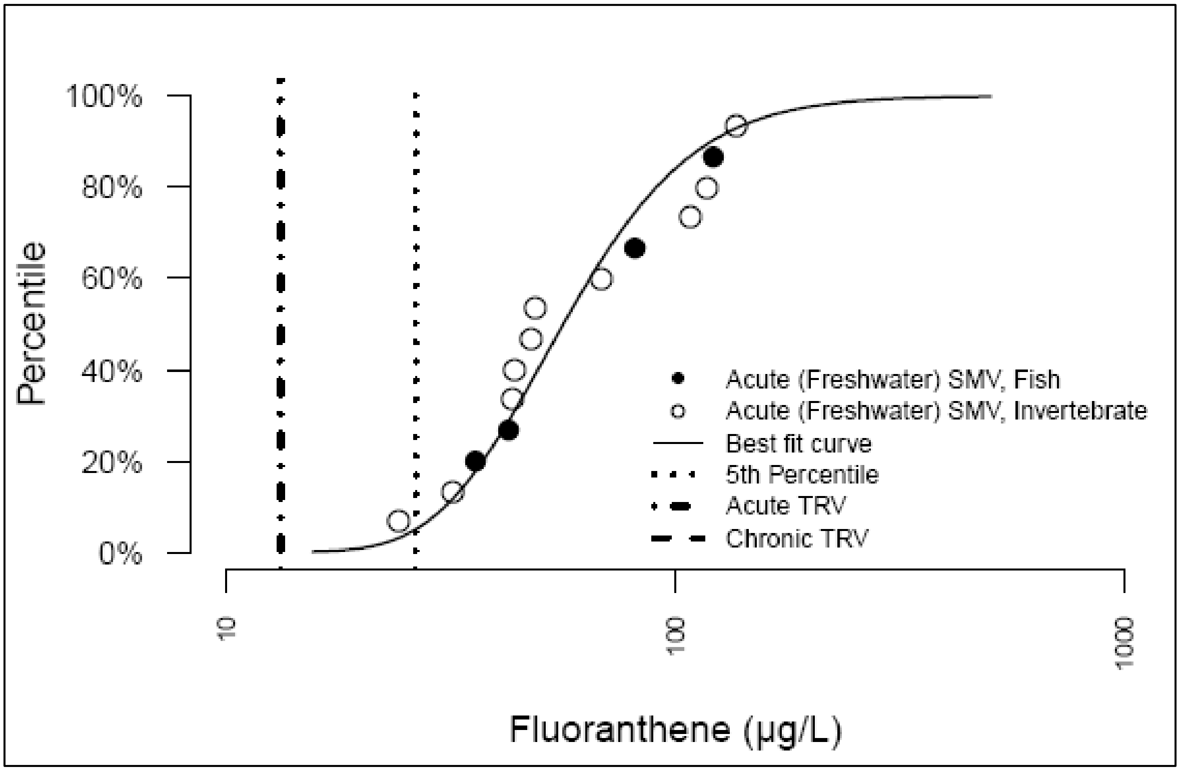


Figure 6-16. Acute freshwater SSD for fluoranthene

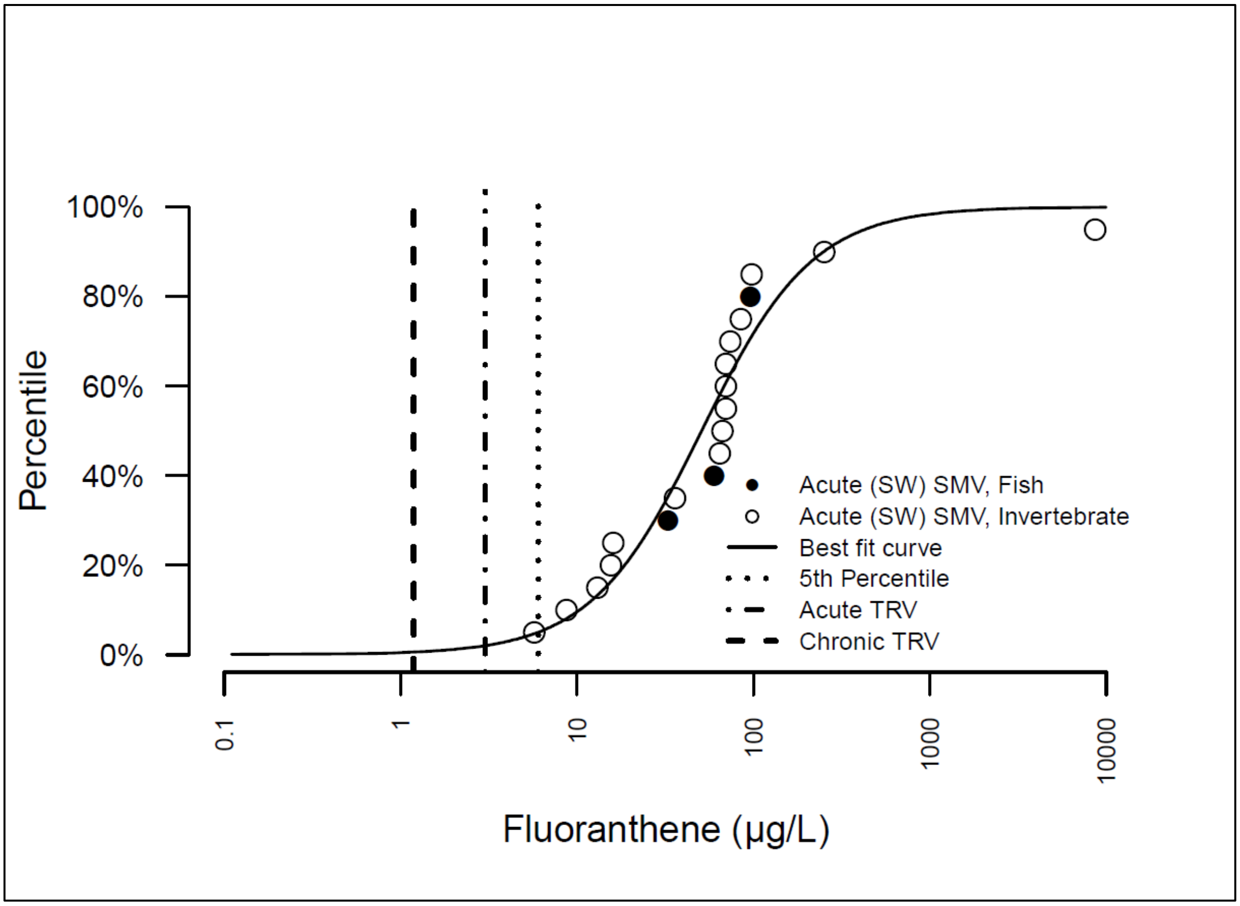


Figure 6-17. Acute saltwater SSD for fluoranthene

Pyrene

The acute and chronic freshwater TRVs for pyrene are based on the lowest acute toxicity value for a freshwater species, a 48-hr mortality EC50 of 4.3 µg/L for *D. magna* from Lampi et al. (2005), which was divided by two to provide the acute TRV of 2.2 µg/L. The lowest chronic toxicity value was a 14-day LC50 of 27.1 µg/L for *Gammarus pulex* from Boxall and Maltby (1997), as reported in USEPA (2016c), to a formulated mixture of three PAHs (pyrene, phenanthrene, and fluoranthene). The next lowest chronic toxicity value for a freshwater species was a 16-day LC50 of 55 µg/L for an amphipod (*H. azteca*); however, this value is uncertain because it is based on the toxicity of a mixture of PAHs in which pyrene was identified as the primary contributor to toxicity. Given the difference between the acute TRV and the lowest chronic toxicity value, dividing the acute EC50 by the ACR of 5.09 from DiToro et al. (2000) provided a more conservative chronic TRV of 0.84 µg/L.

The acute and chronic saltwater TRVs for pyrene are based on the lowest acute toxicity value from USEPA (2016c), which was a 96-hr EC50 of 0.91 µg/L for juvenile clam growth (Pelletier et al. 1997). The EC50 was divided by two to provide the acute saltwater TRV of 0.46 µg/L. The lowest chronic toxicity values for saltwater species were unmeasured concentrations from renewal tests: a 15-day NOEL and LOEL of 40 µg/L for reduced food intake by the bivalves *M. edulis* and *M. galloprovincialis,* respectively; the same study reported a 7-day LOEL of 20 µg/L (USEPA 2016c). As there is uncertainty regarding this TRV—because exposure concentrations were not measured, and it represents a non-standard endpoint (e.g., feeding behavior) that is assumed to have implications for successful growth and survival—a more conservative chronic TRV of 0.18 µg/L was derived from the EC50 using the ACR of 5.09 from DiToro et al. (2000).

BEHP

The acute and chronic freshwater TRVs for BEHP (24.1 and 7.0 µg/L, respectively) are based on an acute toxicity SSD for 4 invertebrate and 12 fish species (Figure 6-18). The 5th percentile of the SSD was divided by two to provide the acute TRV. The chronic TRV was derived from the 5th percentile using an ACR of 6.9 reported in DeFoe et al. (1990) (Appendix D). The ACR is the geometric mean of ACRs for Japanese medaka and rainbow trout exposed to BEHP (DeFoe et al. 1990).

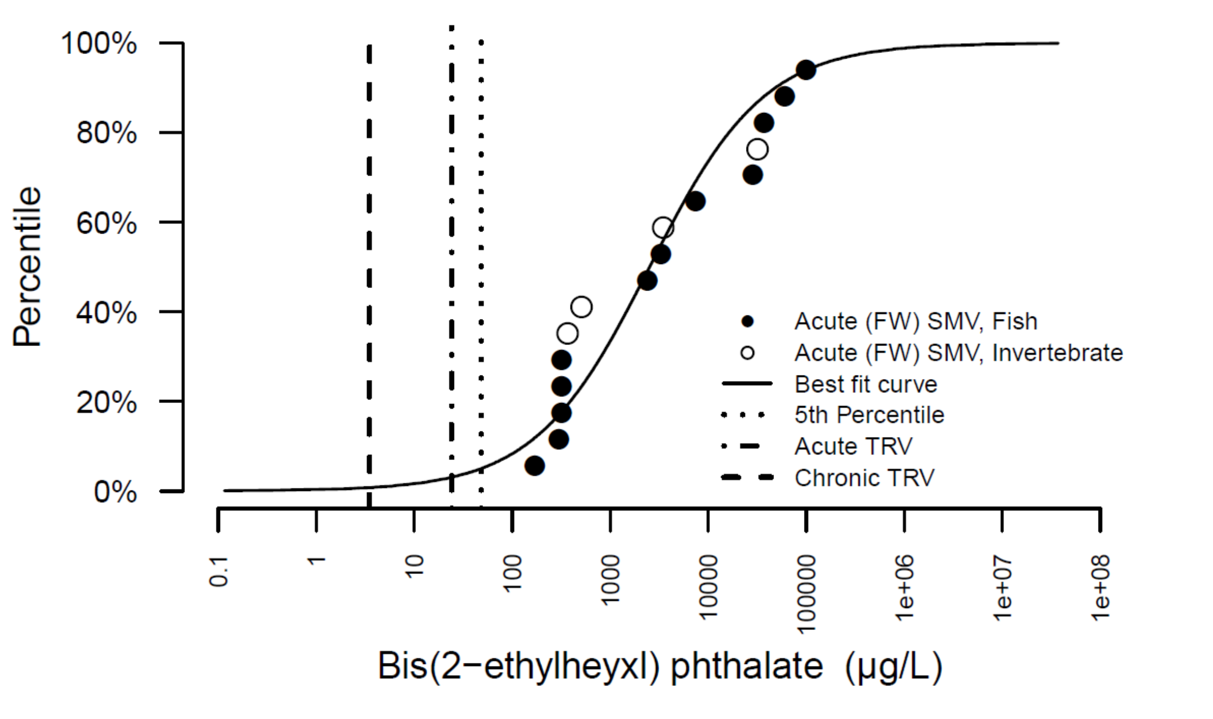


Figure 6-18. Acute freshwater SSD for BEHP

The lowest acute toxicity values for a saltwater species were LC50s of 1,000 µg/L for two invertebrate species (*A. abdita* and *A. bahia*) (Ho et al. 1997); however, these LC50s were based on exposure to contaminated sediment porewater. The only chronic toxicity value based on measured chemistry for a saltwater species was a 28-day LC50 of 125,000 µg/L for *A. bahia* (opossum shrimp) from Horne et al. (1983). The overall lowest chronic value was a 180-day LOEC of 100 µg/L for reproduction of Indian medaka (*O. melastigma*) from Ye et al. (2014), as reported in (USEPA 2016c). In the absence of other data, the LC50 of 1,000 µg/L was divided by two to provide the acute TRV of 500 µg/L, and the LOEC of 100 µg/L was selected as the chronic TRV.

Butyl Benzyl Phthalate

The acute and chronic freshwater TRVs for BBP (107 and 30.9 µg/L, respectively) are based on an acute toxicity SSD for four invertebrate and four fish species. The 5th percentile was divided by two to provide the acute TRV and by the ACR of 6.9 for BEHP from DeFoe et al. (1990) (Figure 6-19).[87] The three invertebrate species represented in the SSD were opossum shrimp (*A. bahia*), an amphipod (*G. minus*), and a cladoceran (*D. magna*), which had SMAVs of 320, 8,700, and 18,000 µg/L, respectively.

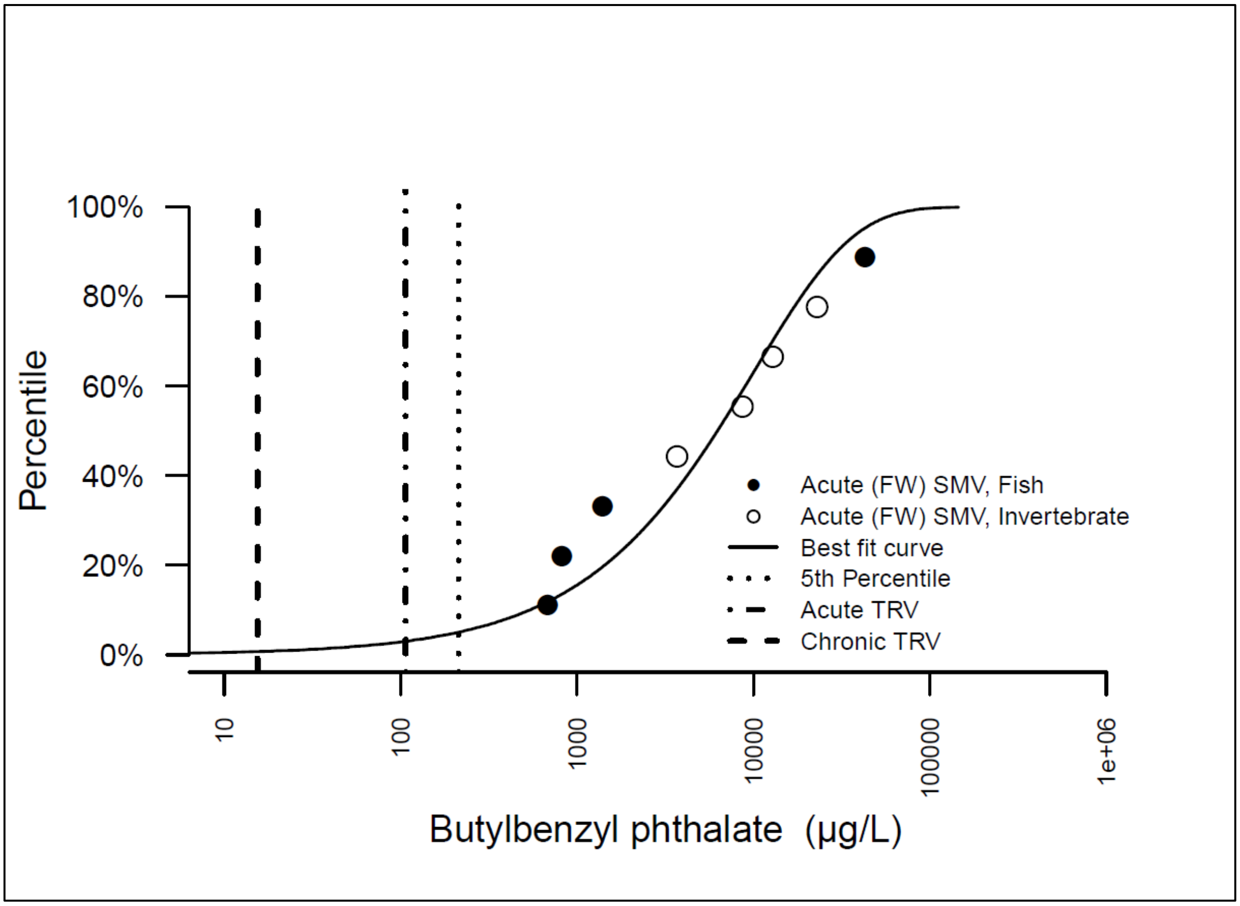


Figure 6-19. Acute freshwater SSD for BBP

[87] The mode of action for phthalate toxicity is expected to be similar, so an ACR for BEHP was used in the absence of an ACR for BBP.

The acute and chronic saltwater TRVs are based on the lowest acute toxicity values for a saltwater species reported in USEPA (2016c). The two lowest acute toxicity values reported were a 165-hr LC50 of 490 µg/L and a 96-hr LC50 of 510 µg/L, both for shiner and perch, from Ozretich et al. (1983). The 165-hr LC50 value was divided by two to provide the acute saltwater TRV of 245 µg/L. Given the lack of acceptable saltwater chronic toxicity data for BBP, the chronic saltwater TRV of 71 µg/L was derived from the LC50 of 490 µg/L using the ACR of 6.9 from DeFoe et al. (1990).

Cyanide

The freshwater and saltwater TRVs for cyanide are based on an acute SSD (Figures 6-20 and 6-21). The 5th percentiles of the freshwater and saltwater SSDs were divided by two to provide the acute TRVs of 32.3 and 6.1 µg/L, respectively.

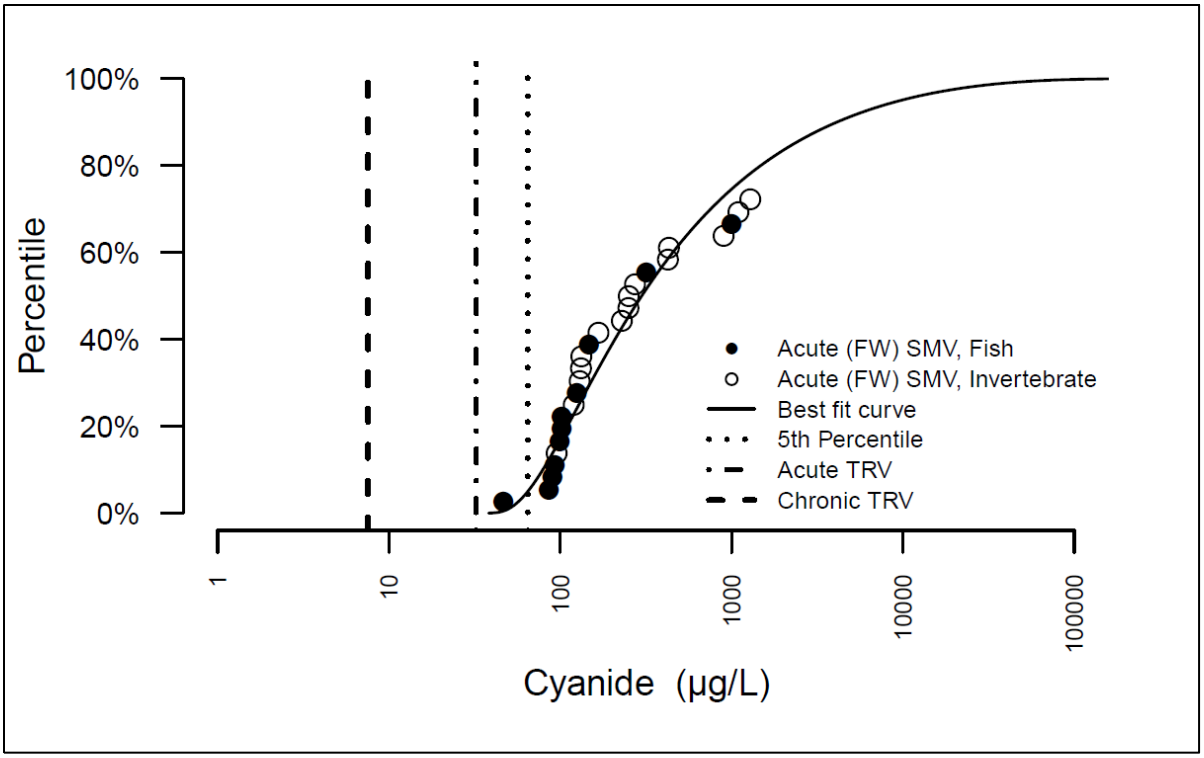


Figure 6-20. Acute freshwater SSD for cyanide

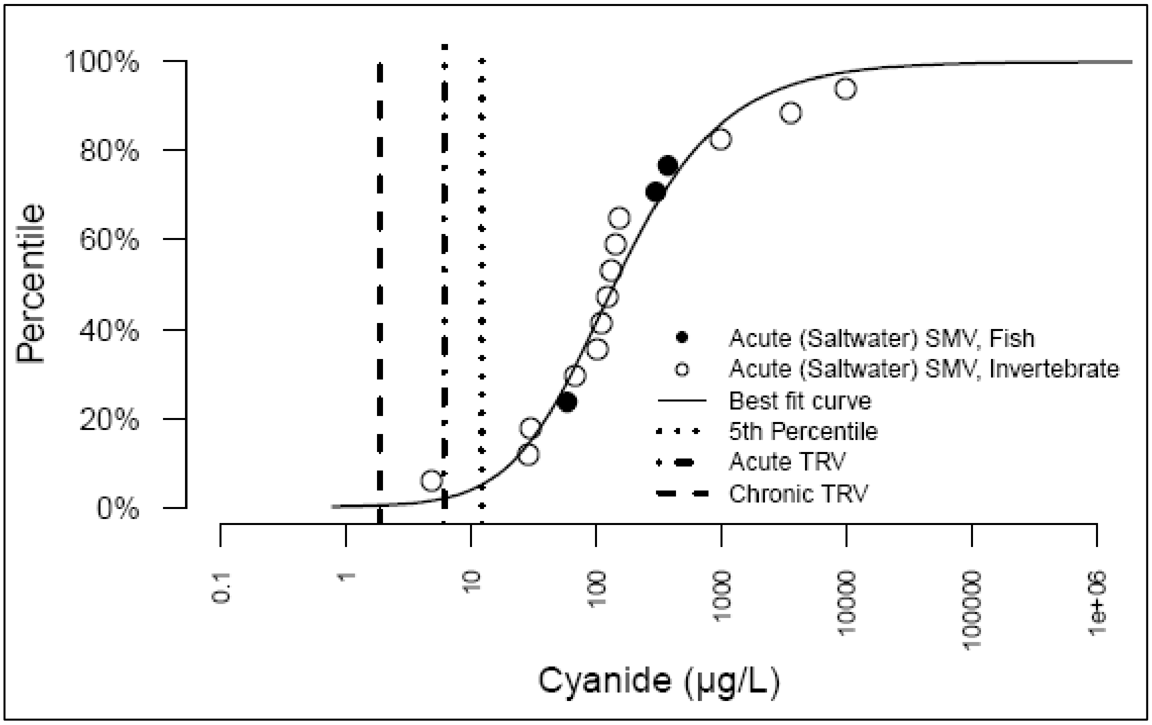


Figure 6-21. Acute saltwater SSD for cyanide

An ACR of 6.45 from Brix et al. (2000) was selected to derive both the freshwater and saltwater chronic TRVs of 7.5 and 1.9 µg/L, respectively. The ACR was derived using two saltwater ACRs (fish and invertebrate) and four freshwater ACRs (three fish, one invertebrate), as reported in USEPA (1985b).

6.2.4 Risk characterization

The following section presents the surface water HQs for benthic invertebrates, as well as uncertainties associated with the HQ calculations.

6.2.4.1 Surface water HQs

HQs were calculated for the surface water COPECs and are presented in Table 6-15. Appendix G lists EPCs, TRVs, and calculated HQs for the surface water COPECs in a single table (Table G2). HQs were < 1.0 for 21 of the 24 COPECs evaluated. EPCs in near-bottom surface water samples exceeded TRVs for three COPECs: copper, 2,3,7,8-TCDD, and cyanide.

Table 6-15. Surface water HQs for benthic invertebrates

COPEC	HQ[a]			
	Estuarine (RM 0–RM 13)[b]		Freshwater (RM 4–RM 17.4)[b]	
	Acute	Chronic	Acute	Chronic
Metals				
Cadmium (dissolved)	0.0015	0.0062	0.0031–0.063[d]	0.010-0.16[d]
Chromium (dissolved)	0.00084	0.0018	0.075	0.11
Copper (dissolved)	0.14–**2.7**[e]	0.14–**2.7**[e]	0.034–0.65[e]	0.055–**1.0**[e]
Lead (dissolved)	0.014	0.14	< 0.001–0.034[e]	0.012–0.67[e]
Mercury (dissolved)	0.0067	0.013	0.013	0.086
Selenium (dissolved)	0.0017	0.0069	na	0.16
Silver (dissolved)	0.0034	0.0095	0.015	0.039
Zinc (dissolved)	0.10	0.41	0.0047–0.044[e]	0.022–0.21[e]
Butyltins				
TBT	0.12[c]	0.76[c]	not a COPEC in freshwater portion	
SVOCs				
BEHP	0.0034	0.017	0.075	0.26
BBP	0.0049	0.017	0.018	0.061
PAHs				
Anthracene	< 0.001	0.0014	0.079	0.21
Benzo(a)anthracene	0.12	0.29	0.16	0.40
Benzo(a)pyrene	0.18	0.45	0.066	0.17
Fluoranthene	0.047	0.12	0.015	0.038
Pyrene	0.32	0.82	0.090	0.24
PCBs				
Total PCBs	0.0072	0.21	0.032	0.14
PCDDs/PCDFS				
2,3,7,8-TCDD	0.0028	**4.3**	0.034	0.14
Organochlorine Pesticides				
4,4′-DDE	0.0012	0.0050	0.0012	0.0021
4,4′-DDT	0.016	0.028	0.0021	0.0038
Total DDx	0.12	0.22	0.015	0.027
Total chlordane	0.076	0.53	0.0047	0.033
Dieldrin	0.0031	0.013	not a COPEC in freshwater portion	
Hexachlorobenzene	< 0.001	< 0.001	< 0.001	< 0.001
Other				
Cyanide	**1.3**	**4.1**	0.23	**1.0**

Bold identifies HQs ≥ 1.0.

Shaded cells identify HQs ≥1 based on acute or chronic TRVs.

[a] HQs based on UCL EPCs presented in Table 6-13 and TRVs presented in Table 6-14.

[b] For BLM applications, a freshwater TRV was used to calculate the HQ if sample-specific salinity was < 3.5 ppth, and an estuarine TRV was used to calculate the HQ if sample-specific salinity was ≥ 3.5 ppth.

[c] HQ based on maximum DL (UCL could not be calculated based on low detection frequency).

[d] HQs based on sample-specific, hardness-based TRVs are presented as a range of values (i.e., each individual sample has a corresponding hardness-based TRV and HQ).

[e] HQs based on sample-specific, BLM-based TRVs are presented as a range of values (i.e., each individual sample has a corresponding BLM-based TRV and HQ).

BBP – butyl benzyl phthalate
BEHP – bis(2-ethylhexyl) phthalate
BLM – biotic ligand model
COPEC – chemical of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
DL – detection limit
EPC – exposure point concentration
HQ – hazard quotient
na – not applicable
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
PCDD – polychlorinated dibenzo-*p*-dioxin
PCDF – polychlorinated dibenzofuran
ppth – parts per thousand
RM – river mile
SVOC – semivolatile organic compound
TBT – tributyltin
TCDD – tetrachlorodibenzo-*p*-dioxin
TEQ – toxic equivalent
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT and 4,4′-DDT)
TRV – toxicity reference value
UCL – upper confidence limit on the mean

As the BLM-based TRVs for copper (saltwater and freshwater), lead (freshwater), and zinc (freshwater) were sample specific, HQs were calculated on a sample-specific basis, rather than based on a UCL. The distinction between freshwater and saltwater/estuarine was determined by the salinity of each sample. NJDEP (2011b) defines freshwater as having salinity < 3.5 ppth. Thus, stations with salinities < 3.5 ppth were evaluated as freshwater and stations with salinity > 3.5 ppth were evaluated as estuarine in the metal BLMs. The ranges of sample-specific HQs for copper, lead, and zinc are provided in Table 6-15. All HQs for lead and zinc were < 1.0; for copper, the maximum HQ was 2.7. Sample-specific copper, 2,3,7,8-TCDD, and cyanide HQs are shown in Figures 6-22 through 6-24.

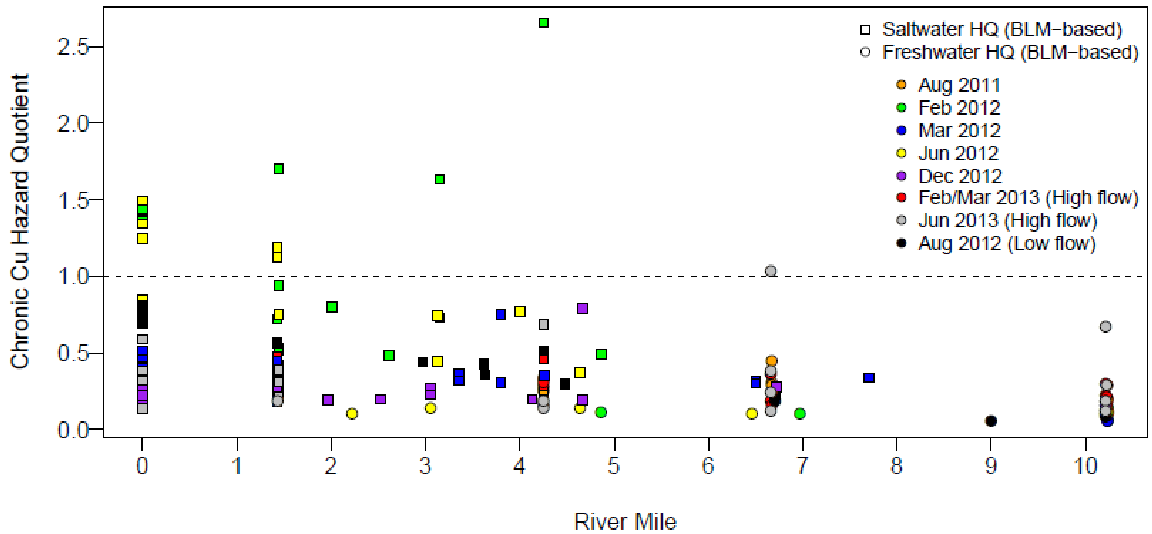


Figure 6-22. Chronic copper BLM-based HQs for individual LPRSA near-bottom surface water samples

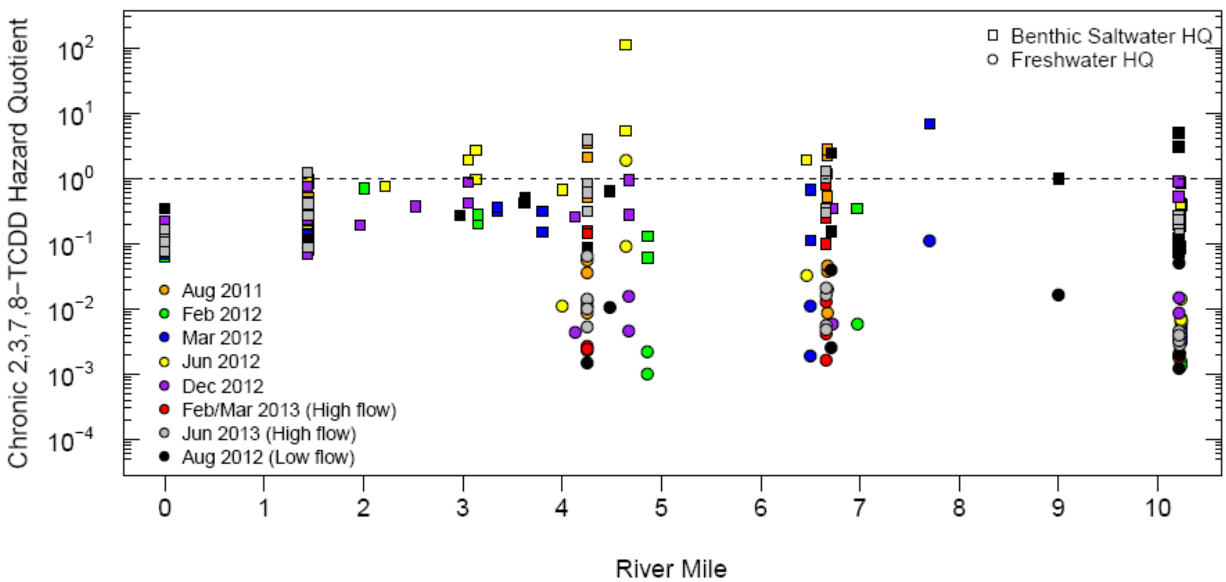


Figure 6-23. Chronic 2,3,7,8-TCDD HQs for individual LPRSA near-bottom surface water samples

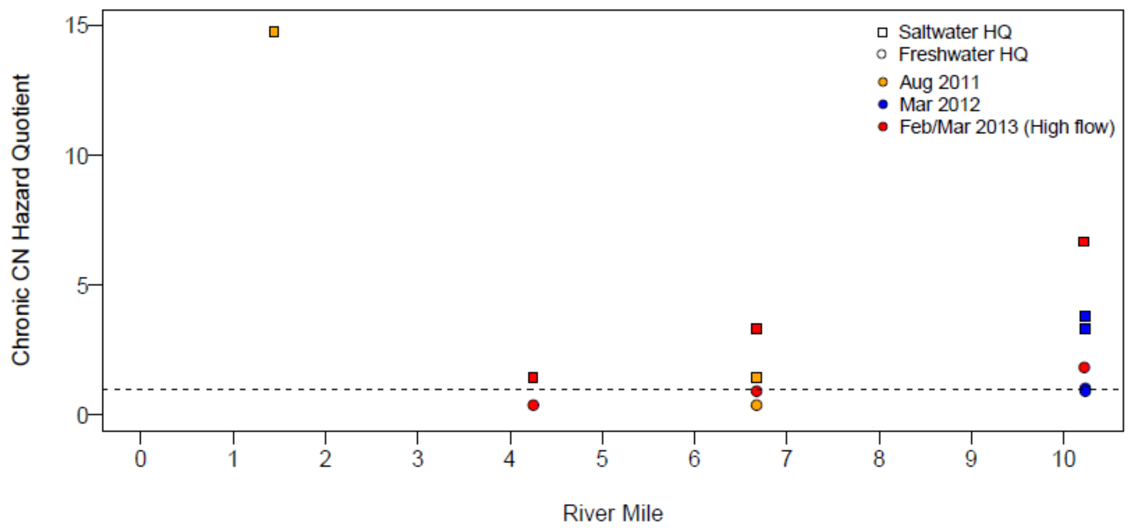


Figure 6-24. Chronic cyanide HQs for individual LPRSA near-bottom surface water samples

6.2.4.2 Uncertainties in risk characterization

This section discusses uncertainties associated with EPCs that could affect HQ calculations for benthic invertebrates and surface water. General TRV uncertainties are discussed in Section 6.2.3.1, and include the use of ACRs and limited availability of toxicity data for some COPECs. The EPC uncertainties addressed in this section that could be evaluated quantitatively are as follows:

- **Treatment of non-detects for EPCs**: The concentrations of sum components (e.g., PCB congeners) that were not detected were assumed to be zero when calculating totals. The effect on HQs of using one-half the DL or the full DL was evaluated. The effects of the first uncertainty on HQ calculations for surface water are presented in Table 6-16. Regardless of how non-detected values in sums were treated (either as zero, one-half the DL, or the full DL), HQs remain less than 1.0.

Table 6-16. Surface water HQs for benthic invertebrates based on uncertainties in EPCs for total PCBs

Uncertainty	Parameter Values/Assumptions		Chronic HQs			
			Estuarine (RM 0–RM 13)		Freshwater (RM 4–RM 17.4)	
	Original	Adjusted	Original	Adjusted	Original	Adjusted
Treatment of non-detects	DL = 0 for non-detects	use of one-half the DL or the full DL for non-detects[a]	0.21	0.18	0.14	0.14

Bold identifies HQs ≥ 1.0.

[a] HQs are the same, regardless of treatment of non-detected values as one-half the DL or as the full DL.

DL – detection limit
EPC – exposure point concentration
HQ – hazard quotient
RM – river mile
PCB – polychlorinated biphenyl

6.2.4.3 Comparison to background

For the three surface water COPECs with HQs > 1 (i.e., copper, 2,3,7,8-TCDD, and cyanide), surface water chemical concentration data from one freshwater background location (sampled multiple times between 2011 and 2013) above Dundee Dam were compared to concentration data from LPRSA surface water collected between RM 4 and RM 10.2. No estuarine background surface water data were available for comparison to LPRSA surface water data.

Copper

The freshwater data shown in Figure 6-25 are the LPRSA samples with salinities < 3.5 ppth, which were compared to the freshwater BLM-based TRVs for copper.

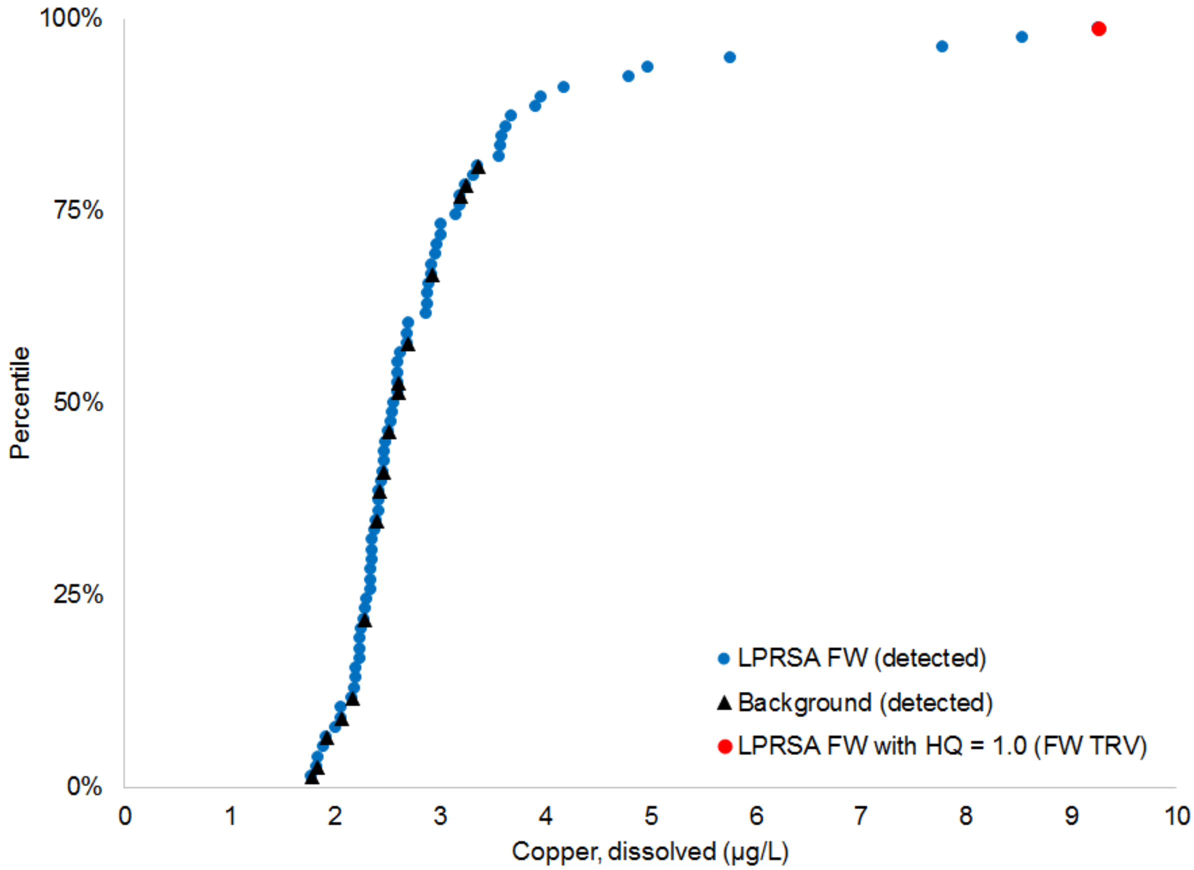


Figure 6-25. Dissolved copper concentrations in LPRSA near-bottom freshwater and background surface water samples

Background concentrations of dissolved copper ranged from 1.78 to 3.36 mg/L. Dissolved copper in approximately 23% (14 out of 60) of near-bottom freshwater samples were outside the range of background concentrations of dissolved copper (Figure 6-25). As shown in Figure 6-25, only one freshwater LPRSA sample had a BLM-based HQ ≥ 1.0; copper concentrations in all other freshwater LPRSA and background samples were below the sample-specific BLM-based TRV.

2,3,7,8-TCDD

2,3,7,8-TCDD was detected in only one of the freshwater background samples at a concentration of 0.00422 ng/L (Figure 6-26). Approximately 62% (44 out of 71) of all near-bottom freshwater samples exceeded the detected background concentration of 2,3,7,8-TCDD. As shown in Figure 6-26, the 2,3,7,8-TCDD concentration in only one freshwater LPRSA sample was above the freshwater TRV.

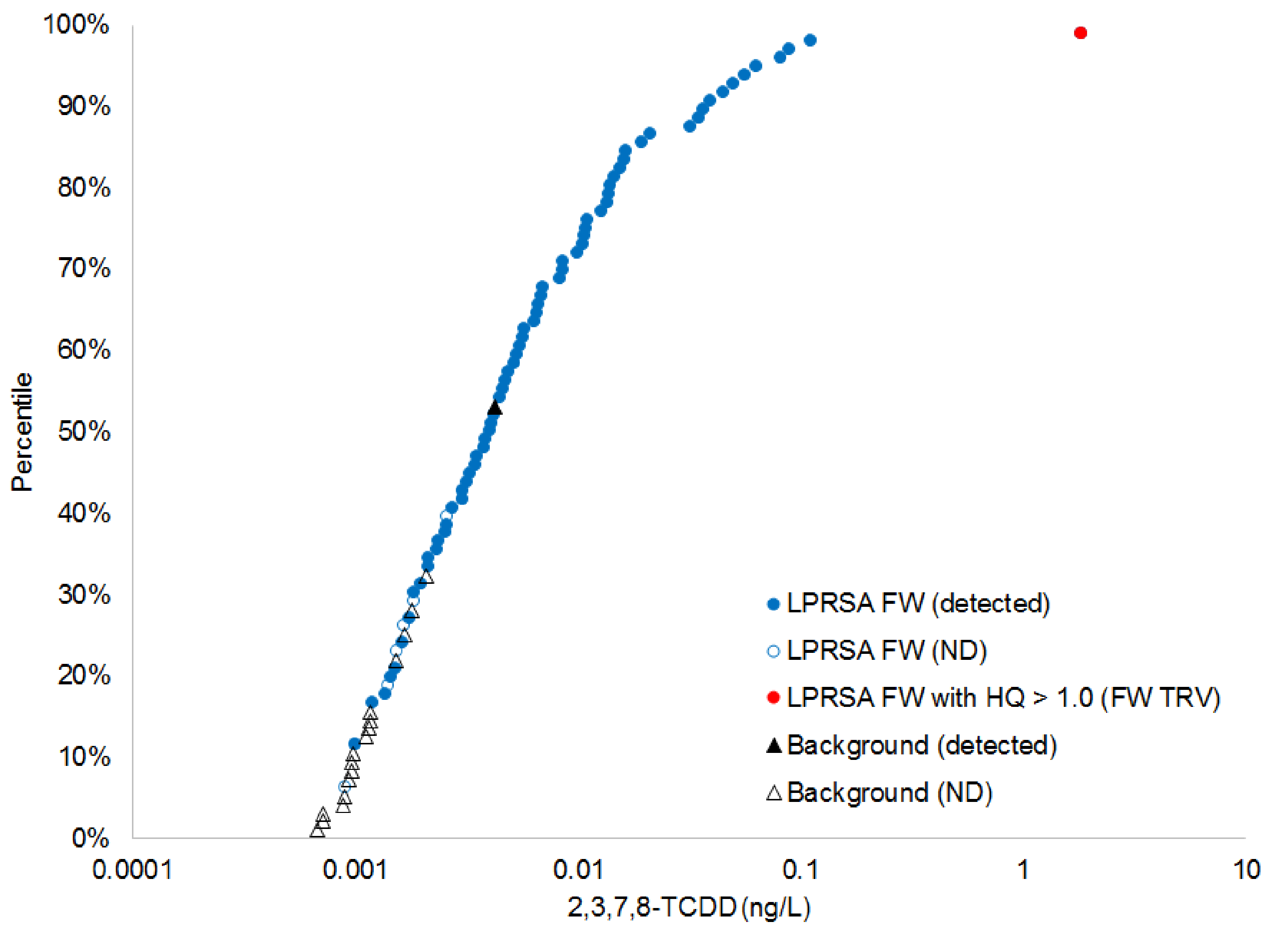


Figure 6-26. 2,3,7,8-TCDD concentrations in LPRSA near-bottom and background surface water samples

Cyanide

Cyanide was detected in only one of the freshwater background samples at a concentration of 0.003 mg/L. The DL for cyanide was 0.01 mg/L. Cyanide in the LPRSA was detected at concentrations greater than the DL and the freshwater TRV in only two of the six near-bottom freshwater samples (Figure 6-27).

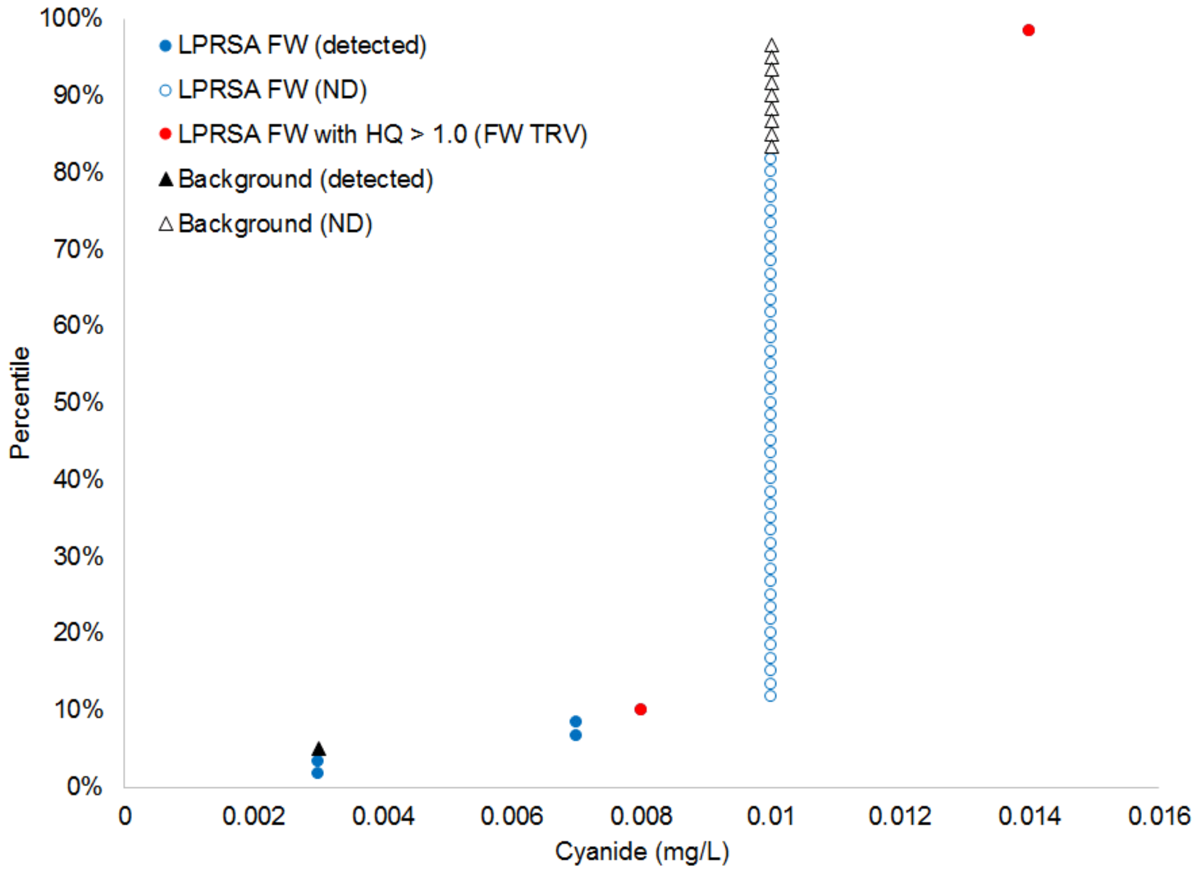


Figure 6-27. Cyanide concentrations in LPRSA near-bottom freshwater and background surface water samples

6.2.5 Summary of key uncertainties

The primary uncertainty associated with the surface water risk characterization is the use of EPCs based on whole-water samples rather than the dissolved, bioavailable form for hydrophobic, nonionic organic chemicals. General TRV uncertainties include the use of ACRs and the limited availability of toxicity data for some COPECs (i.e., PAHs), as discussed in Section 6.2.3.1.

Additionally, some of the selected surface water TRVs may be overly protective of invertebrates, because the TRVs were based on toxicity to fish (i.e., SSD driven by sensitive fish species), as described in Table 6-14.

6.2.6 Summary

HQs were < 1.0 for 21 of the 24 COPECs evaluated. Three of the surface water COPECs—copper, 2,3,7,8-TCDD, and cyanide—had a range of HQs, some of which were ≥ 1.0. These COPECs are further evaluated in Section 6.4.

Risks from exposure to copper were estimated using the BLM. The copper BLM is a predictive toxicity model that considers the effect of water chemistry characteristics on copper bioavailability. Two versions of the BLM were applied for derivation of copper TRVs: a saltwater BLM and a freshwater BLM. The saltwater BLM was developed to predict copper toxicity to the highly sensitive larval life stage of *M. galloprovincialis*. In saltwater, *Mytilus* is the most sensitive genus to copper; it was the basis for the BLM-based, sample-specific TRVs when the salinity of the sample was 3.5 ppth or greater. The freshwater BLM was developed for numerous fish and invertebrate species, and is the basis for the freshwater AWQC for copper. Invertebrates are among the organisms most sensitive to copper, and represent the nine most sensitive genera considered in the current water quality criteria (WQC) for copper. The freshwater copper BLM was used to derive sample-specific TRVs when the salinity of a sample was less than 3.5 ppth. As the copper TRVs are driven by the sensitivity of invertebrates, it is expected that potential risks from exposure of invertebrates to copper in both the freshwater and estuarine portions are reasonably estimated. The primary uncertainty related to using the freshwater BLM-based copper WQC (USEPA 2007b) for acute and chronic freshwater TRVs is that the chronic TRV is derived using an ACR. The primary uncertainty related to using the saltwater BLM for derivation of copper TRVs in saltwater is that chronic toxicity data are limited.

Risk to benthic invertebrates from exposure to 2,3,7,8-TCDD was estimated using a saltwater TRV of 1.65×10^{-5} ng 2,3,7,8-TCDD/L, which was based on the sensitivity of early life stage eastern oyster (*C. virginica*) to sublethal injections of 2,3,7,8-TCDD. 2,3,7,8-TCDD was detected in 84.4% of estuarine near-bottom surface water samples (Table 6-13). The saltwater TRV was 50 times greater than the minimum DL for 2,3,7,8-TCDD, so HQs for all near-bottom surface water samples (including non-detects) were ≥ 1.0, ranging from 50 to 1.1×10^5. As such, the saltwater TRV is expected to be conservative and protective of benthic invertebrates.

As the saltwater TRVs for cyanide are based on toxicity data indicating that invertebrates are more sensitive to cyanide than fish, it is expected that potential risks from exposure of fish between RM 0 and RM 13 (i.e., the estuarine portion) to cyanide are reasonably estimated. However, cyanide was infrequently detected in the LPRSA, in only 7% of all LPRSA near-bottom samples.

6.3 Invertebrate Tissue Assessment

The tissue assessment was conducted for benthic invertebrates (including the benthic invertebrate community, macroinvertebrates, and mollusks). The following species were evaluated:

- Infaunal invertebrates – laboratory-exposed worm tissue (*Nereis virens* [estuarine] and *L. variegatus* [freshwater])
- Macroinvertebrates – blue crab (*Callinectes sapidus*) tissue
- Mollusks – *in situ* caged mussels (ribbed [estuarine] and eastern elliptio [freshwater] mussels)

Tissue data were compared to tissue TRVs to calculate HQs. This section summarizes the COPECs identified from the SLERA, describes the derivation of tissue exposure and effects concentrations, presents the HQs, and discusses the uncertainties associated with the tissue assessment.

6.3.1 COPECs

Benthic invertebrate tissue COPECs were identified in the SLERA (Appendix A) as COIs with maximum concentrations equal to or exceeding their screening-level TRVs (Table 6-17). In the SLERA, COPECs were screened by species; any chemical identified as a COPEC for any of the invertebrate species was evaluated for all invertebrate species.

Table 6-17. Benthic invertebrate tissue COPECs

COPEC	
Metals	
Arsenic	Methylmercury/mercury[a]
Cadmium	Nickel
Chromium	Selenium
Cobalt	Silver
Copper	Vanadium
Lead	Zinc
PAHs	
Total LPAHs	Total HPAHs
PCBs	
Total PCBs	PCB TEQ - fish[b]
PCDDs/PCDFs	
2,3,7,8-TCDD	Total TEQ - fish[b]
PCDD/PCDF TEQ - fish[b]	
Organochlorine Pesticides	
Dieldrin	Total DDx
Heptachlor epoxide	

Note: COPECs are those COIs for which the maximum concentration exceeded its TSV. If a TSV was exceeded based on any invertebrate species evaluated in the SLERA, it was retained as a COPEC for all invertebrate species.

a Although the TRVs were based on total mercury in tissue, TRVs were also compared to methylmercury. Typically, more than 50% of total mercury in lower trophic level fish and invertebrate tissue is in the form of methylmercury. Methylmercury made up 84% of the mercury in blue crab collected in 2009, but only 14% in bioaccumulation worms.

b The evaluation of PCDDs/PCDFs and PCBs using a TEQ approach is highly uncertain for assessing toxicity to invertebrates, because there is limited evidence of ligand activation of the Ah (dioxin) cellular receptor in these organisms. As a result, they are not susceptible to the dioxin-like effects reported for vertebrates (e.g., fish) (Van den Berg et al. 1998). In addition, TEFs are available only for fish, birds, and mammals. However per USEPA comments (USEPA 2015c), PCDDs/PCDFs were screened by comparing the TSV based on 2,3,7,8-TCDD to benthic invertebrate tissue concentrations of 2,3,7,8-TCDD and fish TEQs.

Ah – aryl hydrocarbon
BERA – baseline ecological risk assessment
COI – chemical of interest
COPEC – chemical of potential ecological concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
LPAH – low-molecular-weight polycyclic aromatic hydrocarbon
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
PCDD – polychlorinated dibenzo-*p*-dioxin
PCDF – polychlorinated dibenzofuran
SLERA – screening-level ecological risk assessment
TCDD – tetrachlorodibenzo-*p*-dioxin
TEF – toxic equivalency factor
TEQ – toxic equivalent
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT and 4,4′-DDT)
TRV – toxicity reference value
TSV – toxicity screening value
USEPA – US Environmental Protection Agency

A number of COIs could not be screened as part of the SLERA (Appendix A) because no tissue TSVs were available. These COIs are presented in Section 5.5.2, as are the implications of not being able to evaluate these COIs.

6.3.2 Exposure

EPCs were calculated for each of the COPEC-invertebrate tissue pairs. Tissue EPCs were calculated as the UCLs using all available composite samples for worms (whole-body tissue from bioaccumulation testing), blue crab (calculated whole-body,[88] muscle, and hepatopancreas tissue[89]), and mussels (soft tissue). UCLs were calculated using USEPA's ProUCL® statistical package (Version 5.0.00) (USEPA 2013d) as described in Section 4.3.7.[90] COPEC summary concentrations of benthic invertebrate tissue samples are presented in Appendix C. A summary of benthic invertebrate tissue EPCs is presented in Table 6-18.

Table 6-18. Benthic invertebrate tissue EPCs

		EPC				
		Blue crab			Mussels[a]	Worms
COPEC	Unit (ww)	Whole Body	Hepatopancreas	Muscle	Whole Body	Whole Body
Metals						
Arsenic	mg/kg	1.4	ne	ne	0.0	1.4
Cadmium	mg/kg	0.11	ne	ne	0.0090	0.11
Chromium	mg/kg	1.4	ne	ne	13	21
Cobalt	mg/kg	0.076	ne	ne	0.075	0.69

88 Methods for the calculation of reconstituted whole-body blue crab tissue concentrations from individual muscle, hepatopancreas, and carcass concentrations are presented in Section 4.3.3. Whole-body concentrations were evaluated as EPCs for those COPECs for which selected TRVs were based on whole-body concentrations.

89 Muscle and hepatopancreas EPCs were developed for blue crab and methylmercury/mercury for comparison to TRVs based on these specific tissue types (Section 6.6.3).

90 The UCL recommended by USEPA's ProUCL® software (typically the 95% UCL, but in some cases the 97.5% or even the 99% UCL) was used. The selected UCL statistic is presented in Appendix C1.

Table 6-18. Benthic invertebrate tissue EPCs

COPEC	Unit (ww)	EPC				
		Blue crab			Mussels[a]	Worms
		Whole Body	Hepatopancreas	Muscle	Whole Body	Whole Body
Copper	mg/kg	24.6	ne	ne	0.35	5.8
Lead	mg/kg	0.36	ne	ne	0.13	6.4
Mercury	µg/kg	140	67	200	8.0	59
Methyl mercury	µg/kg	120	49	190	3.2	2.9
Nickel	mg/kg	1.0	ne	ne	6.6	13
Selenium	mg/kg	0.79	ne	ne	0.052	0.54
Silver	mg/kg	0.61	ne	ne	0.0014	0.028
Vanadium	mg/kg	0.12	ne	ne	0.087	0.59
Zinc	mg/kg	36.4	ne	ne	1.6	34
PAHs						
Total HPAHs	µg/kg	110	ne	ne	220	2000
Total LPAHs	µg/kg	83	ne	ne	82	540
PCBs						
Total PCBs	µg/kg	350	ne	ne	24	240
PCB TEQ - fish[b]	ng/kg	0.78	ne	ne	0.021	0.20
PCDD/PCDFs						
2,3,7,8-TCDD	ng/kg	57	ne	ne	2.2	38
PCDD/PCDF TEQ- fish[b]	ng/kg	62	ne	ne	2.3	38
Total TEQ - fish[b]	ng/kg	63	ne	ne	2.3	39
Pesticides						
Dieldrin	µg/kg	6.8	ne	ne	2.7	1.6
Heptachlor epoxide	µg/kg	6.3	ne	ne	0.96	0.36
Total DDx	µg/kg	68	ne	ne	5.3	16

[a] Mussel EPCs are based on day 0-normalized concentrations.

[b] TEQ calculated using the Kaplan-Meier approach.

COPEC – chemical of potential ecological concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
EPC – exposure point concentration
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
LPAH – low-molecular-weight polycyclic aromatic hydrocarbon
ne – not evaluated
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
PCDD– polychlorinated dibenzo-*p*-dioxin
PCDF –polychlorinated dibenzofuran
TCDD – tetrachlorodibenzo-*p*-dioxin
TEQ – toxic equivalent
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT and 4,4′-DDT)
ww – wet weight

PCDDs/PCDFs were evaluated as both 2,3,7,8-TCDD and TEQs based on fish TEFs for invertebrates. The TEQ approach of evaluating the toxicity of PCDDs/PCDFs and PCBs to invertebrates is uncertain, because there is limited evidence of ligand activation of the Ah (dioxin) cellular receptor in these organisms. As a result, they are not susceptible to the dioxin-like effects reported for vertebrates (e.g., fish) (Van den Berg et al. 1998). In fact, TEFs are available for only fish, birds, and mammals.

6.3.3 Effects

This section presents the effects data (i.e., TRVs) selected for the COPECs identified in the SLERA. A range of TRVs was evaluated. The selection was based on a comprehensive review of the primary literature and an assessment of acceptability. TRVs were also based on previous LPRSA Region 2 documents. The following subsections describe the methods used to identify TRVs and the selected TRVs for each COPEC. Selected TRVs are consistent with the comments, responses, and directives received from USEPA on June 30, 2017 (USEPA 2017b), September 18, 2017 (USEPA 2017e), July 10, 2018 (USEPA 2018), January 2, 2019 (CDM 2019), and March 5, 2019 (USEPA 2019); during face-to-face meetings or conference calls on July 24, September 27, October 3, November 6, 2017, July 24, 2018, and August 16, 2018; and via additional deliverables and communications between the CPG and USEPA from August through December 2017, July through September 2018, and January through June 2019.

6.3.3.1 Methods for deriving tissue TRVs

This section describes the TRV selection process, the generation of TRVs from SSDs, the selection of TRVs for regulated metals, general uncertainties associated with TRVs, and uncertainties associated with the SSD approach.

TRV Selection Process

Two sets of benthic tissue TRVs were used for the derivation of HQs in this BERA. One set of TRVs was based on previous documents developed by USEPA Region 2 for LPRSA:

- USEPA's revised draft of the LPR restoration project FFS (Louis Berger et al. 2014)
- USEPA's first draft of the LPRSA FFS (Malcolm Pirnie 2007b)

The second set of TRVs was selected by first conducting a literature search for relevant toxicological studies, as described in Appendix E. These studies were then evaluated for acceptability of use. For those studies considered acceptable (as described in Appendix E), NOAELs and lowest-observed-adverse-effect levels (LOAELs) were derived. TRVs were then selected for each COPEC and benthic invertebrate group pair based on an evaluation of all acceptable NOAEL and LOAEL TRVs. Details regarding the literature search and study acceptability are presented in Appendix E. Receptor group-specific TRVs (i.e., decapod-specific TRVs) and tissue-specific TRVs (i.e., blue crab hepatopancreas- and muscle tissue-specific TRVs) were developed, when toxicity data were available.

TRV Derivation Based on SSDs

When sufficient data were available (i.e., data for at least five species), TRVs were generated using an SSD approach. The LOAEL TRV was selected as the 5th percentile of the SSD, and the NOAEL TRV was extrapolated from the LOAEL TRV using an uncertainty factor of 10, though there is uncertainty associated with the use of extrapolation factors. When data were insufficient, the lowest acceptable LOAEL and highest NOAEL below the LOAEL from the same study were selected as TRVs.

An SSD is a statistical model that can be used to calculate a chemical concentration protective of a predetermined percentage of a group of species. SSDs are intended to provide an indication of both the total range and distribution of species sensitivities in natural communities, even when the actual range of sensitivities is unknown (Stephan 2002). In practice, SSDs are most commonly presented as a cumulative distribution function (CDF) of the toxicity of a chemical to a group of laboratory test species.

All toxicity data for various invertebrate species meeting the TRV selection criteria were considered in constructing the SSDs. LOAELs represent the lowest concentrations at which an adverse effect is observed, whereas NOAELs indicate the concentration at which no adverse effect is observed. However, HQs greater than or equal to 1.0 based on NOAELs do not indicate whether an adverse effect can be expected. Therefore, LOAELs were considered appropriate for developing SSDs to determine the potential for an adverse effect. For each chemical, a single effects threshold (the final species LOAEL) was determined for inclusion in the SSD considering all acceptable LOAELs for that species.

For studies reporting acute LOAELs (i.e., mortality endpoints with < 28 days of observation and no growth or reproduction data reported in the same study), chronic LOAELs as inputs into the SSD dataset were estimated using ACRs (Table 6-19).

Table 6-19. Chemical-specific ACRs applied to acute LOAELs

COPEC	ACR	Source
Arsenic	3.8	USEPA (1985a)
Cadmium	9.1	USEPA (2001); Raimondo et al. (2007)
Copper	3.22	USEPA (2007b)
Total DDx	3.6	Raimondo et al. (2007)
Total PCBs	8.4	USEPA (1980d)

ACR – acute-to-chronic ratio
COPEC – chemical of potential ecological concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
LOAEL – lowest-observed-adverse-effect level
PCB – polychlorinated biphenyl
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT and 4,4′-DDT)
USEPA – US Environmental Protection Agency

When multiple studies evaluated the same species, the data were processed before being incorporated into the SSD. For any given toxicological endpoint (i.e., survival, growth, or reproduction), including sub-lethal effects related to population-level effects, the geometric mean of all chronic LOAELs for that endpoint was calculated to determine the final endpoint value. If LOAELs for multiple endpoints were available, the lowest value among the endpoints was selected. For example, if toxicological data for survival and growth were reported in multiple studies for a particular species, first the geometric mean of all survival data and the geometric mean of all growth data were independently calculated, then the lower of the survival and growth geometric means was selected as the final species LOAEL.

After final species LOAELs were calculated for each species, final species LOAELs were ranked from lowest to highest, and the cumulative percent frequency value for each data point was calculated using Equation 6-1 (Stephan et al. 1985):

$$CPF = Rank \times \left(\frac{100}{n+1} \right)$$ **Equation 6-1**

Where:

CPF = cumulative percent frequency
n = number of data points used to develop the SSD

The cumulative percent frequency value of each data point was then plotted against the final species LOAEL, yielding the typically S-shaped SSD plot with effect concentrations on the x-axis and cumulative frequency values on the y-axis.

Several theoretical distribution models were then fit to the final species LOAELs and their corresponding empirical cumulative frequency distributions using @RISK software. @Risk software provides rankings of several goodness-of-fit statistics, including the Akaike's information criterion (AIC), Bayesian information criterion (BIC), chi-squared, Kolmogorov-Smirnov (K-S), and A-D fit statistics.

For the estimation of tissue SSD TRVs herein, the selection of distributions focused on the AIC statistic, which corresponds to the fit of a theoretical distribution to the entire empirical distribution, as well as a visual inspection of several curve fits. When the "best" AIC value did not correspond to a model with reasonable visual fit to the lower tail of the empirical data, the rankings of goodness-of-fit statistics (i.e., AIC, BIC, chi-squared, K-S, and A-D) for each distribution fit by @Risk were summed, resulting in a general indication of the best-fitting distribution(s). The top-ranked distributions (based on the sum of ranked statistics) were then compared visually. The TRV was calculated based on the distribution with the best visual fit among the top-ranked distributions. If multiple distributions had similarly good visual fits, then the TRV was calculated as the geometric mean of the 5th percentile estimates for all accepted models.

The distributions selected for each SSD are described in Section 6.3.3.2. Consistent with AWQC derivation methods (Stephan et al. 1985), the 5th percentile of the distribution was selected as the TRV. The 5th percentile concentration is assumed to protect 95% of the invertebrates species present in the LPRSA.

TRVs for Regulated Metals

Regulated metals in aquatic tissue were evaluated consistently with USEPA (2015b, 2015c, 2016g) recommendations. TRVs for the evaluation of regulated metals in this BERA were developed as follows:

- For copper and lead, TRVs from the revised draft of the LPR restoration project FFS (Louis Berger et al. 2014) were used.
- For all regulated metals COPECs (i.e., metals COPECs other than mercury and selenium), TRVs were developed from toxicological literature based on the approach outlined above. When sufficient data were available (i.e., data for at least five species), TRVs were generated using an SSD approach. When data were insufficient, the lowest acceptable LOAEL and highest NOAEL below the LOAEL from the same study were selected as TRVs. The exception is for copper and zinc, which are nutritionally essential minerals; their nutritionally optimal concentrations were selected as the NOAEL TRV, and the next highest LOAEL was selected as the LOAEL TRV.

Individual toxicological studies compiled from the USACE Environmental Residue Effects Database (ERED), extensive literature searches, and CPG's TRV database were reviewed. Once these studies had been evaluated, those that were deemed acceptable for the development of TRVs were compiled (Appendix E).

TRV Uncertainty – General

USEPA defines a TRV as a "dose above which ecologically-relevant effects might occur to wildlife species following chronic dietary exposure and below which it is reasonably expected that such effects will not occur" (USEPA 2003e). Studies in the literature that reflect the specific ecological species selected for evaluation and focus on the specific compounds associated with the site were often not available. This process usually requires the selection of toxicity studies conducted under a wide variety of testing methodologies and protocols. Common differences among toxicity studies include different test species, time of exposure, exposure route, toxicity endpoints measured, sample size of test species, number of treatments tested, compounds included in the test (mixtures or single compounds), and exposure setting (laboratory or field). Due to the variation among testing methods, a large number of studies (when available[91]) were examined, and key studies that related to the assessment endpoints identified in this BERA were selected to derive appropriate TRVs. This process involved professional judgement, which could have led to the generation of multiple values, depending on the criteria used to evaluate the studies. For this BERA, an approach that utilized multiple TRVs for specific contaminants was used to present a range of NOAEL and LOAEL values, providing risk managers with a better understanding of the inherent risks associated with exposure to compounds in the LPRSA.

The processes for risk assessment TRV selection all contains some degree of uncertainty due to the issues identified above. Section 6.3.3.2 describes the uncertainty associated with the individual TRVs selected for this BERA for each COPEC (both those derived by CPG and those derived by USEPA).

TRV Uncertainty – TRVs based on SSDs

Compared to a LOAEL based on a single study and test species, tissue-based SSDs provide a measure of community sensitivity by incorporating not only toxicity data for many species, but also multiple toxicity values (typically LOAELs) for the same species from different studies. The use of SSDs is also conservative, in that the SSD TRV tends to be selected from the lower tail of the SSD, most often the 5th percentile value.[92] There are some uncertainties associated with SSDs (and SSD TRVs) that should be considered in interpreting risk estimates based on SSD-derived TRVs.

Tissue TRVs developed for the risk assessment herein using SSDs were based on several statistical and biological assumptions. The statistical assumptions pertained to: 1) the number of samples included in the SSD, 2) the type of effects associated with the toxicity values included in the SSDs (i.e., ACR-adjusted acute values versus chronic values), and 3) the appropriateness (e.g., goodness-of-fit) of the distribution(s) used to describe the SSDs. Biological assumptions were made regarding whether the actual community of interest was sufficiently protected by the SSD TRV (i.e., 5th percentile

[91] The toxicity data for invertebrate tissue TRVs were generally limited.

[92] This value is sometimes referred to as the hazard concentration, or HC5.

value), which was typically based on non-site-specific species, and whether the theoretical distribution fit to SSD data was representative of the actual community of interest. Additionally, there were statistical and biological uncertainties associated the use of LOAELs to construct SSDs, as described above.

The number of samples included in an SSD impacts the reliability and stability of the TRV derived from that SSD. As the number of samples increases, the stability of the TRV tends to increase. This is particularly true of TRVs selected from the tail of an SSD, which is associated with a greater level of uncertainty (than the median or similar percentiles). Wheeler et al. (2002) and Newman et al. (2000) indicated that datasets of between 10 and 55 data points, depending on the distribution and spread of the data, were required for a low-percentile SSD TRV (e.g., 5th percentile) to be stable, regardless of the dataset from which the SSD was developed. Roman et al. (1999) concluded that when fewer than five data points are available to derive an SSD, a TRV based on the lowest toxicity value is an estimated low-percentile SSD TRV, but that increasing the number of toxicity values (i.e., n > 5) in the SSD allows for greater confidence in a low-percentile TRV. These criteria were used to determine if data were sufficient to generate an SSD.

While the use of ACRs allows for more data to be used in SSD development, uncertainty in the use of an ACR to calculate chronic LOAELs for use in the SSD should be considered. Allard et al. (2010) highlights how adsorption, distribution, metabolism, and excretion rates vary considerably between acute and chronic exposures, and concludes that it is unwise to use TRVs based on acute studies when assessing risks from chronic exposures. ACRs used in this BERA were not available from specific studies or for species associated with the acute tissue LOAELs; instead, they were based on ACRs calculated from aquatic toxicity data. The applicability of water exposure-based ACRs to acute tissue-based toxicity data is unknown, but this approach has been used previously for assessments of the LPRSA (The Louis Berger Group et al. 2014).

ACRs vary widely and are influenced by biological, chemical, and environmental factors (Raimondo et al. 2007). Furthermore, because effects are associated with a critical tissue concentration at the site of action (e.g., metals sorbed to the gill), exposure duration theoretically does not affect the threshold concentration. In practice, whole-body tissue effects thresholds can be lower in acute studies, because the whole-body concentration does not reach equilibrium with the concentration at the site of action in the short amount of time that the chemical can bioaccumulate. The low exposure in chronic tests over a longer time period (relative to an acute exposure) makes it possible for species to sequester the toxicant away from the site of action (e.g., organic chemicals in fatty tissues or metals in muscle, bone, or granules), effectively reducing the chemical's toxicity and allowing it to accumulate to a greater body burden. Thus, using water exposure-based ACRs may underestimate chronic tissue LOAELs. In most cases, the ACR-adjusted values are similar to chronic toxicity values, so removing the ACR-adjusted values would have only a minor effect on TRV estimation. The expected influence of ACR-estimated LOAELs on the resulting TRVs is discussed for each applicable COPEC in Section 6.3.3.2.

The selection of a theoretical distribution to represent the empirical toxicity values in an SSD can have a substantial effect on a calculated TRV. Depending on the shape of a distribution and its goodness-of-fit to the lower tail of the empirical data, SSD TRVs can differ by one or more orders of magnitude. Visual inspection of curve fits to the empirical data, while essential for making informed statistical decisions, is subjective. Goodness-of-fit statistics provide an objective means of ranking distributions against one another, but the statistics may not reflect the goodness-of-fit of a curve to the lower tail of a distribution. Instead, statistics can be biased toward the fit at the middle of a distribution. Even the A-D statistic, which quantifies a distribution's fit to the tails of empirical data, can be heavily influenced by the fit of a distribution to the upper tail, rather than the lower tail from which the SSD TRV is estimated. In some cases (i.e., when several distributions provide similar results), neither the visual fit nor goodness-of-fit statistics provide an unambiguous "best" distribution.

TRV Uncertainty – Regulated metals

The use of a tissue residue approach for metals (except for methylmercury and selenium[93]) is highly uncertain because of the wide range of strategies used by organisms to store, detoxify, and excrete bioaccumulated metals (e.g., fish and invertebrates may regulate their body burdens of some metals, although metals regulation, and the strategy thereof, is species and metal specific) (Adams et al. 2011; USEPA 2007e). The USEPA framework for metals risk assessment (USEPA 2007e) recommends against the use of a tissue residue approach, stating that the critical body residue (CBR) approach for metals "does not appear to be a robust indicator of toxic dose."

As summarized by Adams et al. (2011), total metals concentrations in whole-body tissue do not reflect the biologically or metabolically active portion of metal that is available to contribute to toxicity at the site of action. Metals are transformed into different chemical species when they are transferred from one media to another, including within and between organisms in the food web, which may result in a multitude of metal species (with varying toxicities). Once taken up, internal transport, storage, detoxification, and elimination mechanisms further alter the metal species present and their distributions.

Trace metal accumulation patterns differ among organisms and among metals, even for the same organism. Some organisms can accumulate rather high metal concentrations without apparent negative effects, whereas other organisms show signs of toxicity at much lower whole-body tissue concentrations. Metals in an organism can be metabolically active (and potentially toxic) or stored in non-toxic storage depots

93 Selenium tissue residue TRVs based on dietary exposures included only those exposures involving organic forms of selenium in the diet. Exposure to organic forms of selenium in the diet is the most environmentally relevant exposure (DeForest and Adams 2011).

(Rainbow 2002, 2007). Thus, the same tissue residue that, in one case, results in an adverse effect can, in another case, be non-toxic (e.g., Kraak et al. 1992; Andres et al. 1999; Hook and Fisher 2002). Furthermore, because internal fate and transport processes are rate dependent, tissue burdens associated with the toxicity of metals to aquatic organisms strongly depend on exposure scenario and exposure time.

Thus, differences in metal uptake, detoxification, metabolism, and elimination kinetics of the organisms further limit the utility of whole-body tissue concentrations in predicting risk.

6.3.3.2 Selected TRVs for benthic invertebrates

Benthic invertebrate tissue TRVs selected are presented in Table 6-20, and TRVs for regulated metals are presented in Table 6-21. These TRVs are described in detail in the sections below, and toxicity data used to derive the selected TRVs are presented in Appendix E.

Table 6-20. Benthic invertebrate tissue TRVs

COPEC	Units (ww)	Tissue Type	Range of TRVs[a]								Document
			TRV-A[b]				TRV-B[c]				
			NOAEL	LOAEL	Endpoint	Source	NOAEL	LOAEL	Endpoint	Source	
Metals											
Methylmercury/ mercury	µg/kg	hepato-pancreas[d]	100[e]	1,000	survival (shore crab)	Bianchini and Gilles (1996)	48	95	reproduction (copepod)	Hook and Fisher (2002)	revised FFS (Louis Berger et al. 2014)
	µg/kg	muscle[f]	340	na	survival (lobster)	Canli and Furness (1995)					
	µg/kg	whole body[g]	48	95	reproduction (copepod)	Hook and Fisher (2002)					
Selenium	mg/kg	whole body	0.050[h]	0.51	growth (midge)	Malchow et al. (1995)	no value[i]	no value[i]	na	na	na
PAHs											
Fluoranthene (HPAH)	µg/kg	whole body	8,100	22,200	growth and reproduction (amphipod)	Schuler et al. (2007)	na	na	na	na	na
Fluorene (LPAH)	µg/kg	whole body	11,000[e]	111,000	mortality (amphipod)	Lee et al. (2002)	na	na	na	na	na
Total LPAHs	µg/kg	whole body	na	na	na	na	78	780	reproduction (estuarine polychaete)	Emery and Dillon (1996a)	revised FFS (Louis Berger et al. 2014)
Total HPAHs	µg/kg	whole body	na	na	na	na	66	660	reproduction (blue mussel)	(Eertman et al. 1993; Eertman et al. 1995)	revised FFS (Louis Berger et al. 2014); updated by USEPA (2017a)
PCDDs/PCDFs											
2,3,7,8-TCDD	ng/kg	whole body	300[e]	3,000	survival (crayfish)	Ashley et al. (1996)	0.15[j]	1.3[j]	reproduction (eastern oyster)	Wintermyer and Cooper (2003)	revised FFS (Louis Berger et al. 2014)

COPEC	Units (ww)	Tissue Type	Range of TRVs[a]								Document
			TRV-A[b]				TRV-B[c]				
			NOAEL	LOAEL	Endpoint	Source	NOAEL	LOAEL	Endpoint	Source	
PCDD/PCDF TEQ -fish[k]	ng/kg	whole body	300[e]	3,000	survival (crayfish)	Ashley et al. (1996)	0.15[j]	1.3[j]	reproduction (eastern oyster)	Wintermyer and Cooper (2003)	revised FFS (Louis Berger et al. 2014)
Total TEQ - fish[k]	ng/kg	whole body	300[e]	3,000	survival (crayfish)	Ashley et al. (1996)	0.15[j]	1.3[j]	reproduction (eastern oyster)	Wintermyer and Cooper (2003)	revised FFS (Louis Berger et al. 2014)
PCBs											
Total PCBs	µg/kg	whole body	52[e]	520	survival, growth, and reproduction (10 species)	SSD-derived 5th percentile value	6.4	17	reproduction (eastern oyster)	Chu et al. (2000); Chu et al. (2003)	revised FFS (Louis Berger et al. 2014), updated by USEPA (2017d)
PCB TEQ - fish[k]	ng/kg	whole body	300[e]	3,000	survival (crayfish)	Ashley et al. (1996)	0.15[j]	1.3[j]	reproduction (eastern oyster)	Wintermyer and Cooper (2003)	revised FFS (Louis Berger et al. 2014)
Organochlorine Pesticides											
Dieldrin	µg/kg	whole body	8.0	80	survival (pink shrimp)	Parrish et al. (1973)	1.6	8.0	survival (pink shrimp)	Parrish et al. (1973)	revised FFS (Louis Berger et al. 2014)
Heptachlor epoxide	µg/kg	whole body	10	140	survival (American oyster)	Schimmel et al. (1976)	no value[i]	no value[i]	na	na	na
Total DDx	µg/kg	whole body	11[e] (1.0[e,l])	110 (10[l])	survival, growth, and reproduction (6 species)	SSD-derived 5th percentile value	60	130	survival (amphipod)	(Nimmo et al. 1970)	revised FFS (Louis Berger et al. 2014)

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

b TRVs were derived from the primary literature review based on the process identified in Section 6.3.3.1.

[c] TRVs were based on USEPA's revised draft of the LPR restoration project FFS (Louis Berger et al. 2014) or first draft of the LPR restoration project FFS (Malcolm Pirnie 2007b), as described in Section 6.3.3.1.

[d] Mercury TRV based on Norway lobster and shore crab hepatopancreas tissue concentrations was selected for comparison to LPRSA blue crab hepatopancreas tissue.

[e] NOAEL extrapolated from LOAEL using an uncertainty factor of 10.

[f] Mercury TRV based on Norway lobster muscle tissue concentration was selected for comparison to LPRSA blue crab muscle tissue.

[g] Mercury TRV based on whole-body tissue concentrations was selected for comparison to LPRSA worm and bivalve tissue.

[h] NOAEL based on DL.

[i] No TRVs were recommended by USEPA in the revised FFS (Louis Berger et al. 2014) or draft FFS (Malcolm Pirnie 2007b).

[j] TRV derived from a field study.

[k] The evaluation of PCDDs/PCDFs and PCBs using a TEQ approach is highly uncertain for assessing the toxicity to invertebrates, given that TEFs are available only for fish, birds, and mammals. However, per USEPA comments (USEPA 2015c), PCDDs/PCDFs were screened by comparing the TSV based on 2,3,7,8-TCDD to benthic invertebrate tissue concentrations of 2,3,7,8-TCDD and fish TEQs.

[l] An alternate SSD distribution was also selected based on a conservative distribution fit; see Section 6.3.3.2 for additional description.

COPEC – chemical of potential ecological concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
DL – detection limit
FFS – focused feasibility study
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
LOAEL – lowest-observed-adverse-effect level
LPAH – low-molecular-weight polycyclic aromatic hydrocarbon
LPR – Lower Passaic River
LPRSA – Lower Passaic River Study Area
na – not applicable
NOAEL – no-observed-adverse-effect level
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
PCDD – polychlorinated dibenzo-*p*-dioxin
PCDF – polychlorinated dibenzofuran
SSD – species sensitivity distribution
TCDD – tetrachlorodibenzo-*p*-dioxin
TEF – toxic equivalency factor
TEQ – toxic equivalent
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT and 4,4′-DDT)
TRV – toxicity reference value
USEPA – US Environmental Protection Agency
ww – wet weight

Table 6-21. Benthic invertebrate tissue TRVs for regulated metals

COPEC	Units (ww)	Tissue Type	Range of TRVs[a]								
			TRV-A[b]				TRV-B[c]				
			NOAEL	LOAEL	Endpoint	Source	NOAEL	LOAEL	Endpoint	Source	Document
Metals											
Arsenic	mg/kg	whole body	0.064[d]	0.64	survival and growth (7 species)	SSD-derived 5th percentile value	no value[e]	no value[e]	na	na	na
Cadmium	mg/kg	whole body	0.024[d]	0.24	survival, growth, and reproduction (29 species)	SSD-derived 5th percentile value	no value[e]	no value[e]	na	na	na
Chromium	mg/kg	whole body	1.5	3.5	survival and growth (amphipod)	Norwood et al. (2007)	no value[e]	no value[e]	na	na	na
Cobalt	mg/kg	whole body	1.2	2.6	survival and growth (amphipod)	Norwood et al. (2007)	no value[e]	no value[e]	na	na	na
Copper	mg/kg	whole body	na[f]	na[f]	na[f]	na[f]	5	12	survival (clam)	Absil et al. (1996)	revised FFS (Louis Berger et al. 2014)
Lead	mg/kg	whole body	4.0[d]	40	survival (amphipod)	Spehar et al. (1978)	0.52	2.6	survival (amphipod)	Borgmann & Norwood (1999)	revised FFS (Louis Berger et al. 2014)
Nickel	mg/kg	whole body	0.10[g]	1.1[g]	survival (copepod)	Borgmann et al. (2001)	no value[e]	no value[e]	na	na	na
Silver	mg/kg	whole body	0.49	0.59	growth and reproduction (water flea)	Naddy et al. (2007)	no value[e]	no value[e]	na	na	na
Vanadium	mg/kg	whole body	na[f]	na[f]	na[f]	na[f]	no value[e]	no value[e]	na	na	na

COPEC	Units (ww)	Tissue Type	Range of TRVs[a]								
			TRV-A[b]				TRV-B[c]				
			NOAEL	LOAEL	Endpoint	Source	NOAEL	LOAEL	Endpoint	Source	Document
Zinc	mg/kg	whole body[h]	8.0[d]	80	survival *(bivalve)*	King et al. (2004)	no value[e]	no value[e]	na	na	na
	mg/kg	whole body[i]	5.1[d]	51	survival (crustacean)	Muyssen et al. (2006)					
	mg/kg	whole body[j]	80	na[f]	survival (polychaete)	King et al. (2004)					

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

b TRVs were derived from the primary literature review based on the process identified in Section 6.3.3.1.

c TRVs were based on USEPA's revised draft of the LPR restoration project FFS (Louis Berger et al. 2014) or first draft of the LPR restoration project FFS (Malcolm Pirnie 2007b), as described in Section 6.3.3.1.

d NOAEL extrapolated from LOAEL based on an uncertainty factor of 10.

e No TRVs were selected by USEPA in the revised FFS (Louis Berger et al. 2014) or draft FFS (Malcolm Pirnie 2007b).

f No TRVs were selected, see Section 6.3.3.2 for further explanation.

g TRV derived from a field study.

h Zinc whole-body TRV based on bivalve was selected for comparison to LPRSA bivalve whole-body tissue.

i Zinc whole-body TRV based on crustacean was selected for comparison to LPRSA blue crab whole-body tissue.

j Zinc whole-body TRV based on polychaete was selected for comparison to LPRSA worm whole-body tissue.

COPEC – chemical of potential ecological concern
FFS – focused feasibility study
LOAEL – lowest-observed-adverse-effect level
LPR – Lower Passaic River
LPRSA – Lower Passaic River Study Area
na – not applicable
NOAEL – no-observed-adverse-effect level
SSD - species sensitivity distribution
TRV – toxicity reference value
USEPA – US Environmental Protection Agency
ww – wet weight

Mercury and Methylmercury

Three acceptable TRV studies were available that evaluated the toxicity of mercury or methylmercury to benthic invertebrates based on whole-body tissue concentrations of total mercury that resulted in LOAELs (Kopfler 1974; Biesinger et al. 1982; Hook and Fisher 2002). LOAELs were available for only three species or types of invertebrates (a copepod [*Acartia tonsa]*, a cladocern [*D. magna*], and a bivalve [*Crassostrea virginica*]), and were therefore insufficient to develop an SSD. Tissue LOAELs ranged from 95 µg/kg wet weight (ww) for reproductive effects in marine copepods (Hook and Fisher 2002) to 23,000 µg/kg ww for mortality in *C. virginica* (Kopfler 1974).

For worms and mussels, the lowest LOAEL (95 µg/kg ww) based on whole-body copepod tissue was selected as the LOAEL TRV, and the NOAEL from the same study (48 µg/kg ww) was selected as the NOAEL TRV. These same TRVs were also selected for mercury (Louis Berger et al. 2014) based on Hook and Fisher (2002). There are uncertainties associated with the selected TRVs due to the limited dataset (three studies).

For blue crab, decapod-specific TRVs were recommended for specific tissue types. Two acceptable TRV studies were also available for the toxicity of mercury based on hepatopancreas or muscle tissue concentrations in decapods (Canli and Furness 1995; Bianchini and Gilles 1996). These studies were conducted with shore crab and Norway lobster, decapod species that are more closely related to blue crab than copepod, which was the organism used in the TRV study with the lowest LOAEL for whole-body tissue (Hook and Fisher 2002). For hepatopancreas tissue, the lowest LOAEL was based on decreased survival in shore crab at a concentration of 1,000 µg/kg ww (Bianchini et al. 1982); no NOAEL was available from that study. The NOAEL TRV (100 µg/kg ww) was extrapolated from the LOAEL TRV using an uncertainty factor of 10. There is uncertainty associated with the derivation of a NOAEL based on an extrapolation factor. For muscle tissue, only a NOAEL was available; the highest NOAEL was based on effect on survival in Norway lobster at a concentration of 340 µg/kg ww[94] (Canli and Furness 1995). The TRVs selected for blue crab were the LOAEL of 1,000 µg/kg ww for hepatopancreas tissue and the NOAEL of 340 µg/kg ww for muscle tissue. There are uncertainties associated with the selected TRVs due to the limited dataset (two studies).

Selenium

One study examining effects on growth was found to meet TRV acceptability criteria. A LOAEL was available only for one species (a midge [*Chironomus decorus*]) and data were therefore insufficient to develop an SSD. In this study, Malchow et al. (1995) exposed midges (*C. decorus*) to diet-based selenate for 96 hrs and showed a statistically significant (15%) decrease in growth (relative to control) at a tissue concentration of

[94] Value was converted from the original dry weight value of 1.7 mg/kg reported in the paper using a moisture content of 80%.

0.51 mg/kg ww.[95] Because there were no other studies, the 96-hr NOAEL and LOAEL of 0.05[96] and 0.51 mg/kg ww, respectively, were selected as the NOAEL and LOAEL TRVs.

No TRVs for selenium were available in the revised FFS (Louis Berger et al. 2014) or draft FFS (Malcolm Pirnie 2007b).

There are uncertainties associated with the selected TRVs because of the very limited dataset (one study). The selected LOAEL TRV of 0.51 mg/kg ww is associated with a 15% decrease in growth in a freshwater midge species (*Chironomus decorus*), but this effect is not likely to result in population-level effects (Vandermeer and Goldberg 2003). It is also unclear whether the tissue concentration represents the actual effect level, because while tissue concentrations greater than the selected LOAEL TRV (as high as 0.89 mg/kg ww) are reported after 48 hrs of exposure, no effect on growth is reported. Furthermore, the LOAEL of 0.51 mg/kg ww (2.55 mg/kg dry weight [dw], as reported in the original paper) is within the range of 0.40 to 4.5 mg/kg dw, the aquatic invertebrate selenium background concentration range reported by DOI (1998). The selected TRVs thus may overestimate the potential for adverse effects in invertebrates based on selenium tissue concentrations.

Fluorene and LPAH

No LPAH mixture studies were identified; instead the lowest LOAEL for a single LPAH was selected and used as a surrogate for LPAHs. Two studies examining mortality of single LPAHs (fluorene and naphthalene) were found to meet TRV acceptability criteria. Three LOAELs were identified for two species (two amphipods [*Hyalella azteca* and *Diporeia* sp.]) ranging from 111 to 2,040 mg/kg ww. An SSD was not constructed because data were for different LPAHs. The lowest LOAEL was 111,000 µg/kg ww for increased mortality in *H. azteca* following 10 days of exposure to aqueous fluorene was selected as the LOAEL TRV (Lee et al. 2002). No NOAEL was identified from this study, so the NOAEL was extrapolated from the LOAEL using a factor of 10. There is uncertaintly associated in the use of an extrapolation factor with the selected TRVs for fluorene due to the limited dataset (only two toxicity studies measuring mortality), and in comparing TRVs based on a single PAH (fluorene) to an LPRSA concentration based on an LPAH sum.

A LOAEL TRV of 780 µg/kg ww and a NOAEL TRV of 78 µg/kg ww were also selected for LPAH (Louis Berger et al. 2014) based on an eight-week chronic toxicity study of the polychaete *Nereis arenaceodentata* exposed to phenanthrene (Emery and Dillon 1996b). Two control groups were evaluated: a carrier control (actetone) group and a seawater control group. Reproductive endpoints of fecundity and juvenile production were studied. The LOAEL TRV of 780 µg/kg ww was associated with a 33%

[95] Value was converted from dry weight to wet weight assuming 80% moisture.

[96] NOAEL value is based on DL reported; concentration was not detected in midges with the dose associated with no adverse effect at 96 hrs.

decrease in fecundity and a 36% decrease in juvenile production relative to the carrier control (acetone) group, but was not different from the seawater control. Thus, there is uncertainty associated with whether an adverse effect would be expected. The NOAEL was extrapolated from the LOAEL using a factor of 10; there is uncertainty associated with the use of extrapolation factors to derive NOAELs.

Fluoranthene and HPAH

No HPAH mixture studies were identified, instead the lowest LOAEL for a single HPAH was selected and used as a surrogate for HPAHs. Eleven studies examining growth, reproduction, and mortality of single HPAHs (fluoranthene, benzo(a)pyrene and pyrene) were found to meet TRV acceptability criteria. Sixteen LOAELs were identified for seven species (two copepods [*Coullana* sp. and *Schizopera knabeni*], a freshwater oligochaete [*L. variegatus*], two amphipods [*Diporeia* sp. and *Hyalella azteca*], a midge [*Chironomus tentans*], and a polychaete [*Armandia brevis*]) ranging from 23 to 1,200 mg/kg ww. The lowest LOAEL was associated with a reduced length and reduced number of offspring of the amphipod *H. azteca* at a tissue concentration of 22,200 µg/kg ww following a 28-day exposure to aqueous fluoranthene (Schuler et al. 2007). No adverse effects were observed in this study at a tissue burden of 8,100 µg/kg ww, this value was selected as the NOAEL TRV. There is also uncertainty associated with comparing TRVs based on a single PAH (fluoranthene) to an LPRSA concentration based on an HPAH sum.

A LOAEL TRV of 660 µg/kg ww and a NOAEL TRV of 66 µg/kg ww were also selected (Louis Berger et al. 2014; USEPA 2017a) based on a study that observed adverse effects on gametogenesis in blue mussels after a five-week exposure period to fluoranthene in water at a concentration of 2 µg/L (Eertman et al. 1993; Eertman et al. 1995). Eertman et al. (1993) presented data for tissue concentrations after a four-week exposure period to fluoranthene concentrations of 0.5, 1, and 6 µg/L. The LOAEL TRV of 660 µg/kg ww was estimated from regression relationships developed for tissue concentrations, water concentrations, and length of exposure. The NOAEL TRV of 66 µg/kg ww was extrapolated from the LOAEL using a factor of 10. It is unclear what effect the impaired gametogenesis would have on population-level reproductive success. There is also uncertainty associated with estimating the LOAEL tissue concentration from data for different water concentrations and a shorter time period, comparing TRVs based on a single PAH (fluoranthene) to an LPRSA concentration based on an HPAH sum, and using an extrapolation factor to derive the NOAEL.

TEQ - Fish

Only one study was identified for 2,3,7,8-TCDD and TEQ that met TRV acceptability criteria. Ashley et al. (1996) reported that at a LOAEL of 3,000 ng/kg ww for survival of crayfish, a 25% mortality, was observed. 2,3,7,8-TCDD was administered using a single 1-mL/kg injection of TCDD dissolved in dimethyl sulfoxide (DMSO) and mixed with corn oil in an experiment lasting 40 to 60 days. No NOAEL was identified from this study, so a NOAEL of 300 ng/kg was estimated as the LOAEL divided by 10. There is uncertainty associated with the derivation of a NOAEL based on an extrapolation factor. There is high uncertainty associated with these TRVs. They are based on a 45-day laboratory study wherein wild-caught crayfish (*Pacifastacus leniusculus*) were dosed via cephalothoracic injection at three doses and a control, using three or four crayfish per treatment (Ashley et al. 1996). Tissue concentrations were not reported, only the dose that was injected. The NOAEL of 300 ng/kg and LOAEL of 3,000 ng/kg were identified based on lethality. The use of wild-caught animals, the lack of tissue concentrations, and the limited number of test organisms in each treatment introduces a large amount of uncertainty in the derived values. Additionally, the measured endpoint—mortality—is generally not the most sensitive endpoint based on chemical exposure.

A LOAEL TRV of 1.3 ng/kg ww and a NOAEL TRV of 0.15 ng/kg ww were also selected for 2,3,7,8-TCDD and TEQ - fish (Louis Berger et al. 2014) based on Wintermyer and Cooper (2003), a 10-month field study of adult eastern oysters transplanted to two locations (one at Arthur Kill [a Newark Bay estuary], and the other in Sandy Hook, New Jersey). The study examined reproductive endpoints measuring the success rate of egg fertilization and early development of those fertilized eggs in a 48-hr assay. Reported tissue concentrations were based on one composite of seven oysters from each site. The reproductive endpoint measuring early development of fertilized eggs was based on one sample and did not provide any measure of variability in tissue concentrations. The LOAEL TRV of 1.3 ng/kg ww was based on oysters deployed at the Arthur Kill site, where 23% fertilization success occurred, whereas the NOAEL TRV of 0.15 ng/kg ww was based on oysters deployed to Sandy Hook, where 54% fertilization success occurred. A true control group was not used for comparison. The no-effect threshold was based on a lower fertilization effect at the Sandy Hook site.

The TRVs associated with oyster reproduction provide a lower-bound estimate of invertebrate toxicity from exposure to 2,3,7,8-TCDD, while the TRVs associated with crayfish survival provide a higher-bound estimate of invertebrate toxicity from exposure to 2,3,7,8-TCDD. The selected LOAELs span three orders of magnitude (from 1.3 to 3,000 ng/kg), indicating that 1) there is a wide range of toxicity and effects among benthic invertebrate species, and/or 2) there is a range of uncertainty associated with PCDD/PCDF toxicity data for benthic invertebrates. The high uncertainty associated with both ends of this range should be considered in the comparison of LPRSA tissue concentrations of invertebrates to evaluate the potential for risk.

Total PCBs

Nine studies examining growth, reproduction, and mortality endpoints of PCBs were found to meet TRV acceptability criteria. For 10 invertebrate species (3 species of shrimp [*Penaeus aztecus*, *Palaemontes pugio*, and *Penaeus duorarum*], 2 species of amphipod [*Gammarus psuedolimnaeus* and *D. magna*], 2 species of arthropod [*Limulus polyphemus* and *Chironomus riparius*], 2 species of polychaete [*Armandia brevis* and *Nereis diversicolor*], and 1 species of bivalve [*C. virginica*]), 16 LOAELs were identified, ranging from 1,100 to 552,000 µg/kg ww, using both aqueous and sediment-based exposures. An ACR of

8.4 was applied to 5 acute LOAEL values to derive chronic LOAEL values (USEPA 1980d) (Table 6-19), and the range of LOAELs including ACR-adjusted values was 130 to 552,000 µg/kg ww. An SSD was developed using both chronic and ACR-derived LOAELs (Figure 6-28). The best-fit distribution curve was described by a gamma distribution. The 5th percentile of the SSD (520 µg/kg ww) was selected as the LOAEL TRV (Figure 6-28). This SSD-derived LOAEL is less than the lowest measured LOAEL reported from the literature: a tissue residue of 1,100 µg/kg ww was associated with 33% mortality in grass shrimp after 96 hrs of exposure to aqueous PCB Aroclor 1016 (Hansen et al. 1974) (Appendix E). The geometric mean of ACR-adjusted LOAELs for grass shrimp was 650 µg/kg ww,[97] also greater than the SSD-derived LOAEL. Thus, the SSD-derived LOAEL represents a conservatively extrapolated value that is less than those empirically measured in the reviewed toxicity studies. The NOAEL TRV (52 µg/kg ww) was extrapolated from the LOAEL TRV using an uncertainty factor of 10. There is uncertainty associated with the derivation of a NOAEL based on an extrapolation factor.

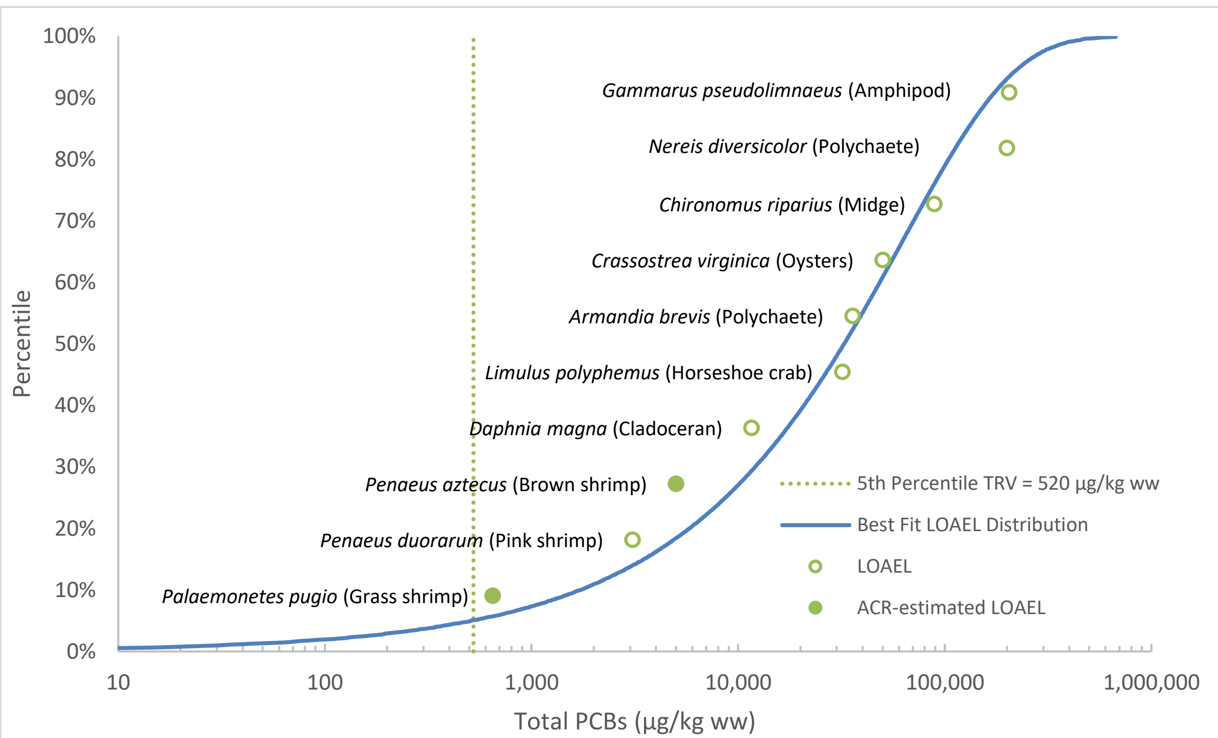


Figure 6-28. Invertebrate whole-body tissue SSD of total PCBs

A LOAEL TRV of 17 µg/kg ww and a NOAEL TRV of 6.4 µg/kg ww were also selected (Louis Berger et al. 2014) based on two studies (Chu et al. 2000; Chu et al. 2003). The

[97] Appendix E presents the toxicity data for grass shrimp; two acute LOAELs of 1,100 and 27,000 µg/kg ww were available (ACR-adjusted LOAELs were 130 and 3,200 µg/kg ww, respectively). The geomean of these two ACR-adjusted LOAELs is 650 µg/kg ww.

first study examined PCB uptake and accumulation in eastern oysters exposed for 30 days to an algal diet contaminated with PCB Aroclors, and measured total PCB accumulation within the oysters (Chu et al. 2000). Exposure concentrations of 0.1 and 0.1 µg/L were associated with egg tissue concentrations of 100 and 671 µg/kg dw (20 and 134 µg/kg ww), respectively. The second study examined PCB accumulation and adverse reproductive effects measured by number of spawned oysters after 76 days of exposure to 0.35 µg/L, and no-adverse-effect on reproduction after 30 days of exposure to 0.10 µg/L PCBs (Chu et al. 2003). An extrapolated LOAEL egg tissue concentration of 52 µg/kg ww was derived for the exposure of 0.35 µg/L using a regression based on the exposure concentrations (0.1 and 1 µg/L) and egg tissue concentrations (20 and 134 µg/kg ww) reported by Chu et al. (2000). The egg tissue LOAEL and NOAEL of 52 and 20 µg/kg ww, respectively, were then converted to an adult tissue LOAEL and NOAEL of 17 and 6.4 µg/kg ww, respectively, based on the adult:egg lipid ratio of 0.25:0.08.

There is uncertainty associated with the selected TRVs. Chu et al. (2003) noted that no dose-responsive relationship was observed among the females that had spawned. In addition, it should be noted that no PCB analysis was conducted on eggs in the present study, and that Chu et al. (2003) stated that PCB concentrations in their study might have exceeded those found in Chu et al. (2000). These studies used different doses and exposure conditions and assumed a linear relationship between dose and egg tissue.

Dieldrin

Two studies examining mortality were found to meet TRV acceptability criteria. Acute-based LOAELs were available for four species (a midge [*Chironomus riparius*], American oyster [*Crassostrea virginica*], pink shrimp [*P. duorarum*], and grass shrimp [*Palaemonetes pugio*]), and were therefore insufficient to develop an SSD. Parrish et al. (1973) reported an acute LOAEL of 80 µg/kg ww for pink shrimp, wherein 25% mortality was observed after 96 hrs of exposure to dieldrin in water. Whole-body tissues of pink shrimp with 0% mortality were below DLs (< 10 µg/kg ww), so an acute NOAEL of 8 µg/kg ww was derived as the LOAEL divided by 10. There is uncertainty associated with the derivation of a NOAEL based on an extrapolation factor.

A LOAEL TRV of 8.0 µg/kg ww and a NOAEL TRV of 1.6 µg/kg ww were also selected (Louis Berger et al. 2014) based on data from Parrish et al. (1973) and an applied ACR. The LOAEL of 8.0 µg/kg ww used an ACR of 10. The NOAEL of 1.6 µg/kg ww was based on the tissue residues reported in the control group (16 µg/kg ww) divided by an ACR of 10; tissue residues were below DLs (< 10 µg/kg ww) in the lowest treatment level group with 0% mortality. There is uncertainty associated with the use of an extrapolation factor to derive the TRV.

Total DDx

Eight studies examining mortality, growth, and reproductive endpoints were found to meet TRV acceptability criteria. Eleven LOAELs were identified for six species (three amphipod species [*D. magna*, *H. azteca*, and *Leptocherius plumulosus*], two polychaete species [*Armandia brevis* and *Neanthes arenaceodentata*], and one decapod crustacean [*P. duorarum*]), with LOAELs ranging from 130 to 266,000 µg/kg ww. Five of these studies were aqueous exposures, five were sediment-based exposures, and one was a diet-based exposure. An ACR of 3.6 was applied to four acute LOAEL values to derive chronic LOAEL values (Raimondo et al. 2007) (Table 6-19), and the range of LOAELs including ACR-adjusted values was 130 to 74,000 µg/kg. An SSD was developed using both chronic and ACR-derived LOAELs (Figure 6-29). Too few chronic studies were available to derive a total DDx SSD without ACR-estimated chronic LOAELs. There is uncertainty associated with the use of ACR-estimated chronic LOAELs. Using @Risk, the gamma distribution was determined to best fit the final species LOAELs (Figure 6-29). This determination was based on several goodness-of-fit statistics;[98] the gamma distribution also had a better visual fit to the SSD data than did the beta-general distribution. The 5th percentile of the SSD was selected as the LOAEL TRV (110 µg/kg ww). The NOAEL TRV (11 µg/kg ww) was extrapolated from the LOAEL TRV using an uncertainty factor of 10. There is uncertainty associated with the derivation of a NOAEL based on an extrapolation factor. Additionally, consistent with 2017 communications between CPG and USEPA, an alternative distribution (beta general) was selected to derive an alternative 5th percentile LOAEL TRV (10 µg/kg ww) (Figure 6-29) as a conservative SSD-derived estimate. An alternative NOAEL (1.0 µg/kg ww) was extrapolated from the alternative 5th percentile LOAEL TRV using an uncertainty factor of 10. There is uncertainty associated with the derivation of a NOAEL based on an extrapolation factor. This distribution appears to have a shape similar to that of the LOAEL values, and closely matches the low tail of the empirical dataset. However, the beta general distribution does not visually fit the data as well as the gamma distribution (e.g., without bias), nor can goodness-of-fit statistics be calculated to substantiate the beta general distribution. Both TRVs are included herein to bracket the uncertainty associated with selecting a single theoretical distribution.

[98] Standard statistics produced by @Risk include the A-D, chi-squared, AIC, BIC, and K-S statistics.

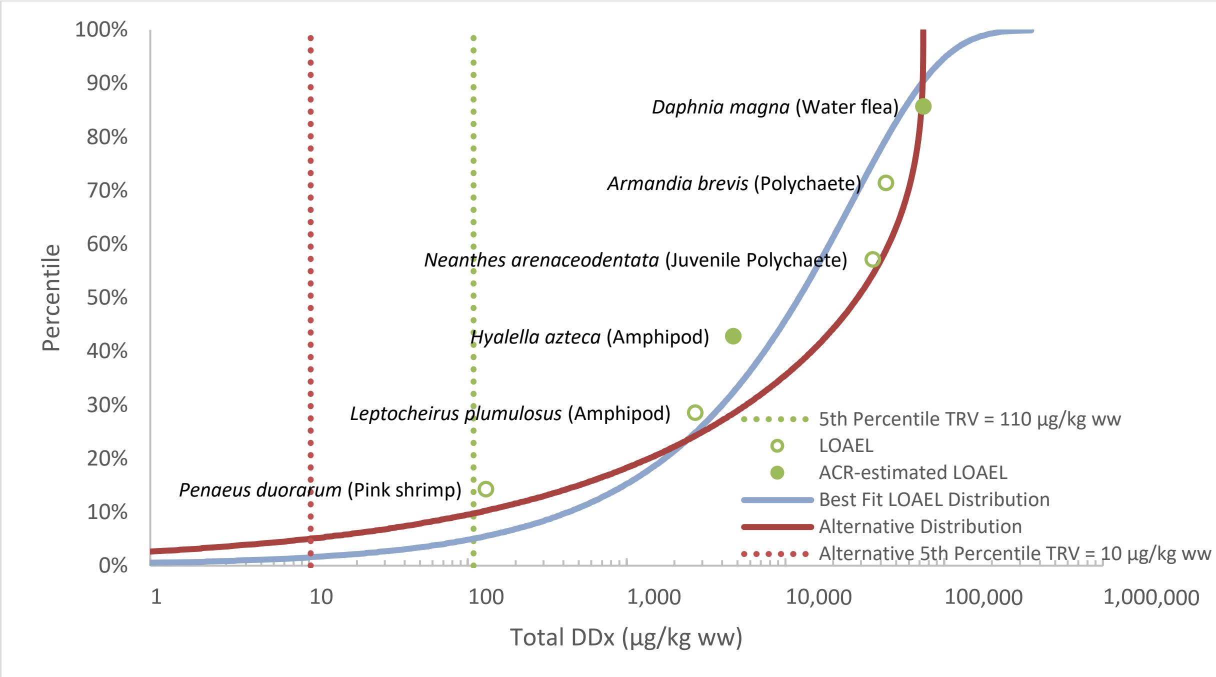


Figure 6-29. Invertebrate whole-body tissue total DDx SSD toxicity data

The SSD-derived LOAEL of 110 µg/kg ww is similar to but less than the lowest measured LOAEL reported from the literature: a tissue residue of 130 µg/kg ww was associated with 33% mortality in pink shrimp after 56 days of exposure to aqueous DDx (Nimmo et al. 1970) (Appendix E). The alternative SSD-derived LOAEL of 10 µg/kg ww is an order of magnitude less than this lowest measured LOAEL. Thus, the alternative SSD-derived LOAEL of 10 µg/kg ww represents a conservatively extrapolated value that is much less than those empirically measured in the reviewed toxicity studies.

The LOAEL TRV of 130 µg/kg ww based on Nimmo et al. (1970) was also selected (Louis Berger et al. 2014). The LOAEL was based on the body burdens of pink shrimp that had died at day 28 of the experiment.

Heptachlor Epoxide

One study examining mortality was found to meet TRV acceptability criteria (Schimmel et al. 1976). Three LOAELs were available for three species (two shrimp species [*P. duorarum* and *P. vulgaris*] and one bivalve species [*C. virginica*]), and data were therefore insufficient to develop an SSD. These studies reported LOAEL values ranging from 140 to 2,500 µg/kg ww. Schimmel et al. (1976) exposed organisms to aqueous heptachlor epoxide for 96 hrs. The lowest LOAEL among the three test species was 140 µg/kg ww, which resulted in 30% mortality to *C. virginica*, compared to 13% mortality in the control; this value was selected as the LOAEL TRV. The NOAEL for *C. virginica* from the same study (10 µg/kg ww) was selected as the NOAEL TRV; this value was uncertain given that it was based on a DL. There is uncertainty associated with both TRVs, given the limited dataset (one study) used to evaluate a severe effect (mortality).

Regulated Metals

Arsenic

Seven studies examining growth and mortality effects in seven species of invertebrate were found to meet TRV acceptability criteria. These studies reported seven LOAEL values ranging from 0.63 to 92 mg/kg ww (Figure 6-30). An ACR of 3.8 was applied to three acute LOAEL values to derive chronic LOAELs (USEPA 1985a) (Table 6-19). An SSD was developed using both chronic LOAELs and ACR-derived LOAELs (Figure 6-30). Six of the studies used aqueous arsenic exposures; the remaining study was a combined diet- and water-based exposure. Study duration ranged from 96 hrs to 30 days. The distribution of final species LOAELs was best described by a Levy distribution. The 5th percentile of the SSD was selected as the LOAEL TRV (0.64 mg/kg ww) (Figure 6-30). The SSD-derived LOAEL was similar to the lowest LOAEL value derived from the literature: a residue of 0.63 mg/kg ww associated with reduced growth in mayflies after 12 days of exposure (Irving et al. 2008) (Appendix E2). The NOAEL TRV (0.064 mg/kg ww) was extrapolated from the LOAEL TRV using an uncertainty factor of 10. There is uncertainty associated with the derivation of a NOAEL based on an extrapolation factor. There is uncertainty associated with the use of an extrapolation factor to derive the NOAEL. No TRVs for arsenic were available in the revised FFS (Louis Berger et al. 2014) or draft FFS (Malcolm Pirnie 2007b).

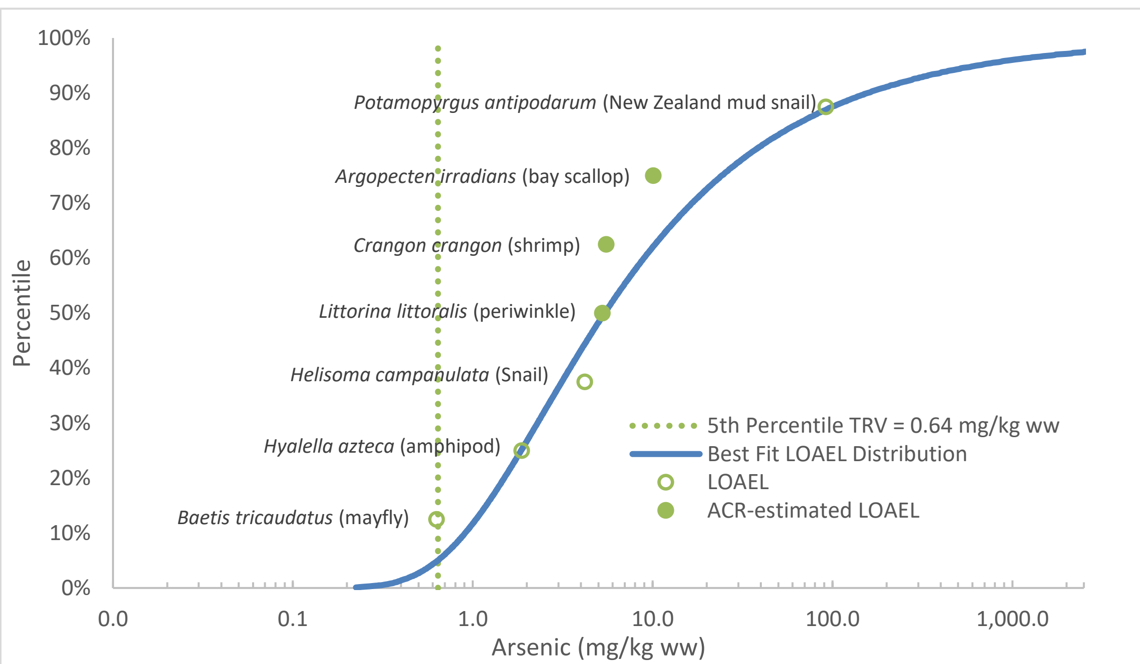


Figure 6-30. Invertebrate whole-body tissue arsenic SSD toxicity data

Cadmium

Sixty studies examining effects on growth, reproduction, and survival for 40 species of invertebrate were found to meet TRV acceptability criteria and were subsequently used to derive a cadmium SSD (Figure 6-31). These studies reported LOAEL values ranging from 0.2 to 3,400 mg/kg ww. An ACR of 9.1 (AWQC final saltwater ACR, equivalent to Raimondo et al. (2007) median metals ACR) was applied to 21 acute LOAEL values to derive chronic LOAEL TRVs (USEPA 2001) (Table 6-19); the range of LOAELs, including ACR-adjusted values, was 0.02 to 3,400 mg/kg ww. An SSD was developed using both chronic and ACR-derived LOAELs (Figure 6-31). Forty-eight of the studies used water-based cadmium exposures, eight used diet-based exposures, four used sediment-based exposures, and one used a combined diet- and water-based exposure. Study duration ranged from 24 hrs to 37 weeks. The best fit was described by a log-logistic distribution (Figure 6-31). The 5[th] percentile of the SSD (0.24 mg/kg ww) was selected as the LOAEL TRV. This SSD-derived LOAEL is similar to the lowest measured LOAEL reported from the literature: a tissue residue of 0.2 mg/kg ww associated with reduced reproduction in water fleas after two weeks of exposure (Sofyan et al. 2007) (Appendix E). The NOAEL TRV (0.024 mg/kg ww) was extrapolated from the LOAEL TRV using an uncertainty factor of 10. There is uncertainty associated with the derivation of a NOAEL based on an extrapolation factor.

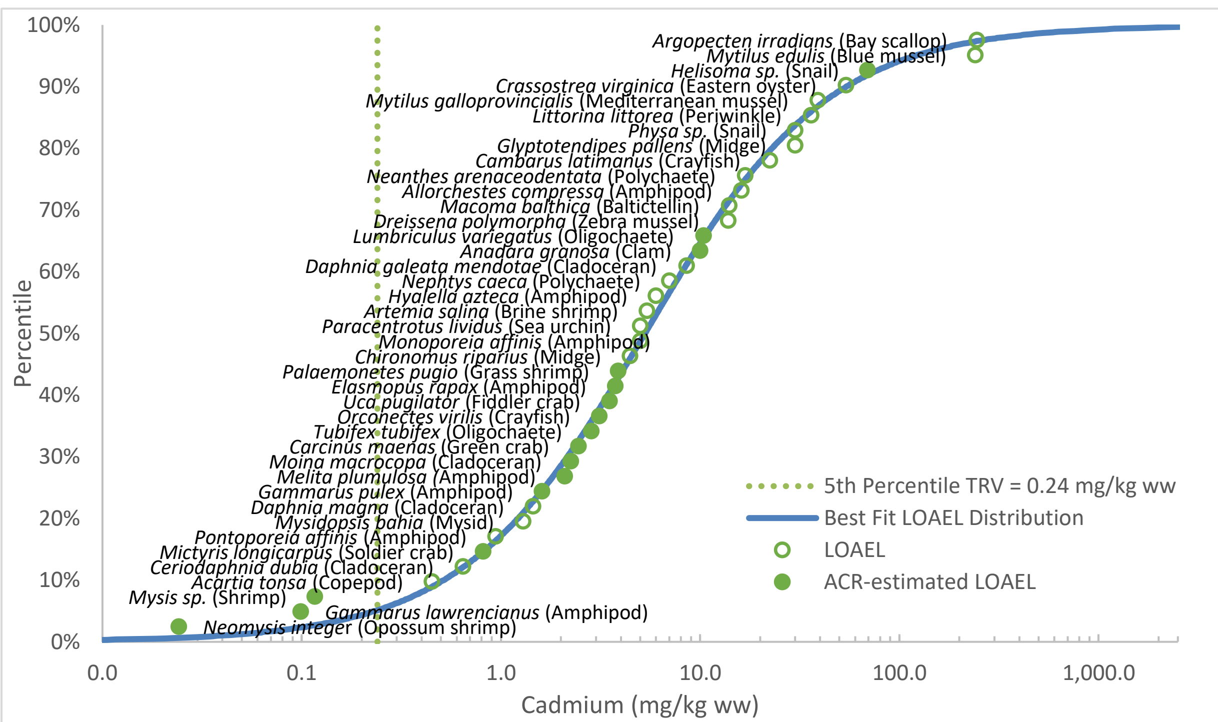


Figure 6-31. Invertebrate whole-body tissue cadmium SSD toxicity data

The highest NOAEL below the selected LOAEL TRV was from a study that exposed a cladoceran to aqueous cadmium for seven days (Sofyan et al. 2007). A whole-body concentration of 0.12 mg/kg ww was associated with no significant effect on reproduction; this value was selected as the NOAEL TRV.

There is high uncertainty associated with the evaluation of tissue residues of cadmium. Cadmium can be sequestered in detoxified forms away from the site of action, contributing to uncertainty about the relationship between whole-body concentration and adverse toxic effects (Amiard et al. 1987). Exposure conditions such as metal bioavailability, exposure route, or exposure time all contribute to the regulation, bioaccumulation, and fraction of metabolically active cadmium causing toxicity, resulting in a high degree of uncertainty in the tissue residue effects threshold (Adams et al. 2011).

No TRVs for cadmium were available in the revised FFS (Louis Berger et al. 2014) or draft FFS (Malcolm Pirnie 2007b).

Chromium

Two studies examining mortality and reproductive effects were found to meet TRV acceptability criteria. LOAELs were available for only two species or types of invertebrates (an amphipod [*H. azteca*] and a polychaete [*Neanthes arenaceodentata*]), and data were therefore insufficient to develop an SSD. These studies reported LOAEL values that ranged from 3.5 to 6.0 mg/kg ww. Norwood et al. (2007) reported the lowest LOAEL of 3.5 mg/kg ww, which was selected for TRV derivation. Norwood et al. (2007) exposed *H. azteca* to water- and diet-based chromium for four weeks. The selected LOAEL and NOAEL values for chromium corresponded to the LC50 and LC25 (concentration that is lethal to 25% of an exposed population) values reported by Norwood et al. (2007). The mortality rate of the LC25 was within the range of control mortality, so this value was selected as the NOAEL TRV. A LOAEL of 3.5 mg/kg ww (assuming 80% moisture) and a NOAEL of 1.5 mg/kg ww (assuming 80% moisture) were identified. There is uncertainty associated with the limited toxicity dataset for chromium (two studies).

No TRVs for chromium were available in the revised FFS (Louis Berger et al. 2014) or draft FFS (Malcolm Pirnie 2007b).

Cobalt

One study examining mortality was found to meet acceptability criteria. LOAELs were available for only one species (an amphipod [*H. azteca*]), and data were therefore insufficient to develop an SSD. In this study, Norwood et al. (2007) exposed *H. azteca* to water-based cobalt for four weeks. A LOAEL of 2.6 mg/kg ww (assuming 80% moisture) and a NOAEL of 1.2 mg/kg ww (assuming 80% moisture) were identified and selected as TRVs (Norwood et al. 2007). There is uncertainty associated with the limited toxicity dataset for cobalt (one study).

No TRVs for cobalt were available in the revised FFS (Louis Berger et al. 2014) or draft FFS (Malcolm Pirnie 2007b).

Copper

Copper tissue burdens associated with adverse effects on invertebrates have been shown to be a function of accumulation rate, internal sequestration, and detoxification mechanism, among other factors (Rainbow 2002). Furthermore, copper is an essential nutrient for invertebrates, which regulate their tissue burdens via a variety of mechanisms (Rainbow 2007).

Rainbow (2007) summarized recent studies of the copper and zinc requirements of invertebrates. Based on theoretical calculations of the enzyme requirements for copper, Rainbow (2007) estimated a whole-body copper requirement of 26.3 mg/kg (assumed dry weight; approximately 5 mg/kg ww) for invertebrates. Some invertebrate taxa, particularly mollusks and crustaceans, contain hemocyanin (a copper-based respiratory pigment) instead of hemoglobin. Thus, the copper requirement these some mollusks and crustaceans is greater than that of other invertebrates. Rainbow (2007) estimated the whole-body copper requirement of the shrimp *Pandalus montagui* to be 38.1 µg/g, including hemocyanin (assumed dry weight; approximately 7.5 mg/kg ww). Rainbow (2007) also reported tissue burdens in barnacles, amphipods, and decapod crustaceans (shrimps and crabs) from uncontaminated sites, ranging from 14.9 to 77.3 mg/kg dw (approximately 5 to 15 mg/kg ww, assuming 20% moisture content). These findings are supported by a study by Lee and Shiau (2002) on tiger prawn (*Penaeus monodon*), a common marine aquaculture species, the respiratory pigment of which is hemocyanin. In *P. monodon*, copper deficiency is seen at 7.27 mg/kg and sufficiency at 7.5 to 9 mg/kg ww. Studies on *D. magna* (Bossuyt and Janssen 2003; Lam and Wang 2008), which has hemoglobin as its respiratory pigment, have shown copper deficiency at between 0.16 and 1.1 mg/kg ww, and copper sufficiency at between 2.0 and 14 mg/kg ww. Based on these studies, a copper nutritional threshold of 7.5 mg/kg ww was identified. No appropriate LOAELs were identified for the derivation of an invertebrate tissue copper TRV.

A LOAEL TRV of 12 mg/kg ww and NOAEL TRV of 5 mg/kg ww were selected (Louis Berger et al. 2014) based on increased mortality of clam following chronic (40-day) aqueous copper exposure (Absil et al. 1996). The higher threshold TRVs were documented as derived from dry weight tissue concentrations (assuming 80% moisture) at day 40, associated with 46 and 0% mortality for the LOAEL and NOAEL, respectively (Louis Berger et al. 2014).

Lead

Three studies examining mortality were found to meet TRV acceptability criteria. LOAELs were available for only three species (two amphipods [*H. azteca* and *Gammarus psuedolimnaeus*] and a decapod crustacean [*Penaeus indicus*]), and data were therefore insufficient to develop an SSD. These studies reported LOAEL values ranging from 40 to 200 mg/kg ww. Spehar et al. (1978) reported the lowest LOAEL of 40 mg/kg ww after exposing the amphipod *G. psuedolimnaeus* to water-based lead concentrations for 28 days; this study was selected for TRV derivation. A LOAEL of 40 mg/kg ww was reported and selected as the LOAEL TRV. No NOAEL was identified in the study, so the NOAEL TRV (4.0 µg/kg ww) was extrapolated from the LOAEL TRV using an uncertainty factor of 10. There is uncertainty associated with the derivation of a NOAEL based on extrapolation factors, and there is a limited dataset for lead toxicity to invertebrates.

A NOAEL and LOAEL of 0.52 and 2.6 mg/kg ww, respectively, were also selected for lead (Louis Berger et al. 2014) based on increased mortality of the amphipod *H. azteca* in a four-week spiked sediment toxicity test (Borgmann and Norwood 1999). The LOAEL TRV was derived from the reported LOAEL of 5.2 mg/kg ww by applying an interspecies extrapolation factor of 2 to the LC25 (Borgmann and Norwood 1999). The NOAEL TRV was derived by extrapolating by a factor of 5 from the LOAEL TRV. There is uncertainty associated with the derivation of a NOAEL based on an extrapolation factor. Additionally, the use of field-collected sediment from the western basin of Hamilton Harbor creates uncertainty surrounding the presence of additional metals within the exposure sediment (as reported: 12 mg/kg dw cobalt, 1198 mg/kg dw chromium, 101 mg/kg dw copper, 45,400 mg/kg dw iron, 1,720 mg/kg dw manganese, 47 mg/kg dw nickel, 113 mg/kg dw lead, and 1,240mg/kg dw zinc). The presence of additional metals or other contaminants could have played a role in observed effects.

Nickel

One study examining growth and mortality was found to meet TRV acceptability criteria. LOAELs were available for only one species or types of invertebrates (an amphipod [*H. azteca*], and therefore data were insufficient to develop an SSD. In this study, Borgmann et al. (2001) exposed *H. azteca* to nickel-spiked field-collected sediments for 28 days. A LOAEL of 1.1 mg/kg ww (assuming 80% moisture) was identified, associated with 75% increased mortality; a NOAEL of 0.10 mg/kg ww (assuming 80% moisture) was also identified from the study. These values were selected as TRVs. There is uncertainty associated with the selected TRVs, as there is a very limited dataset for nickel toxicity to invertebrates (one study). In addition, uncertainty due to the use of field-collected sediments in the study should also be considered; Borgmann et al. (2001) did not report pre-spiked sediment chemistry, so the presence of additional metals or other contaminants could have played a role in observed effects.

No TRVs were available in the revised FFS (Louis Berger et al. 2014) or draft FFS (Malcolm Pirnie 2007b).

Silver

One study examining growth and reproduction was found to meet TRV acceptability criteria. LOAELs were available for only one species or types of invertebrates (an amphipod [*D. magna*]), and data were therefore insufficient to develop an SSD. In this study, Naddy et al. (2007) exposed *D. magna* to water-based silver for seven days. A LOAEL of 0.59 mg/kg ww associated with reduced growth and reproduction and a

NOAEL of 0.49 mg/kg ww were identified; these values were selected as TRVs. There is uncertainty associated with the limited toxicity dataset for silver (one study).

No TRVs were available in the revised FFS (Louis Berger et al. 2014) or draft FFS (Malcolm Pirnie 2007b).

Vanadium

No acceptable TRVs were identified. Two studies presented NOAEL values, but these values were not selected for the derivation of a NOAEL TRV. NOAELs identified by Miramand et al. (1981) and Miramand et al. (1982) were from bioaccumulation studies of aqueous exposure to vanadium of four species of marine invertebrates (a crab, a shrimp, a sea cucumber, and a sea urchin species) for three weeks (Miramand et al. 1981; Miramand et al. 1982). The highest reported tissue concentration was 0.80 mg/kg ww for sea cucumber. However, the experimental protocols did not investigate adverse effects and none were reported.

No TRVs were available in the revised FFS (Louis Berger et al. 2014) or draft FFS (Malcolm Pirnie 2007b).

Zinc

Zinc tissue burdens associated with adverse effects in invertebrates have been shown to be a function of duration and route of exposure, among other factors (Rainbow 2002). Furthermore, zinc is an essential nutrient for invertebrates, which are able to regulate their tissue burdens via a variety of mechanisms (Rainbow 2007).

Rainbow (2007) reviewed and summarized the recent studies of nutritional zinc requirements of invertebrates. Based on these studies, Rainbow (2007) estimated a whole-body zinc requirement of 34.5 mg/kg dw (approximately 6.9 mg/kg ww). This value was identified as the nutritional threshold for worms and bivalves.

The lowest chronic LOAEL for bivalves greater than the zinc nutritional threshold was selected as the bivalve-specific LOAEL TRV. King et al. (2004) reported a LOAEL of 80 mg/kg ww associated with 15% mortality of *M. anomala* during a 96-hr acute zinc toxicity test. No NOAEL was available from this study, so a NOAEL TRV of 8.0 mg/kg ww was derived using an uncertainty factor of 10. There is uncertainty associated with the derivation of a NOAEL based on an extrapolation factor.

The only identified toxicity datum for polychaetes was a NOAEL of 80 mg/kg ww for *Nephtys australiensis* exposed to aqueous zinc for a 96-hr acute toxicity test (King et al. 2004). This value was selected as the worm-specific NOAEL TRV. There is uncertainty associated with the use of an unbounded NOAEL in assessing potential risks.

Rainbow (2007) reported tissue burdens in amphipods and crustaceans (shrimps and crabs) from uncontaminated sites as ranging from 57.5 to 481 mg/kg dw (approximately 12 to 96 mg/kg ww assuming 80% moisture content). This range is confirmed in similar studies by Lam and Wang (2008) and Muyssen and Janssen (2002) that examined zinc-sufficient diets of *D. magna*; these studies reported whole-body tissue concentrations of 20 mg/kg ww (converted from dry weight assuming 80% moisture) and 45 mg/kg ww, respectively. Based on this information, 45 mg/kg ww was identified as the nutritional threshold for crustaceans. The lowest chronic LOAEL for crustaceans greater than the zinc nutritional threshold was selected. Muyssen et al. (2006) reported a LOAEL of 51 mg/kg ww associated with reduced *D. magna* survival during a 21-day aqueous zinc exposure. No NOAEL was available from this study, so a NOAEL TRV (5.1 mg/kg ww) was extrapolated from the LOAEL TRV using an uncertainty factor of 10. There is uncertainty associated with the derivation of a NOAEL using an extrapolation factor.

No TRVs for zinc were available in the revised FFS (Louis Berger et al. 2014) or draft FFS (Malcolm Pirnie 2007b).

6.3.4 Risk characterization

This section presents the tissue HQs for benthic invertebrates, as well as uncertainties associated with the HQ calculations. Invertebrate tissue data were not available from the Passaic River above Dundee Dam, Jamaica Bay/Lower Harbor, or Mullica River/Great Bay, so a background comparison could not be conducted.

6.3.4.1 Tissue HQs

Invertebrate species (blue crab, mussel, and worm) tissue LOAEL HQs are presented in Table 6-22. Appendix G lists EPCs, TRVs, and calculated HQs for the benthic invertebrate tissue COPECs in a single table (Table G1). LOAEL HQs for whole-body blue crab were ≥ 1.0 for mercury, methylmercury, selenium, total PCBs, 2,3,7,8-TCDD, PCDD/PCDF TEQ, and total TEQ using a range of TRVs. LOAEL HQs for whole-body mussels were ≥ 1.0 for total HPAHs, 2,3,7,8-TCDD, PCDD/PCDF TEQ, and total TEQ using a range of TRVs. LOAEL HQs for worms were ≥ 1.0 for selenium, total HPAHs, total PCBs, 2,3,7,8-TCDD, PCDD/PCDF TEQ, and total TEQ using a range of TRVs. LOAEL HQs for all species were < 1.0 for total LPAHs, PCB TEQ, dieldrin, heptachlor epoxide, and total DDx using a range of LOAEL TRVs. Invertebrate species tissue NOAEL HQs are presented in Table 6-22, as are LOAEL and NOAEL HQs for blue crab hepatopancreas and muscle tissue; LOAEL and NOAEL HQs were < 1.0 for both tissue types.

Table 6-22. Invertebrate tissue LOAEL and NOAEL HQs

COPEC	Range of Invertebrate Tissue HQs[a,b]							
	HQ Based on TRV-A[c]					HQ Based on TRV-B[d]		
	Blue Crab			Mussels	Worms	Blue Crab	Mussels	Worms
	Whole Body	Hepato-pancreas	Muscle	Whole Body		Whole Body		
LOAEL HQs								
Metals								
Mercury	**1.5**	0.067	ne	0.084	0.62	**1.5**	0.084	0.62
Methyl mercury	**1.3**	0.049	ne	0.034	0.031	**1.3**	0.034	0.031
Selenium	**1.5**	ne	ne	0.10	**1.1**	na	na	na
PAHs								
Total HPAHs	0.0050	ne	ne	0.0099	0.090	0.17	0.33	**3.0**
Total LPAHs	0.00075	ne	ne	0.00074	0.0049	0.11	0.11	0.69
PCBs								
Total PCBs	0.67	ne	ne	0.046	0.46	**21**	**1.4**	**14**
PCB TEQ - fish	0.00026	ne	ne	7.0×10^{-6}	6.7×10^{-5}	0.6	0.016	0.15
PCDD/PCDFs								
2,3,7,8-TCDD	0.019	ne	ne	0.00073	0.013	**44**	**1.7**	**29**
PCDD/PCDF TEQ - fish	0.021	ne	ne	0.00077	0.013	**48**	**1.8**	**29**
Total TEQ - fish	0.021	ne	ne	0.00077	0.013	**48**	**1.8**	**30**
Pesticides								
Dieldrin	0.085	ne	ne	0.034	0.020	0.85	0.34	0.2
Heptachlor epoxide	0.045	ne	ne	0.0069	0.0026	na	na	na
Total DDx	0.62 (**6.8**[e])	ne	ne	0.048 (0.53[e])	0.15 (**1.6**[e])	0.52	0.041	0.12
NOAEL HQs								
Metals								
Mercury	**2.9**	0.67	0.59	0.17	**1.2**	**2.9**	0.17	**1.2**
Methyl mercury	**2.5**	0.49	0.56	0.067	0.06	**2.5**	0.067	0.06

COPEC	Range of Invertebrate Tissue HQs[a,b]							
	HQ Based on TRV-A[c]					HQ Based on TRV-B[d]		
	Blue Crab			Mussels	Worms	Blue Crab	Mussels	Worms
	Whole Body	Hepato-pancreas	Muscle	Whole Body		Whole Body		
Selenium	**16**	ne	ne	**1.0**	**11**	na	na	na
PAHs								
Total HPAHs	0.014	ne	ne	0.027	0.25	**1.7**	**3.3**	**30**
Total LPAHs	0.0075	ne	ne	0.0075	0.049	**1.1**	**1.1**	**6.9**
PCBs								
Total PCBs	**6.7**	ne	ne	0.46	**4.6**	**55**	**3.8**	**38**
PCB TEQ - fish	0.0026	ne	ne	0.00007	0.00067	**5.2**	0.14	1.3
PCDD/PCDFs								
2,3,7,8-TCDD	0.19	ne	ne	0.0073	0.13	**380**	**15**	**250**
PCDD/PCDF TEQ - fish	0.21	ne	ne	0.0077	0.13	**410**	**15**	**250**
Total TEQ - fish	0.21	ne	ne	0.0077	0.13	**420**	**15**	**260**
Pesticides								
Dieldrin	0.85	ne	ne	0.34	0.20	**4.3**	**1.7**	**1.0**
Heptachlor epoxide	0.63	ne	ne	0.096	0.036	na	na	na
Total DDx	**6.2** (**68**[e])	ne	ne	0.48 (**5.3**[e])	**1.5** (**16**[e])	**1.1**	0.088	0.27

Bold identifies HQs ≥ 1.0.

Shaded cells identify HQs ≥ 1.0 based on a LOAEL TRV.

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

b HQs were based on EPCs from Table 6-18 and TRVs presented in Table 6-20.

c TRVs were derived from the primary literature review based on the process identified in Section 6.3.3.1.

d TRVs were based on USEPA's revised draft of the LPR restoration project FFS (Louis Berger et al. 2014) or first draft of the LPR restoration project FFS (Malcolm Pirnie 2007b).

e HQs in parenthesis were based on additional alternative SSD-derived LOAELs evaluated (see text in Section 6.3.3.2 for details).

COPEC – chemical of potential ecological concern
DDD – dichlorodiphenyldichloroethane
ne – not evaluated
NOAEL – no-observed-adverse-effect level

DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
EPC – exposure point concentration
FFS – focused feasibility study
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
LPAH – low-molecular-weight polycyclic aromatic hydrocarbon
LPR – Lower Passaic River
na – not applicable
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
PCDD– polychlorinated dibenzo-p-dioxin
PCDF –polychlorinated dibenzofuran
SSD – species sensitivity distribution
TCDD – tetrachlorodibenzo-p-dioxin
TEQ – toxic equivalent
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT and 4,4′-DDT)
TRV – toxicity reference value
USEPA – US Environmental Protection Agency

6.3.4.2 Uncertainties in risk characterization

COPEC-specific TRV uncertainties are discussed in Section 6.3.3. There are a number of uncertainties associated with selected TRVs; COPEC-specific TRV uncertainties should be carefully considered when determining risk conclusions.

Uncertainty associated with the EPCs is considered low, since a sufficient number of detected samples were available to derive UCLs for all COPEC-benthic invertebrate pairs. However, since TEQs are based on fish TEFs (because no TEFs exist for benthic invertebrates), the uncertainty associated with TEQs is high. There is limited evidence for ligand activation of the Ah (dioxin) cellular receptor in these organisms, so they are not susceptible to the dioxin-like effects reported for vertebrates (e.g., fish) (Van den Berg et al. 1998).

6.3.4.2 Tissue EFs for regulated metals

Invertebrate species (blue crab, mussel, and worm) tissue NOAEL and LOAEL exceedance factors (EFs) for regulated metals are presented in Table 6-23. Blue crab LOAEL EFs were ≥ 1.0 for arsenic, copper, and silver using a range of TRVs. Mussel LOAEL EFs were ≥ 1.0 for chromium and nickel using a range of TRVs. Worm LOAEL EFs were ≥ 1.0 for arsenic, chromium, lead, and nickel using a range of TRVs.

Table 6-23. Invertebrate tissue LOAEL and NOAEL EFs for regulated metals

COPEC	Range of EFs[a,b]					
	EF Based on TRV-A[c]			EF Based on TRV-B[d]		
	Blue Crab	Mussels[e]	Worms	Blue Crab	Mussels[e]	Worms
LOAEL EF						
Arsenic	**2.2**	0.0	**2.2**	na	na	na
Cadmium	0.46	0.038	0.46	na	na	na
Chromium	0.40	**3.7**	**6.0**	na	na	na
Cobalt	0.029	0.029	0.27	na	na	na
Copper	na[f]	na[f]	na[f]	**2.1**	0.029	0.48

COPEC	Range of EFs[a,b]					
	EF Based on TRV-A[c]			EF Based on TRV-B[d]		
	Blue Crab	Mussels[e]	Worms	Blue Crab	Mussels[e]	Worms
Lead	0.0090	0.0033	0.16	0.14	0.050	**2.5**
Nickel	0.91	**6.0**	**12**	na	na	na
Silver	**1.0**	0.0024	0.047	na	na	na
Vanadium	na[f]	na[f]	na[f]	na	na	na
Zinc	0.71	0.020	nc	na	na	na
NOAEL EF						
Arsenic	22	0.0	22	na	na	na
Cadmium	**4.6**	0.38	**4.6**	na	na	na
Chromium	0.93	**8.7**	**14**	na	na	na
Cobalt	0.063	0.063	0.58	na	na	na
Copper	na[f]	na[f]	na[f]	**4.9**	0.070	**1.2**
Lead	0.090	0.033	**1.6**	0.69	0.25	**12**
Nickel	**10**	**66**	**130**	na	na	na
Silver	**1.2**	0.0029	0.057	na	na	na
Vanadium	na[f]	na[f]	na[f]	na	na	na
Zinc	**7.1**	0.20	0.43	na	na	na

Bold identifies EFs ≥ 1.0.

Shaded cells identify EFs ≥ 1.0 based on a LOAEL TRV.

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

[b] HQs were based on EPCs from Table 6-18 and TRVs presented in Table 6-21.

[c] TRVs were derived from the primary literature review.

[d] TRVs were based on USEPA's revised draft of the LPR restoration project FFS (Louis Berger et al. 2014) or first draft of the LPR restoration project FFS (Malcolm Pirnie 2007b)

[e] Mussel EPCs were based on day 0-normalized concentrations.

[f] No TRVs based on the primary literature review were recommended for copper or vanadium; see Section 6.3.3.2 for details.

COPEC – chemical of potential ecological concern
EF – exceedance factor
EPC – exposure point concentration
LPR – Lower Passaic River
na – not applicable
NOAEL – no-observed-adverse-effect level

FFS – focused feasibility study
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
TRV – toxicity reference value
USEPA – US Environmental Protection Agency

6.3.5 Summary of key uncertainties

The primary uncertainty associated with the benthic invertebrate tissue risk characterization is the high uncertainty pertaining to the risk estimates for inorganic metals evaluated in this risk assessment. This uncertainty is due to the varying ways that invertebrates take up, bioaccumulate, and regulate metals within tissues. In addition, there are a limited number of toxicity studies available for several organic COPECs (i.e., selenium, 2,3,7,8-TCDD, dieldrin, heptachlor epoxide, and endosulfan). The applicability of so few values to the broader invertebrate community is uncertain. Specific uncertainties associated with TRVs are discussed in Section 6.3.3.1 and 6.3.3.2. For the COPECs with TRVs based on 5th percentile LOAELs determined from SSDs (i.e., total PCBs, total DDx, arsenic, and cadmium), the range of the empirical LOAELs and number of data points (i.e., number of species included in the SSD) are shown in Table 6-24 to provide context of uncertainty for SSD-derived values.

Table 6-24. Uncertainty evaluation of invertebrate tissue TRVs based on SSDs

COPEC	TRV[a] Unit (ww)	TRV[a] NOAEL	TRV[a] LOAEL	No. of Species (count of LOAELs in SSD)	No. ACR-adjusted LOAELs /No. LOAELs in SSD	Empirical LOAEL Range	Notes on Key Uncertainties
Total PCBs	µg/kg	400	520	n = 10	2 / 10	1,100–552,000	SSD-derived LOAEL < lowest measured LOAEL
Total DDx	µg/kg	60	110 (10[b])	n = 6	2 / 6	130–266,000	SSD-derived LOAEL and alternative SSD-derived LOAEL both < lowest measured LOAEL
Arsenic	mg/kg	0.064[c]	0.64	n = 7	3 / 7	0.63–92	SSD-derived LOAEL within range of measured LOAELs
Cadmium	mg/kg	0.12	0.24	n = 29	16 / 29	0.2–3,400	SSD-derived LOAEL within range of measured LOAELs

Note: TRVs included in this table are based on SSDs that are based on TRVs derived from the general literature search.

a TRVs were derived from the primary literature review based on the process identified in Section 6.3.3.1.

b An alternative SSD distribution was also selected based on a conservative distribution fit; see Section 6.3.3.2 for additional description.

c NOAEL extrapolated from LOAEL using an uncertainty factor of 10.

ACR – acute-to-chronic ration
COPEC – chemical of potential ecological concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
SSD – species sensitivity distribution
TRV – toxicity reference value
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT and 4,4′-DDT)
ww – wet weight

6.3.6 Summary

Of the COPECs evaluated in whole-body tissue for three species of benthic invertebrate, seven had LOAEL HQs ≥ 1.0 for blue crab, four had LOAEL HQs ≥ 1.0 for mussels, and six had LOAEL HQs ≥ 1.0 for worms, all using a range of TRVs (Table 6-25). No LOAEL or NOAEL HQs were ≥ 1.0 for blue crab hepatopancreas and muscle tissue.

Table 6-25. Summary of invertebrate tissue LOAEL HQs

COPEC[b]	Range of LOAEL HQs[a]						Key Uncertainties
	HQ Based on TRV-A[c]			HQ based on TRV-B[d]			
	Blue Crab[e]	Mussels[e]	Worms[e]	Blue Crab[e]	Mussels[e]	Worms[e]	
Mercury	**1.5**	0.084	0.62	**1.5**	0.084	0.62	• TRVs based on limited dataset (3 studies)
Methylmercury	**1.3**	0.034	0.031	**1.3**	0.034	0.031	
Selenium	**1.5**	0.10	**1.1**	na	na	na	• TRV-A based on 15% reduction in growth and on a limited dataset (1 study); TRV-A within the range of aquatic invertebrate selenium background concentrations reported by DOI (1998)
Total HPAHs	0.0050	0.0099	0.090	0.17	0.33	**3.0**	• Both TRVs based on individual PAH (fluoranthene) • TRV-B based on impaired gametogenesis
Total PCBs	0.67	0.046	0.46	**21**	**1.4**	**14**	• TRV-A based on SSD less than lowest measured LOAEL evaluated • TRV-B based on whole-body tissue concentrations interpolated from measured egg tissue concentrations
2,3,7,8-TCDD	0.019	0.00073	0.013	**44**	**1.7**	**29**	• TRV-A based on injected (not measured) concentration in crayfish • TRV-B based on uncontrolled field data and limited sample size (n=1 tissue composite); LOAEL based on relative reduction at Arthur Kill site compared to Sandy Hook site • Evaluation as TEQ (based on fish TEFs) questionable for invertebrates because of limited evidence for ligand activation of the Ah (dioxin) cellular receptor in these organisms; as a result, they are not susceptible to the dioxin-like effects reported for vertebrates (Van den Berg et al. 1998).
PCDD/PCDF TEQ - fish	0.021	0.00077	0.013	**48**	**1.8**	**29**	
Total TEQ - fish	0.021	0.00077	0.013	**48**	**1.8**	**30**	
Total DDx	0.62 (**6.8**[e])	0.048 (0.53[e])	0.15 (**1.6**[e])	0.52	0.041	0.12	• TRV-A and alternative TRV-A based on SSDs less than lowest measured LOAEL • Alternative TRV-A based on relatively poor visual and statistical fit to the empirical data and likely overestimates toxicity

Bold identifies HQs ≥ 1.0.

[a] The NJDEP acknowledges that the BERA for the LPRSA identifies unacceptable risk. However, the NJDEP's *Ecological Evaluation Technical Guidance*, August 2018 (NJDEP 2018), does not advocate the use of more than one set of TRVs for individual contaminant-receptor pairs. It is the NJDEP's position that a single TRV set (NOAEL and LOAEL) that evaluates the more sensitive species and endpoints to characterize risk to invertebrates, fish, birds and wildlife should be selected in a BERA, not two sets of TRVs as presented in this document. It is the NJDEP's position that, for the LPRSA, use of one

conservative TRV set derived for sensitive receptors and sensitive endpoints most clearly demonstrates the degree of risk for individual contaminant-receptor pairs and ensures protection of threatened, endangered, and species of special concern.

[b] Only COPECs with HQs ≥ 1.0 based on a LOAEL TRV are included in the table.

[c] TRVs were derived from the primary literature review.

[d] TRVs were based on USEPA's revised draft of the LPR restoration project FFS (Louis Berger et al. 2014) or first draft of the LPR restoration project FFS (Malcolm Pirnie 2007b).

[e] Whole-body tissue data.

[f] HQs in parenthesis were based on additional alternative SSD-derived LOAEL evaluated (see text in Section 6.3.3.2 for details).

Ah – aryl hydrocarbon

COPEC – chemical of potential ecological concern

DDD – dichlorodiphenyldichloroethane

DDE – dichlorodiphenyldichloroethylene

DDT – dichlorodiphenyltrichloroethane

FFS – focused feasibility study

HPAH – high-molecular-weight polycyclic aromatic hydrocarbon

HQ – hazard quotient

LOAEL – lowest-observed-adverse-effect level

LPR – Lower Passaic River

na – not applicable

nc – not calculated

PAH – polycyclic aromatic hydrocarbon

PCB – polychlorinated biphenyl

PCDD – polychlorinated dibenzo-*p*-dioxin

PCDF – polychlorinated dibenzofuran

SSD – species sensitivity distribution

TCDD – tetrachlorodibenzo-*p*-dioxin

TEF – toxic equivalency factor

TEQ – toxic equivalent

total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT and 4,4′-DDT)

TRV – toxicity reference value

USEPA – US Environmental Protection Agency

Risk estimates for regulated metals are presented in Table 6-26. There is high uncertainty associated with the risk estimates for inorganic metals due to the varying ways invertebrates uptake, bioaccumulate, and regulate metals within tissues. Of the regulated metal COPECs evaluated in whole-body tissue for three species of benthic invertebrate, three had LOAEL EFs ≥ 1.0 for blue crab using a range of TRVs (Table 6-26). Two COPECs had LOAEL EFs ≥ 1.0 for mussels using a range of TRVs. Four COPECs had LOAEL EFs ≥ 1.0 for worms using a range of TRVs.

Table 6-26. Summary of invertebrate tissue LOAEL EFs for regulated metals

COPEC[b]	Range of LOAEL EFs [a]						Key Uncertainties
	EF Based on TRV-A[c]			EF Based on TRV-B[d]			
	Blue Crab[e]	Mussels[e]	Worms[e]	Blue Crab[e]	Mussels[e]	Worms[e]	
Arsenic	**2.2**	0.0	**2.2**	nc	nc	nc	• Tissue-residue approach not recommended for regulated metals[f] • TRV-A derived using SSD
Chromium	0.40	**3.7**	**6.0**	nc	nc	nc	• Tissue-residue approach not recommended for regulated metals[f] • TRV-A based on limited dataset (2 studies)
Copper	nc	nc	nc	**2.1**	0.029	0.48	• Tissue-residue approach not recommended for regulated metals[f]
Lead	0.0090	0.0033	0.16	0.14	0.050	**2.5**	• Tissue-residue approach not recommended for regulated metals[f] • TRV-A based on limited dataset (3 studies) • TRV-B based on field-collected sediment
Nickel	0.91	**6.0**	**12**	nc	nc	nc	• Tissue-residue approach not recommended for regulated metals[f] • TRV-A based on limited dataset (1 study) and on field-collected sediment
Silver	**1.0**	0.0024	0.047	nc	nc	nc	• Tissue-residue approach not recommended for regulated metals[f] • TRV-A based on limited dataset (1 study)

Bold identifies EFs ≥ 1.0.

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

[b] Only regulated metals with EFs ≥ 1.0 based on a LOAEL TRV are included in the table.

[c] TRVs were derived from the primary literature review.

[d] TRVs were based on USEPA's revised draft of the LPR restoration project FFS (Louis Berger et al. 2014) or first draft of the LPR restoration project FFS (Malcolm Pirnie 2007b).

[e] Whole-body tissue data.

[f] USEPA framework for metals risk assessment (USEPA 2007e) recommends against the use of a tissue residue approach, stating that the CBR approach for metals "does not appear to be a robust indicator of toxic dose."

CBR – critical body residue
COPEC – chemical of potential ecological concern
EF – exceedance factor
FFS – focused feasibility study
LOAEL – lowest-observed-adverse-effect level
LPR – Lower Passaic River
nc – not calculated
SSD – species sensitivity distribution
TRV – toxicity reference value
USEPA – US Environmental Protection Agency

6.4 Identification of Preliminary COCs, and Risk Conclusions

Based on the results of SQT WOE analysis (and the quantitative analysis of uncertainty), it appears that benthic invertebrate communities in 1 to 19% of the LPRSA have been potentially impacted by sediment contamination, and that communities in 29 to 75% of the LPRSA have not been impacted or have had low impacts. Moderate benthic invertebrate risk was determined to exist in 24 to 53% of the LPRSA; moderate risk may be due to moderate chemical impacts exacerbated by other comfounding factors (e.g., habitat). COCs are not being proposed based on the WOE analysis results.

The potential for risk to benthic invertebrates was also evaluated using the surface water and tissue LOEs. Benthic invertebrate (blue crab, *in situ* mussel, and worm from bioaccumulation testing) tissue and surface water concentrations were compared to TRVs based on the literature to derive risk estimates (HQs) in the risk characterization. COPECs and species pairs with effect-level HQs ≥ 1.0 (based on an acute or chronic TRV for surface water, or a LOAEL TRV for tissue and diet) in at least one LOE were proposed as preliminary COCs (Tables 6-27 and 6-28).

Table 6-27. Summary of preliminary COCs for benthic invertebrates

Preliminary COC[b]	Range of LOAEL HQs[a]							
	Invertebrate Tissue LOE						Surface Water LOE	
	LOAEL HQ Based on TRV-A[c]			LOAEL HQ Based on TRV-B[d]			HQ Based on Estuarine TRVs[e]	HQ Based on Freshwater TRVs[e]
	Blue Crab	Mussels	Worms	Blue Crab	Mussels	Worms		
Mercury	**1.5**	0.084	0.62	**1.5**	0.084	0.62	0.0067 (acute), 0.013 (chronic)	0.013 (acute), 0.086 (chronic)
Methylmercury	**1.3**	0.034	0.031	**1.3**	0.034	0.031	not a COPEC	
Selenium	**1.5**	0.10	**1.1**	na	na	na	0.0017 (acute), 0.0069 (chronic)	0.16 (chronic)
Cyanide	ne[f]			ne[f]			**1.3** (acute), **4.1** (chronic)	0.23 (acute), **1.0** (chronic)
Total HPAHs	0.0050	0.0099	0.090	0.17	0.33	**3.0**	not a COPEC	
Total PCBs	0.67	0.046	0.46	**21**	**1.4**	**14**	0.0072 (acute), 0.21 (chronic)	0.032 (acute), 0.14 (chronic)
2,3,7,8-TCDD	0.019	0.00073	0.013	**44**	**1.7**	**29**	0.0028 (acute), **4.3** (chronic)	0.034 (acute), 0.14 (chronic)
PCDD/PCDF TEQ - fish	0.021	0.00077	0.013	**48**	**1.8**	**29**	not a COPEC	
Total TEQ - fish	0.021	0.00077	0.013	**48**	**1.8**	**30**	not a COPEC	
Total DDx	0.62 (**6.8**[g])	0.048 (0.53[g])	0.15 (**1.6**[g])	0.52	0.041	0.12	0.12 (acute), 0.22 (chronic)	0.015 (acute), 0.027 (chronic)

Bold identifies HQs ≥ 1.0.

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

b Only COPECs with HQs ≥ 1.0 based on a LOAEL TRV are included in this table.

c HQs for tissue were based on TRVs derived from the primary literature review.

d HQs for tissue were based on TRVs based on USEPA's revised draft of the LPR restoration project FFS (Louis Berger et al. 2014) or first draft of the LPR restoration project FFS (Malcolm Pirnie 2007b).

e HQs for surface water were derived using EPCs based on UCLs.

f Cyanide was not evaluated using the tissue LOE; this chemical was not analyzed in LPRSA tissue.

g HQs in parenthesis were based on an additional alternative SSD-derived LOAEL evaluated (see text in Section 6.3.3.2 for details).

BLM – biotic ligand model
COC – chemical of concern
COPEC – chemical of potential ecological concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
EPC – exposure point concentration
FFS – focused feasibility study
LPRSA – Lower Passaic River Study Area
ne – not evaluated
PCB – polychlorinated biphenyl
PCDD – polychlorinated dibenzo-*p*-dioxin
PCDF – polychlorinated dibenzofuran
SSD – species sensitivity distribution
TCDD – tetrachlorodibenzo-*p*-dioxin
TEQ – toxic equivalent

HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
LOE – line of evidence
LPR – Lower Passaic River
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT and 4,4′-DDT)
TRV – toxicity reference value
UCL – upper confidence limit on the mean
USEPA – US Environmental Protection Agency

Table 6-28. Summary of regulated metals preliminary COCs for benthic invertebrates

	Range of LOAEL EFs/HQs[a]							
	Invertebrate Tissue LOE						Surface Water LOE	
	LOAEL EF Based on TRV-A[c]			LOAEL EF Based on TRV-B[d]			HQ Based on Estuarine TRVs[e]	HQ Based on Freshwater TRVs[e]
Preliminary COC[b]	**Blue Crab**	**Mussels**	**Worms**	**Blue Crab**	**Mussels**	**Worms**		
Arsenic	**2.2**	0.0[c]	**2.2**	nc	nc	nc	not a COPEC	
Chromium	0.40	**3.7**	**6.0**	nc	nc	nc	0.00084 (acute), 0.0018 (chronic)	0.075 (acute), 0.11 (chronic)
Copper	nc	nc	nc	**2.1**	0.029	0.48	0.14 (acute), **2.7** (chronic)	0.034 (acute), **1.0** (chronic)
Lead	0.0090	0.0033	0.16	0.14	0.050	**2.5**	0.014 (acute), 0.14 (chronic)	< 0.001–0.034 (acute), 0.012–0.67 (chronic)
Nickel	0.91	**6.0**	**12**	nc	nc	nc	not a COPEC	
Silver	**1.0**	0.0024	0.047	nc	nc	nc	0.0034 (acute), 0.0095 (chronic)	0.015 (acute), 0.039 (chronic)

Bold identifies HQs ≥ 1.0.

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

[b] Only COPECs with HQs ≥ 1.0 based on a LOAEL TRV are included in this table.

[c] HQs for tissue were based on TRVs derived from the primary literature review.

[d] HQs for tissue were based on TRVs based on USEPA's revised draft of the LPR restoration project FFS (Louis Berger et al. 2014) or first draft of the LPR restoration project FFS (Malcolm Pirnie 2007b).

[e] HQs for surface water were derived using EPCs based on UCLs, except for copper and lead in which HQs were calculated based on individual water samples (because the BLM-based TRVs were sample-specific).

BLM – biotic ligand model
COC – chemical of concern
COPEC – chemical of potential ecological concern
EPC – exposure point concentration
EF – exceedance factor
FFS – focused feasibility study
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
LPR – Lower Passaic River
nc – not calculated
TRV – toxicity reference value
UCL – upper confidence limit on the mean

The results of this invertebrate risk assessment will be used in the FS as a tool for risk managers to make potential remedial decisions for the LPRSA. Determining the potential for unacceptable ecological risk at the population level provides information pertaining to decisions to be made in the FS or other programmatic environmental management framework. The TRVs used to evaluate risks to invertebrates in this BERA are organism-level effects, including those that affect particular attributes of organisms within a population, such as survival, growth, and reproduction. Survival, growth, and reproduction of individual organisms may, in turn, affect the populations of those organisms, depending upon the magnitude and severity of the effect. However, population-level effects—such as size or density of population, population growth, or population survival—are more direct measures of influences on the population as a whole. USEPA guidance states that assessment endpoints should be associated with sustaining the ecological structure and function of populations and communities rather than individual organisms, unless individuals warrant additional protection in specific cases (USEPA 1999). Since BERAs evaluate ***populations*** as assessment endpoints, not individuals, a number of other factors, including the potential magnitude and severity of the effect, should be assessed to determine if risk drivers (defined and identified in Section 13) should be used in developing PRGs and remedial action levels (RALs).

7 Fish Assessment

This section presents the risk assessment for fish species in the LPRSA. The risk assessment for fish evaluated the following assessment endpoint, according to the PFD (Windward and AECOM 2009):

- **Assessment Endpoint No. 5 --** Protection and maintenance (i.e., survival, growth, and reproduction) of omnivorous, invertivorous, and piscivorous fish populations that serve as a forage base for fish and wildlife populations and as a base for sports fisheries.

The potential for risks to fish was characterized quantitatively using four LOEs that evaluated COPECs identified in the SLERA, as follows:

- **Tissue LOE –** comparison of COPEC concentrations in fish tissue to tissue TRVs
- **Dietary LOE –** comparison of COPEC concentrations in fish diet to dietary TRVs
- **Surface water LOE –** comparison of COPEC concentrations in surface water to TRVs
- **Fish egg tissue LOE –** comparison of modeled COPEC concentrations in mummichog egg tissue to TRVs

In addition, several qualitative LOEs involved the evaluation of LPRSA data for mummichog egg counts and gross external and internal health observations. COPECs with calculated HQs ≥ 1.0 were assessed to determine a list of preliminary COCs.

All fish species were evaluated in the tissue and dietary LOEs. The fish species identified in the problem formulation (Section 3) included three general feeding groups: benthic omnivores, invertivores, and piscivores. In accordance with the PFD (Windward and AECOM 2009), at least one fish species was selected for each estuarine and freshwater area (Table 7-1). Fish movement in the LPRSA generally follows the movement of the salt wedge (Table 7-1). Freshwater fish (i.e., common carp, channel cafish, brown bullhead, white sucker, largemouth bass, northern pike, and smallmouth bass) were absent from the lower two reaches of the LPRSA (i.e., below RM 4). Estuarine fish (including white perch and American eel) were generally found throughout the LPRSA, with the exception of mummichog, which were primarily collected in reaches below RM 12.

Table 7-1. Fish species evaluated in the BERA

Species Feeding Group	Species Type	Species Evaluated	2009/2010 Survey Observations
Benthic omnivore	estuarine	mummichog	mummichog collected primarily below RM 12
	freshwater	other forage fish	targeted species infrequently caught; other forage fish caught above RM 6 used as a surrogate forage fish species
		common carp	collected only above RM 4
Invertivore	estuarine	white perch	throughout the LPRSA
	freshwater	channel catfish	collected only above RM 8
		brown bullhead	collected only above RM 6
		white catfish	collected only above RM 2
		white sucker	collected only above RM 6
Piscivore	estuarine/ migratory	American eel	throughout the LPRSA
	freshwater	largemouth bass	collected only above RM 6
		northern pike	collected only above RM 8
		smallmouth bass	collected only above RM 6

LPRSA – Lower Passaic River Study Area
RM – river mile
USEPA – US Environmental Protection Agency

The fish risk assessment process is outlined in Table 7-2. Sections 7.1, 7.2, 7.3, and 7.4 present the fish tissue, dietary, surface water, and fish egg assessments, respectively. Uncertainties associated with various components of these assessments are discussed throughout their respective sections, and key uncertainties are summarized at the end of each section. Sections 7.5 and 7.6 present the qualitative assessments based on mummichog egg counts and fish health observations, respectively. Section 7.7 identifies fish preliminary COCs, which are further evaluated in Section 13.

Table 7-2. Outline of the fish risk assessment

Section Number	Section Title	Section Contents
7.1	Tissue Assessment	for each LOE, presents COPECs based on the SLERA, exposure and effects data, HQs, uncertainty discussion, and summary of risk characterization
7.2	Dietary Dose Assessment	
7.3	Surface Water Assessment	
7.4	Egg Tissue Assessment	
7.5	Mummichog Egg Assessment	presents results of fish mummichog egg count evaluation
7.6	Health Assessment	presents observations of fish health conducted during field studies
7.7	Identification of Preliminary COCs	identifies preliminary COCs

COC – chemical of concern
COPEC – chemical of potential ecological concern
HQ – hazard quotient
LOE – line of evidence
SLERA – screening-level ecological risk assessment

7.1 Tissue Assessment

The tissue assessment was conducted for all fish species feeding groups: benthic omnivores, invertivores, and piscivores. Tissue chemistry EPCs for the fish species (i.e., mummichog, other forage fish [surrogate for banded killifish/darter], common carp, white perch, channel catfish, brown bullhead, white sucker, white catfish, American eel, largemouth bass, smallmouth bass, and northern pike) were compared to whole-body tissue TRVs to calculate HQs. This section summarizes the COPECs identified from the SLERA, describes the derivation of tissue exposure and effects concentrations, presents the HQs, and summarizes the uncertainties associated with the tissue assessment.

7.1.1 COPECs

COPECs for fish tissue were identified in the SLERA (Section 5) as COIs with maximum concentrations equal to or exceeding their screening-level TRVs (Table 7-3). In the SLERA, COPECs were screened by species; any chemical identified as a COPEC for any species was evaluated for all species.

Table 7-3. Fish tissue COPECs

COPEC	
Metals	
Arsenic	Methylmercury/mercury [a]
Cadmium	Selenium
Chromium	Silver
Copper	Zinc
Lead	

COPEC	
PAHs	
Total HPAHs[b]	Total LPAHs[b]
PCBs	
Total PCBs	PCB-TEQ - fish
PCDDs/PCDFs	
2,3,7,8-TCDD	Total TEQ - fish
PCDD/PCDF TEQ - fish	
Organochlorine Pesticides	
Dieldrin	Total DDx
Endosulfan I	

Note: COPECs are those COIs for which the maximum concentration exceeded its TSV. If a TSV was exceeded based on any fish species evaluated in the SLERA, it was retained as a COPEC for all fish.

[a] All but one toxicity test with mercury presented results in terms of total mercury in tissue. However, methylmercury was also evaluated because most of the mercury in fish tissue is present in the organic form (Bloom 1992; Grieb et al. 1990), and methylmercury is the form of mercury most toxic to fish (Sandheinrich and Wiener 2011).

[b] Because PAHs are rapidly metabolized and excreted by fish following uptake, whole-body PAH tissue concentrations do not provide a good measure of the dose at the site of toxic action; thus tissue-based exposure and effects data are not predictive of risks. A screening-level evaluation of PAHs in fish tissue was included in the SLERA, but PAH COPECs in fish tissue are not further evaluated in this BERA (USEPA 2015b, c, 2016g).

BERA – baseline ecological risk assessment
COI – chemical of interest
COPEC – chemical of potential ecological concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
LPAH – low-molecular-weight polycyclic aromatic hydrocarbon
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
PCDD– polychlorinated dibenzo-*p*-dioxin
PCDF – polychlorinated dibenzofuran
SLERA – screening-level ecological risk assessment
TCDD – tetrachlorodibenzo-*p*-dioxin
TEQ – toxicity equivalent
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT, and 4,4′-DDT)
TSV – toxicity screening value
USEPA – US Environmental Protection Agency

A number of COIs could not be screened as part of the SLERA (Appendix A) because no tissue screening levels were available. These COIs are presented in Section 5.5.2, as are the implications of not being able to evaluate these COIs.

7.1.2 Exposure

EPCs were calculated for COPECs identified for each fish species evaluated (or group evaluated, in the case of "other forage fish"). EPCs were calculated as UCLs using all available whole-body tissue data (i.e., individual and composite; calculated whole-body samples) for each fish type. UCLs were calculated using USEPA's ProUCL® statistical package (Version 5.0.00) (USEPA 2013d) as described in Section 4.3.7.[99] If a dataset contained fewer than six detected concentrations, a UCL was not calculated; instead, the maximum concentration was used as the EPC. UCLs could not be calculated for white sucker (n = 5 samples), largemouth bass (n = 3 samples), smallmouth bass (n = 3 samples), or northern pike (n=1 sample), because of the limited numbers of samples available for these species. Therefore, maximum concentrations were used as the EPCs. There is uncertainty associated with risk estimates for species with a very small number of samples. COPEC summary concentrations of fish tissue samples are presented in Appendix C. Uncertainties associated with the use of non-detects in calculations of total PCBs and TEQs - fish are discussed in Section 7.1.4.3. A summary of fish tissue EPCs is presented in Table 7-4.

[99] The UCL recommended by USEPA's ProUCL® software (typically the 95% UCL, but in some cases the 97.5% or even the 99% UCL) was used. The selected UCL statistic is presented in Appendix C1.

Table 7-4. Summary of fish tissue EPCs

COPEC	Unit (ww)	Benthic Omnivores			Invertivore					Piscivore			
		Mummichog	Other Forage Fish	Common Carp	White Perch	Channel Catfish	Brown Bullhead	White Catfish	White Sucker	American Eel	Largemouth Bass	Northern Pike	Smallmouth Bass
Metals													
Arsenic	mg/kg	0.38	0.34	0.15	0.22	0.054	0.13	0.086	0.11[a]	0.28	0.068 [a]	0.12[a]	0.25[a]
Cadmium	mg/kg	0.045	0.058	0.032	0.014	0.014	0.027	0.014	0.013[a]	0.088	0.037[a]	0.0035[a]	0.0089[a]
Chromium	mg/kg	8.7	61	2.8	4.4	0.44	0.78	0.73	2.0 [a]	2.5	0.23[a]	1.1[a]	0.51[a]
Copper	mg/kg	3.1	4.1	1.1	14	1.3	0.86	0.68	1.1[a]	2.6	0.58[a]	0.57[a]	0.80[a]
Lead	mg/kg	2.4	3.0	0.79	0.44	0.3	0.80	0.75	0.30[a]	0.87	0.12[a]	0.033[a]	0.098[a]
Mercury	µg/kg	63	83	80	200	150	110	280	140	260	680	220	300
Methylmercury	µg/kg	53	70	62	170	140	92	250	130[a]	280	520[a]	180[a]	220[a]
Selenium	mg/kg	0.72	0.70	0.82	1.4	0.31	0.77	0.38	0.46[a]	0.77	0.59[a]	0.55[a]	0.69[a]
Silver	mg/kg	0.044	0.046	0.015[a]	0.2	0.014[a]	0.008[a]	0.0033	0.0050[a]	0.025	0.0026[a]	0.0028[a]	0.0028[a]
Zinc	mg/kg	45	36	75	26	20	29.5	17	21[a]	31	16[a]	34[a]	18[a]
PCBs													
Total PCBs	µg/kg	600	550	5,200	2,500	1,700	1,400	3,400	2,900[a]	2,000	7,900[a]	2,000[a]	1,400[a]
PCB TEQ - fish[b]	ng/kg	0.62	0.62	4.4	2.1	1.8	1.3	3.5	3.2[a]	1.2	17[a]	2.3[a]	1.4[a]
PCDD/PCDF													
2,3,7,8-TCDD	ng/kg	49	46	610	190	96	150	210	130[a]	23	180[a]	95[a]	76[a]
PCDD/PCDF TEQ - fish[b]	ng/kg	51	49	620	200	100	160	220	130[a]	24	180[a]	100[a]	76[a]
Total TEQ - fish[b]	ng/kg	51	49	620	200	100	160	230	130[a]	25	180[a]	110[a]	82[a]
Pesticides													
Dieldrin	µg/kg	11	16	55	31	47	30	27	25[a]	54	40[a]	43[a]	20[a]
Endosulfan I	µg/kg	0.8[a]	1.5[a]	4.0[a]	0.22[a]	2[a]	1.1[a]	1.7[a]	0.45[a]	0.59[a]	10[a]	2.1[a]	2.8[a]
Endosulfan II	µg/kg	1.3[a]	1.5[a]	2.7[a]	3.7[a]	0.89[a]	1.6[a]	2.5[a]	1.1[a]	0.56[a]	3.3[a]	5.0[a]	2.8[a]

COPEC	Unit (ww)	Benthic Omnivores			Invertivore					Piscivore			
		Mummichog	Other Forage Fish	Common Carp	White Perch	Channel Catfish	Brown Bullhead	White Catfish	White Sucker	American Eel	Largemouth Bass	Northern Pike	Smallmouth Bass
Total DDx	µg/kg	66	75	650	240	280	160	350	150[a]	260	160[a]	280[a]	230[a]

Note: The UCL was selected as the EPC, except where noted.

[a] Fewer than six detected concentrations were available, so the HQ was based on a maximum concentration rather than a UCL concentration.

[b] TEQ calculated using the Kaplan-Meier approach.

COPEC – chemical of potential ecological concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
EPC – exposure point concentration
HQ – hazard quotient
PCB – polychlorinated biphenyl
PCDD– polychlorinated dibenzo-*p*-dioxin
PCDF –polychlorinated dibenzofuran
TCDD – tetrachlorodibenzo-*p*-dioxin
TEQ – toxic equivalent
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT, and 4,4′-DDT)
UCL – upper confidence limit on the mean
ww – wet weight

7.1.3 Effects

This section presents the effects data (i.e., TRVs) selected from the toxicological literature for the COPECs that were screened into this BERA. These TRVs are consistent with the comments, responses, and directives received from USEPA on June 30, 2017 (USEPA 2017b), September 18, 2017 (USEPA 2017e), July 10, 2018 (USEPA 2018), January 2, 2019 (CDM 2019), and March 5, 2019 (USEPA 2019); during face-to-face meetings or conference calls on July 24, September 27, October 3, November 6, 2017, July 24, 2018, and August 16, 2018; and via additional deliverables and communications between the CPG and USEPA from August through December 2017, July through September 2018, and January through June 2019. A range of TRVs was evaluated. The following subsections describe the overall methods used to identify TRVs.

7.1.3.1 Methods for selecting TRVs

The following subsections describe the general methods used to derive TRVs for fish tissue.

TRV Selection Process

Two sets of fish tissue TRVs were used for the derivation of HQs in this BERA. One set was based on previous documents developed by USEPA Region 2 for the LPRSA:

- USEPA's revised draft of the LPR restoration project FFS (Louis Berger et al. 2014)
- USEPA's first draft of the LPR restoration project FFS (Malcolm Pirnie 2007b)

The second set of TRVs was selected by first conducting a literature search for relevant toxicological studies, as described in Appendix E. These studies were then evaluated for acceptability of use. For those studies considered acceptable, NOAEL and LOAEL tissue TRVs were determined.

TRV Derivation Based on SSDs

When sufficient data were available (i.e., data for at least five species), TRVs were generated using an SSD approach. The LOAEL TRV was selected as the 5th percentile of the SSD, and the NOAEL TRV was extrapolated from the LOAEL TRV using an uncertainty factor of 10. When data were insufficient, the lowest acceptable LOAEL and highest NOAEL below the LOAEL from the same study were selected as TRVs.

An SSD is a statistical model that can be used to calculate a chemical concentration protective of a predetermined percentage of a group of species. SSDs are intended to provide an indication of both the total range and distribution of species sensitivities in natural communities, even when the actual range of sensitivities is unknown (Stephan 2002). In practice, SSDs are most commonly presented as a CDF of the toxicity of a chemical to a group of laboratory test species.

All toxicity data for various fish species meeting the TRV selection criteria were considered in constructing the SSDs. LOAELs represent the lowest concentrations at which an adverse effect is observed, whereas NOAELs indicate the concentration at which no adverse effect is observed. However, HQs greater than or equal to 1.0 based on NOAELs do not indicate whether an adverse effect can be expected. Therefore, LOAELs were considered appropriate for developing SSDs to determine the potential for an adverse effect. For each chemical, a single effects threshold (the final species LOAEL) was determined for inclusion in the SSD considering all acceptable LOAELs for that species.

For studies reporting acute LOAELs (i.e., mortality endpoints with < 28 days of observation and no growth or reproduction data reported in the same study), chronic LOAELs as inputs into the SSD dataset were estimated using ACRs for some COPECs (Table 7-5). ACRs for mercury and total PCBs were based on those reported in the AWQC derivation document (USEPA 1985c, 1980d). Only a single ACR (65) was identified in the AWQC document for DDx. Raimondo et al. (2007) reported ACRs ranging from 3 to 5 (median 3.6) in four studies of chemicals with a DDT-like mode of action. Because it is based on several studies and is therefore more reliable, the Raimondo et al. (2007) median ACR for chemicals with a DDT-like mode of action was used to estimate chronic LOAELs. The ACR for 2,3,7,8-TCDD was the geometric mean of all ACRs reported in Raimondo et al. (2007). Raimondo et al. (2007) evaluated ACRs based on 456 same-species pairs of acute concentrations and MATCs for metals, narcotics, pesticides, and other organic chemicals. Uncertainty associated with the application of ACRs to acute data and the potential effect on the SSD dataset are discussed on a COPEC-specific basis in Section 7.1.3.2.

Table 7-5. Chemical-specific ACRs applied to acute fish tissue LOAELs

COPEC	ACR	Source
Cadmium	9.106	Raimondo et al. (2007)
Mercury	3.731	USEPA (1985c)
Total PCBs	8.4	USEPA (1980d)
Total DDx	3.6	Raimondo et al. (2007)
2,3,7,8-TCDD	8.3	Raimondo et al. (2007)

ACR – acute-to-chronic ratio
COPEC – chemical of potential ecological concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
LOAEL – lowest-observed-adverse-effect level
PCB – polychlorinated biphenyl
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT and 4,4′-DDT)
USEPA – US Environmental Protection Agency

When multiple studies were evaluated the same species, the data were processed before being incorporated into the SSD. For any given toxicological endpoint (i.e., survival, growth, or reproduction), the geometric mean of all chronic LOAELs for that endpoint was calculated to determine the final endpoint value. If LOAELs for multiple endpoints were available, the lowest value among the endpoints was selected. For example, if toxicological data for survival and growth were reported in multiple studies for a particular species, first the geometric mean of all survival data and the geometric mean of all growth data were independently calculated, then the lower of the survival and growth geometric means was selected as the final species LOAEL.

After final species LOAELs were calculated for each species, final species LOAELs were ranked from lowest to highest, and the cumulative percent frequency value for each data point was calculated using Equation 7-1 (Stephan et al. 1985):

$$\mathrm{CPF} = \mathrm{Rank} \times \left(\frac{100}{n+1} \right) \qquad \textbf{Equation 7-1}$$

Where:

CPF = cumulative percent frequency
n = number of data points used to develop the SSD

The cumulative percent frequency value of each data point was then plotted against the final species LOAEL, yielding the typically S-shaped SSD plot with effect concentrations on the x-axis and cumulative frequency values on the y-axis.

Several theoretical distribution models were then fit to the final species LOAELs and their corresponding empirical cumulative frequency distributions using @RISK software. @Risk software provides rankings of several goodness-of-fit statistics, including the AIC, BIC, chi-squared, K-S, and A-D fit statistics.

For the estimation of tissue SSD TRVs herein, the selection of distributions focused on the AIC statistic, which corresponds to the fit of a theoretical distribution to the entire empirical distribution, as well as a visual inspection of several curve fits. In cases where the "best" AIC value did not correspond to a model with reasonable visual fit to the lower tail of the empirical data, the rankings of goodness-of-fit statistics (i.e., AIC, BIC, chi-squared, K-S, and A-D) for each distribution fit by @Risk were summed, resulting in a general indication of the best-fitting distribution(s). The top-ranked distributions (based on the sum of ranked statistics) were then compared visually. The TRV was calculated based on the distribution with the best visual fit among the top-ranked distributions. If multiple distributions had similarly good visual fits, then the TRV was calculated as the geometric mean of the 5th percentile estimates for all accepted models.

The distributions selected for each SSD are described in Section 7.1.3.2. Consistent with AWQC derivation methods (Stephan et al. 1985), the 5th percentile of the distribution was selected as the TRV. The 5th percentile concentration is assumed to protect 95% of the fish species present in the LPRSA.

Further discussion on the derivation of and uncertainties associated with the TRVs is presented in Section 7.1.3.2.

TRVs for Regulated Metals

Regulated metals in aquatic tissue were evaluated, consistent with USEPA (2015b, 2015c, 2016g) guidance. TRVs for regulated metals for evaluation in this BERA were developed as follows.

- For copper and lead, TRVs from the revised draft of the LPR restoration project FFS (Louis Berger et al. 2014) were used.
- For all other regulated metals COPECs (i.e., metals COPECs other than mercury and selenium), toxicological literature was reviewed and TRVs were developed based on the approach outlined above. When sufficient data were available (i.e., data for at least five species), TRVs were generated using an SSD approach. When data were insufficient, the lowest acceptable LOAEL and highest NOAEL below the LOAEL from the same study were selected as TRVs.

Individual toxicological studies compiled from the USACE ERED, extensive literature searches, and CPG's TRV database were reviewed. Once these studies had been evaluated, those that were deemed acceptable for the development of TRVs were compiled (Appendix E).

TRV Uncertainty

General uncertainties associated with selected fish tissue TRVs are the same as those associated with benthic invertebrate tissue TRVs, as discussed in Section 6.3.3.1, although the dataset for fish tissue TRVs is more robust than that of benthic invertebrate tissue TRVs. General uncertainties associated with the derivation of TRVs based on SSDs—including uncertainties regarding the use of ACRs to derive chronic data for use in an SSD and the selection of best-fit curves for SSD datasets—are also detailed in Section 6.3.3.1. Finally, there is high uncertainty associated with the evaluation of metals and PAHs using a tissue residue approach, as discussed in Section 6.3.3.1.

7.1.3.2 Selected TRVs for fish tissue

Fish tissue TRVs are presented in Table 7-6, and TRVs for regulated metals are presented in Table 7-7.

Table 7-6. Fish tissue TRVs

COPEC	Units (ww)	Range of TRVs[a]								
		TRV-A[b]				TRV-B[c]				
		NOAEL	LOAEL	Endpoint	Source	NOAEL	LOAEL	Endpoint	Source	Document
Metals										
Mercury/ methylmercury	µg/kg	35[d]	350	survival, growth, reproduction, and behavior (12 species)	SSD-derived 5th percentile	52	260	growth, survival, reproduction, and behavior (7 species)	Beckvar et al. (2005)	revised FFS (Louis Berger et al. 2014)
Selenium	mg/kg	na[e]	1.6	reproduction (bluegill, sunfish, and fathead minnow)	Coyle et al. (1993); Hermanutz et al. (1992); Ogle and Knight (1989)	no value[f]	no value[f]	na	na	na
PCBs										
Total PCBs	µg/kg	380[d]	3,800	survival, growth, and reproduction (11 species)	SSD-derived 5th percentile value	170	530	smolt seawater preference behavior (Atlantic salmon)	Lerner et al. (2007)	revised FFS (Louis Berger et al. 2014)
PCB TEQ - fish	ng/kg	12[d] (2.3[d,g])	120 (23[g])	survival, growth, and reproduction (7 species)	SSD-derived 5th percentile value	0.89	1.8	prey capture behavior (mummichog)	Couillard et al. (2011)	revised FFS (Louis Berger et al. 2014)
Organochlorine Pesticides										
Total DDx	µg/kg	52[d]	520	survival, growth, reproduction, and behavior (7 species)	SSD-derived 5th percentile value	78	390	growth, survival, reproduction, and behavior (9 species)	Beckvar et al. (2005)	revised FFS (Louis Berger et al. 2014)
Dieldrin	µg/kg	120	200	survival (rainbow trout)	Shubat and Curtis (1986)	8.0	40	survival (rainbow trout)	Shubat and Curtis (1986)	revised FFS (Louis Berger et al. 2014)

Table 7-6. Fish tissue TRVs

COPEC	Units (ww)	Range of TRVs[a]								
		TRV-A[b]				TRV-B[c]				
		NOAEL	LOAEL	Endpoint	Source	NOAEL	LOAEL	Endpoint	Source	Document
Endosulfan II	µg/kg	3.1[d]	31	survival (spot)	Schimmel et al. (1977)	no value[f]	no value[f]	na	na	na
PCDDs/PCDFs										
PCDD/PCDF TEQ - fish	ng/kg	12[d] (2.3[d,g])	120 (23[g])	survival, growth, and reproduction (7 species)	SSD-derived 5th percentile value	0.89	1.8	prey capture behavior (mummichog)	Couillard et al. (2011)	revised FFS (Louis Berger et al. 2014)
Total TEQ - fish	ng/kg									

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

b TRVs were derived from the primary literature review based on process identified in Section 7.1.3.1.

c TRVs were based on USEPA's revised draft of the LPR restoration project FFS (Louis Berger et al. 2014) or first draft of the LPR restoration project FFS (Malcolm Pirnie 2007b), as described in Section 7.1.3.1.

d NOAEL was extrapolated from LOAEL using an uncertainty factor of 10.

e No NOAEL was selected because LOAEL based on ED10 value for the most sensitive species evaluated, below which adverse effects are not expected.

f No TRVs were available in the revised FFS (Louis Berger et al. 2014) or draft FFS (Malcolm Pirnie 2007b).

g An alternate SSD distribution was also selected based on a conservative distribution fit; see Section 7.1.3.1 for details.

COPEC – chemical of potential ecological concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
ED10 – dose that corresponds to a 10% increase in an adverse effect of an exposed population
FFS – focused feasibility study
LOAEL – lowest-observed-adverse-effect level
LPR – Lower Passaic River
LPRSA – Lower Passaic River Study Area
na – not applicable
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
PCDD – polychlorinated dibenzo-*p*-dioxin
PCDF – polychlorinated dibenzofuran
SSD – species sensitivity distribution
TCDD – tetrachlorodibenzo-*p*-dioxin
TEQ – toxic equivalent
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT and 4,4′-DDT)
TRV – toxicity reference value
USEPA – US Environmental Protection Agency
ww – wet weight

Table 7-7. Fish tissue TRVs for regulated metals

COPEC	Units (ww)	Range of TRVs[a]								
		TRV-A[b]				TRV-B[c]				
		NOAEL	LOAEL	Endpoint	Source	NOAEL	LOAEL	Endpoint	Source	Document[a]
Metals										
Arsenic	mg/kg	1.3	2.5	growth (rainbow trout)	Erickson et al. (2011)	no value[d]	no value[d]	na	na	na
Cadmium	mg/kg	0.016[e]	0.16	survival, growth, reproduction (13 species)	SSD-derived 5th percentile value	no value[d]	no value[d]	na	na	na
Chromium	mg/kg	na[f]	na[f]	na	na	no value[d]	no value[d]	na	na	na
Copper	mg/kg	na[f]	na[f]	na	na	0.32	1.5	survival (striped mullet)	Zyadah and Abdel-Baky (2000)	revised FFS (Louis Berger et al. 2014)
Lead	mg/kg	2.5	4.0	growth (brook trout)	Holcombe et al. (1976)	0.4	4.0	reproduction (brook trout)	Holcombe et al. (1976)	revised FFS (Louis Berger et al. 2014)
Silver	mg/kg	0.11	0.24	growth (rainbow trout)	Guadagnolo et al. (2001)	no value[d]	no value[d]	na	na	na
Zinc	mg/kg	287	403	growth (guppy)	Pierson (1981)	no value[d]	no value[d]	na	na	na

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

b TRVs were derived from the primary literature review identified in Section 7.1.3.1.

c TRVs derived based on USEPA's revised draft of the LPR restoration project FFS (Louis Berger et al. 2014) or first draft of the LPR restoration project FFS (Malcolm Pirnie 2007b), as described in Section 7.1.3.1.

d No TRVs were available in the revised FFS (Louis Berger et al. 2014) or draft FFS (Malcolm Pirnie 2007b).

e NOAEL was extrapolated from LOAEL using an uncertainty factor of 10.

f No TRV selected; see Section 7.1.3.2 for further explanation.

COPEC – chemical of potential ecological concern
FFS – focused feasibility study
LOAEL – lowest-observed-adverse-effect level
na – not applicable
NOAEL – no-observed-adverse-effect level
SSD – species sensitivity distribution
TRV – toxicity reference value
USEPA – US Environmental protection agency
ww – wet weight

Mercury and Methylmercury

Fourteen studies examining behavior, growth, reproduction, and mortality in 12 species of fish were found to meet TRV acceptability criteria. These studies reported 17 LOAELs for methylmercury or mercury[100] in tissue ranging from 470 to 22,000 µg/kg ww. An ACR of 3.731 was applied to two acute LOAEL values to derive chronic LOAELs (Table 7-5) (USEPA 1985c). An SSD was developed using both chronic LOAELs and ACR-derived LOAELs (Figure 7-1). Eight of the studies used diet-based mercury exposure; the remaining seven studies were aqueous mercury exposures. Study duration ranged from 48 hrs to multi-generational, multi-year studies. The distribution of final species LOAELs was best described by a Levy distribution. The 5th percentile LOAEL TRV based on the SSD is 350 µg/kg ww (Figure 7-1). The NOAEL TRV (35 µg/kg ww) was extrapolated from the LOAEL TRV using an uncertainty factor of 10. There is uncertainty associated with the derivation of a NOAEL based on an extrapolation factor.

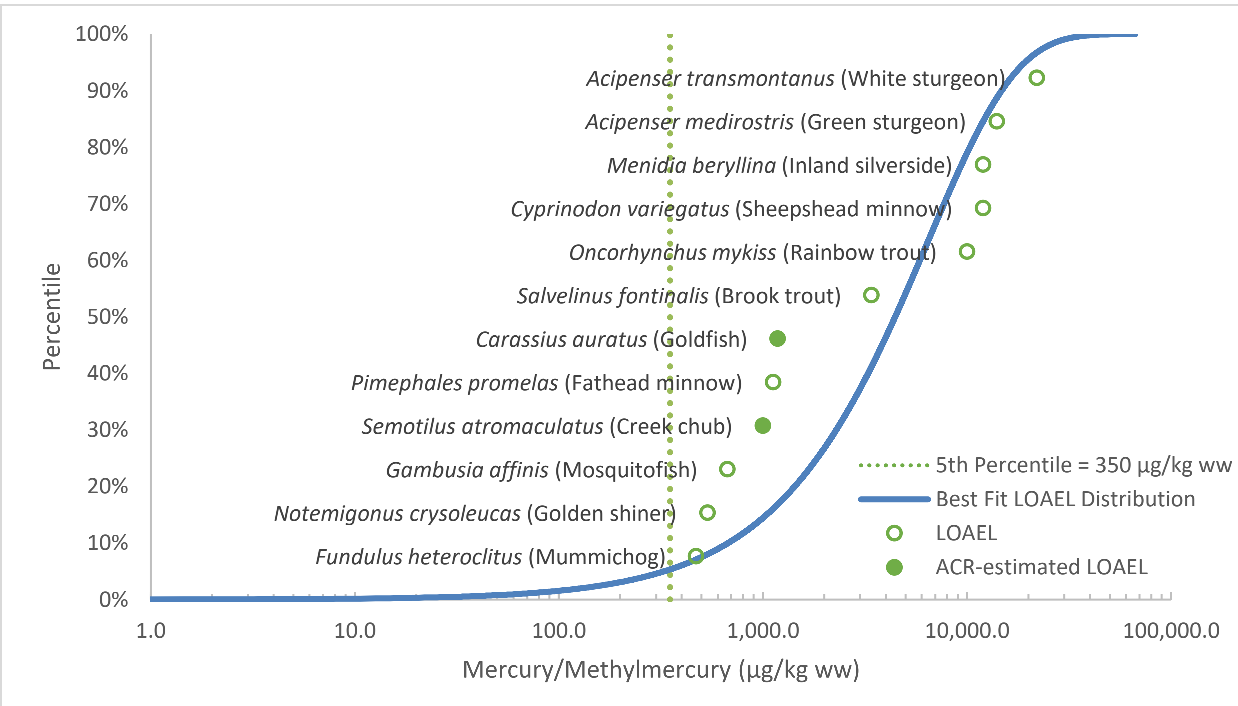


Figure 7-1. Fish chronic whole-body tissue methylmercury/mercury SSD toxicity data

The 5th percentile of the SSD (350 µg/kg ww) was less than the lowest acceptable LOAEL, so it provided a conservative estimate of the mercury tissue concentration at

[100] All of the studies measured total mercury in fish tissue, with the exception of one study that measured methylmercury. Total mercury is expected to closely represent methylmercury because > 95% of the mercury found in fish tissue is generally in the form of methylmercury (Bloom 1992; Grieb et al. 1990).

which 5% of LPRSA species might be adversely affected. The lowest LOAEL of 470 µg/kg ww was for increased mortality in male mummichog; the observed mortality was associated with altered aggressive behavior between males (Matta et al. 2001). Because the fish were confined to aquaria, it was uncertain how this behavioral change would affect fish in the wild. No other adverse effects were reported in this study at this exposure level. Effects of mercury on behavior leading to increased mortality were also reported for golden shiner and mosquitofish (*Gambusia affinis*). Webber and Haines (2003) reported that golden shiner with tissue burdens of 534 µg/kg ww mercury displayed altered predator avoidance behavior. Kania and O'Hara (1974) reported that when mosquitofish with tissue burdens of 670 µg/kg ww total mercury were released into aquaria with unexposed mosquitofish and largemouth bass, the mercury-exposed mosquitofish experienced higher predation. These studies indicated that fish might experience ecologically significant behavioral alterations at mercury tissue concentrations greater than the 5th percentile TRV. Based on a review of data from eight mercury-contaminated sites, Fuchsman et al. (2016) found no clear effects on fish populations attributable to mercury associated with whole-body tissue concentrations from 80 to 1,600 µg/kg ww. Fuchsman et al. (2016) did report observing adverse effects on fish populations at two sites with elevated mercury; however, these effects were not clearly related to mercury because multiple other contaminants were also present. The available data indicate that the selected mercury TRV is conservatively protective of the LPRSA fish population.

The LOAEL of 260 µg/kg ww (Louis Berger et al. 2014) was based on the 5th percentile LOAEL developed by Beckvar et al. (2005) using data derived from the USACE ERED. The NOAEL of 52 µg/kg ww (Louis Berger et al. 2014) was based on the use of an uncertainty factor of 5 and the SSD-derived LOAEL. As reported by Beckvar et al. (2005), eight LOAELs were selected for seven species of fish to derive a mercury 5th percentile LOAEL. LOAELs ranged from 250 to 5,000 µg/kg ww for survival, growth, reproduction, and behavior endpoints. There is some uncertainty associated with the LOAELs used in the SSD derived by Beckvar et al. (2005). Specifically, the second-lowest LOAEL of 300 µg/kg ww is based on striped mullet (*Mugil cephalus*) regeneration rates of amputated caudal fins (Weis and Weis 1978), and it is unclear how this effect would impact growth, survival, or reproduction in fish under conditions found in the LPRSA.

Selenium

Five studies examining growth, reproduction, and survival effects were found to meet TRV acceptability criteria. Six LOAELs were available for four species of fish (white sturgeon [*Acipenser transmontanus*], bluegill, Chinook salmon, and Dolly Varden [*Salvelinus malma*]), and data was therefore insufficient to develop an SSD. These studies reported LOAEL values that ranged from 2.1 to 9.8 mg/kg ww. Two of these studies used diet-based selenium exposures, two studies assumed maternal transfer of selenium, and the last study used both diet-based and aqueous selenium exposures.

Exposure duration ranged from 56 to 140 days. Hamilton et al. (1990) reported the lowest LOAEL of 2.1 mg/kg ww after exposing juvenile Chinook salmon to diet-based selenium (as seleno-DL-methionine) for 90 days. These studies were considered but not selected to derive TRVs. Despite insufficient LOAELs to derive an SSD, recent reviews of selenium toxicity to fish reported in peer-reviewed literature provide information to support the development of an appropriate TRV protective of LPRSA fish populations. DeForest and Adams (2011) conducted a comprehensive evaluation of the selenium toxicity literature available at the time their review was published. They indicated that "The classic pathway of documented Se poisoning in fish is exposure of adult female fish to Se, maternal transfer of the Se to the ovaries and then eggs, and then, if sufficiently high egg Se concentrations are reached, larval deformities and mortality." Because adverse effects are most closely associated with larval life stages, the use of egg tissue-based TRVs is recommended (DeForest and Adams 2011).

In a subsequent study, DeForest et al. (2012) conducted an SSD analysis of selenium toxicity to fish eggs and embryos. This study, based on various EC10s or NOAEL data for 12 species of fish, identified 20 mg/kg dw (5 mg/kg ww assuming 80% moisture content) as the 5th percentile of the distribution (i.e., protective of 95% of species). Based on reported ratios of egg or ovary selenium concentrations to adult whole-body selenium concentration ranging from 1.3 to 2.4, DeForest et al. (2012) estimated that whole-body selenium concentrations of 8.3 to 15.4 mg/kg dw (1.7 to 3.1 mg/kg ww assuming 80% moisture content) would result in egg selenium concentrations of 20 mg/kg dw. The lower end of this range (8.3 mg/kg dw) is similar to the 8.1 mg/kg dw (1.6 mg/kg ww assuming 80% moisture content) recommended by DeForest and Adams (2011) as a whole-body effects threshold protective of fish. This value (1.6 mg/kg ww) is the EC10 of maternal whole-body concentrations associated mortality or edema of larval bluegill sunfish and fathead minnow based on toxicity data reported in the literature (Coyle et al. 1993; Hermanutz et al. 1996; Ogle and Knight 1989). Given the lack of sufficient whole-body LOAEL data to derive an SSD, the threshold of 1.6 mg/kg ww recommended by DeForest and Adams (2011) was selected as the selenium LOAEL TRV. Because this value is representative of an EC10 for the most sensitive species evaluated, below which adverse effects are not expected, no additional (NOAEL) TRV was selected.

No TRVs were available in the draft FFS (Malcolm Pirnie 2007b) or revised FFS (Louis Berger et al. 2014) for selenium in fish tissue.

Total PCBs

Twelve studies examining growth and mortality effects of PCBs were found to meet TRV acceptability criteria. Fourteen LOAELs were identified for 11 fish species (goldfish [*Carassius auratus*], sheepshead minnow, channel catfish, pinfish, spot, coho salmon, rainbow trout, minnow [*Phoxinus phoxinus*], fathead minnow, guppy [*Poecillia reticulate*], and brook trout [*Salvelinus frontalis*]), ranging from 9,300 to

645,000 µg/kg ww using both aqueous and diet-based acute and chronic PCB exposures. Study duration ranged from 5 to 260 days. An ACR of 8.4 was applied to four acute LOAEL values to derive chronic LOAEL values USEPA (1980d) (Table 6-19); the range of LOAELs, including ACR-adjusted values, was 1,670 to 645,000 µg/kg. An SSD was developed using both chronic and ACR-derived LOAELs (Figure 7-2). The distribution of LOAELs was best described by a log-logistic distribution. The 5th percentile TRV based on the SSD was 3,800 µg/kg ww (Figure 7-2). The NOAEL TRV (380 µg/kg ww) was extrapolated from the LOAEL TRV using an uncertainty factor of 10. There is uncertainty associated with the derivation of a NOAEL based on an extrapolation factor.

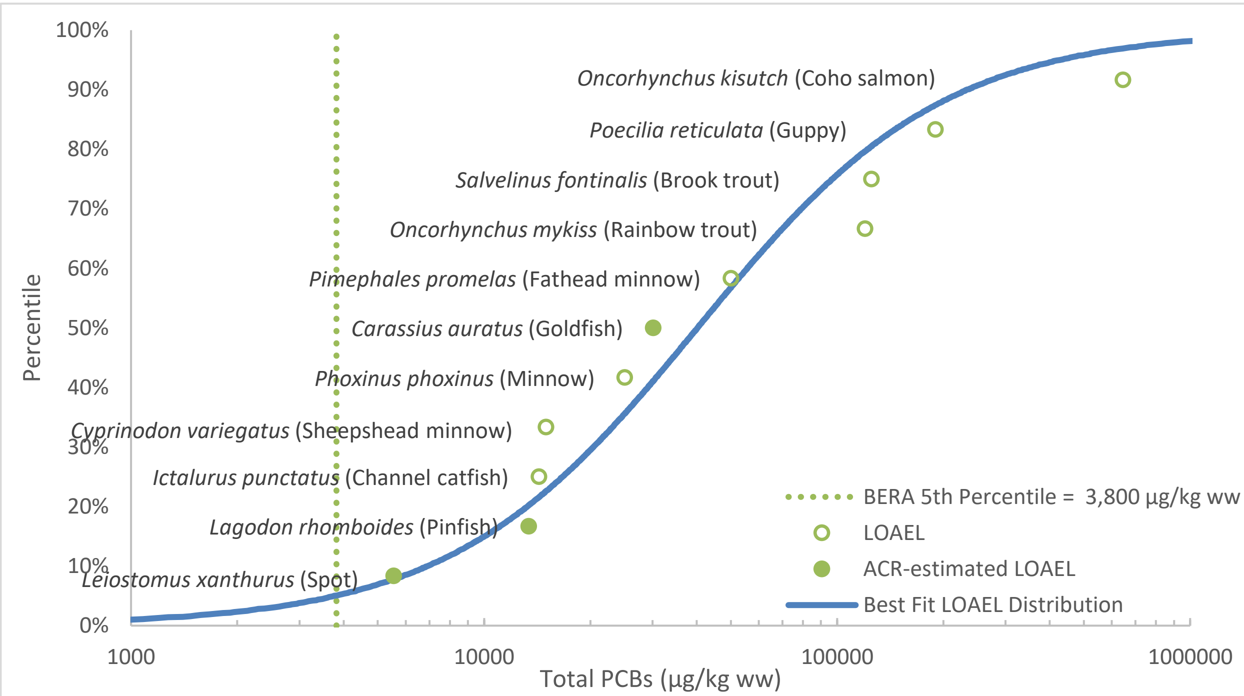


Figure 7-2.Fish chronic whole-body tissue total PCB SSD toxicity data

The SSD-derived LOAEL (3,800 µg/kg ww) is less than the lowest measured LOAEL reported from the literature: a tissue residue of 9,300 µg/kg ww associated with reduced reproduction in sheepshead minnow after 28 days of exposure to aqueous PCB Aroclor 1254 (Hansen et al. 1971) (Appendix E). The low end of the SSD curve is influenced by the two lowest values based on ACR-adjusted LOAELs (Figure 7-2). The removal of ACR-derived LOAELs from the SSD and recalculation of the 5th percentile SSD TRV (assuming the same theoretical distribution type) results in a TRV of 7,600 µg/kg ww.[101] Thus, the SSD-derived LOAEL represents a conservatively

[101] After removing the two lowest LOAEL values, the distribution remained reasonable based on the visual fit of the curve and several goodness-of-fit statistics calculated using @Risk software.

extrapolated value that is less than those empirically measured in the reviewed toxicity studies.

A NOAEL and LOAEL of 170 and 530 µg/kg ww, respectively, were also selected for total PCBs (Louis Berger et al. 2014) based on the behavioral endpoint of smolt seawater preference in Atlantic salmon (*Salmo salar*) during a three-week exposure to Aroclor 1254 (Lerner et al. 2007). The selected NOAEL and LOAEL were based on a decreased smolt seawater preference for Atlantic salmon exposed to 1 and 10 µg/L Aroclor 1254, respectively. There is uncertainty associated with these TRVs, because it is unclear how the effect on salmon smolt seawater preference is relevant to the potential for adverse effects on LPRSA fish populations. The use of these TRVs assumes that behavioral alterations that result in decreased foraging efficiency could correlate with growth effects, and that those alterations that affect predator avoidance and/or critical life stage-specific dispersal/migratory stages could result in reduced survival (Weis et al. 2011; Weis et al. 2001).

Total DDx

Six studies examining reproduction and survival effects of total DDx were found to meet TRV acceptability criteria. Seven LOAELs were identified for seven fish species (goldfish, sunfish *[Lepomis sp.*], cutthroat trout [*Oncorhynchus clarkii*], coho salmon, Chinook salmon, fathead minnow, and brook trout), ranging from 1,100 to 200,000 µg/kg ww using both aqueous and diet-based DDx exposures (Appendix E). Study duration ranged from 38 to 612 days. An SSD was developed using chronic LOAELs (Figure 7-3). The distribution of final species LOAELs was best described by a Weibull distribution. The 5th percentile LOAEL TRV based on the SSD is 520 µg/kg ww. This value was selected as the LOAEL TRV (Figure 7-3). The NOAEL TRV (52 µg/kg ww) was extrapolated from the LOAEL TRV using an uncertainty factor of 10. There is uncertainty associated with the derivation of a NOAEL based on an extrapolation factor. This SSD-derived LOAEL (520 µg/kg ww) is less than the lowest measured LOAEL reported from the literature: a tissue residue of 1,100 µg/kg ww associated with mortality in cutthroat trout after 111 days of exposure to aqueous DDx (Allison et al. 1964) (Appendix E). Thus, the SSD-derived LOAEL represents a conservatively extrapolated value that is less than those empirically measured in the reviewed toxicity studies.

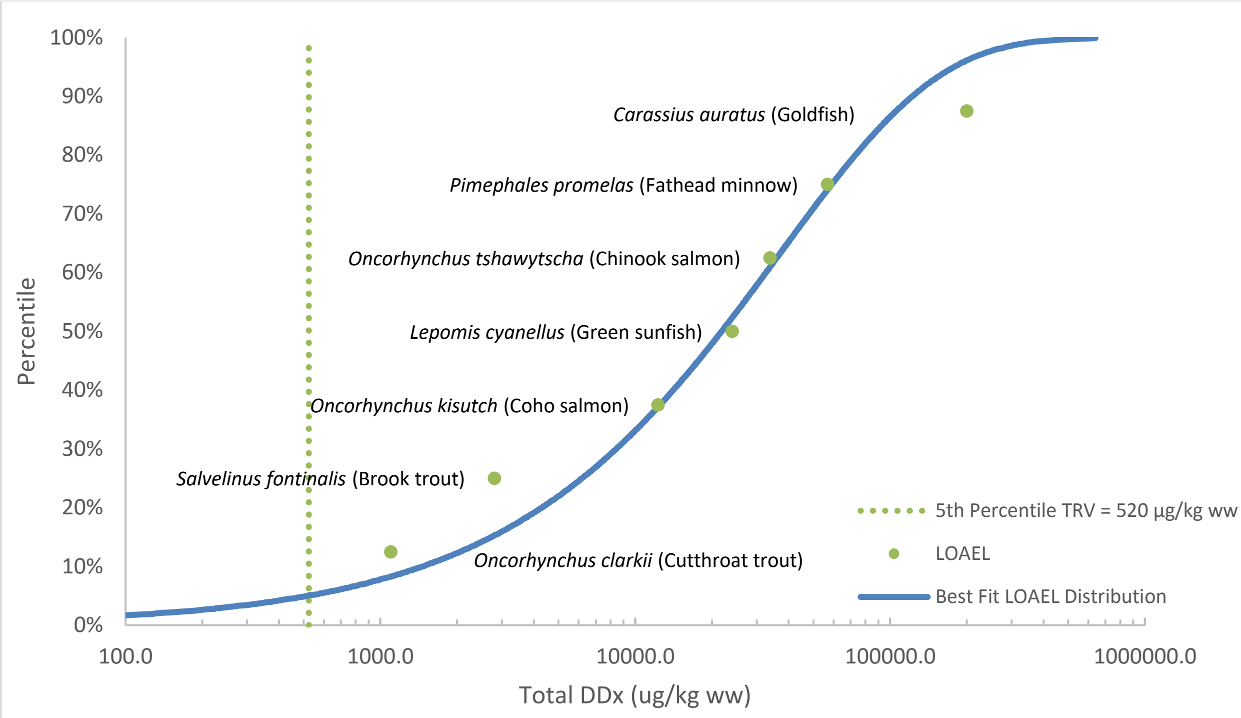


Figure 7-3.Fish chronic whole-body tissue total DDx SSD toxicity data

The total DDx LOAEL of 390 µg/kg ww (Louis Berger et al. 2014) was based on the 5th percentile LOAEL developed by Beckvar et al. (2005) using data derived from studies reported in USACE's ERED. The NOAEL of 78 µg/kg ww (Beckvar et al. 2005) was based on the use of an uncertainty factor of 5 and the SSD-derived LOAEL. As reported by Beckvar et al. (2005), 10 LOAELs were selected to derive a total DDx 5th percentile LOAEL for 9 species of fish. LOAELs ranged from 290 to 112,700 µg/kg ww for survival, growth, reproduction, and behavior endpoints. There is some uncertainty associated with the LOAELs used in the SSD derived by Beckvar et al. (2005). The lowest LOAEL of 290 µg/kg ww was based on data from Berlin et al. (1981), wherein survival of lake trout (*Salvelinus namaycush*) was affected based on data collected from fish hatched from field-collected eggs from Lake Michigan. These eggs had high concentrations of PCBs, DDx, and mercury; the elevated tissue burdens of PCBs and other contaminants may have contributed to toxicity. The next lowest LOAEL of 550 µg/kg ww was based on data from Butler (1969), wherein pinfish (*Lagodon rhomboides*) survival was affected; however, study data demonstrated that survival was not tissue concentration dependent. Finally, the third-lowest LOAEL of 1,650 µg/kg ww was based on goldfish behavior (locomotor activity) reported by Davy et al. (1972). It is unclear how locomotor activity is a direct measure of survival, growth, or reproduction.

TEQ - Fish

Nine studies examining reproduction, growth, and survival effects of dioxins and furans were found to meet TRV acceptability criteria. Eleven LOAELs were identified for seven fish species (whitefish [*Coregonus clupeaformis*], common carp, zebrafish, coho salmon, rainbow trout, Japanese medaka, and fathead minnow), ranging from 85 to 14,400 ng/kg ww using both aqueous and diet-based exposures. Study durations ranged from 6 hrs to 71 days. An ACR of 8.3 was applied to one acute LOAEL value to derive a chronic LOAEL value. An SSD was developed using chronic LOAELs and ACR-derived LOAELs, and two 5th percentile TRVs were developed by fitting several theoretical distributions to the SSD data using @Risk (Figure 7-4). Several models fit reasonably well, both visually and statistically, and the geometric mean 5th percentile TEQ-fish LOAEL of those models (i.e., the Pearson6, log-logistic, and Weibull distributions) was calculated as 120 ng/kg ww. The NOAEL TRV (12 µg/kg ww) was extrapolated from the LOAEL TRV using an uncertainty factor of 10. There is uncertainty associated with the derivation of a NOAEL based on an extrapolation factor. As an alternative, a 5th percentile LOAEL based on the beta general distribution (23 ng/kg ww)—which most accurately predicts the lowest LOAEL but has a relatively poor visual and statistical fit to the empirical data (when compared with the other distributions noted above)—was selected as a conservative SSD-derived estimate, consistent with 2017 communications between CPG and USEPA.

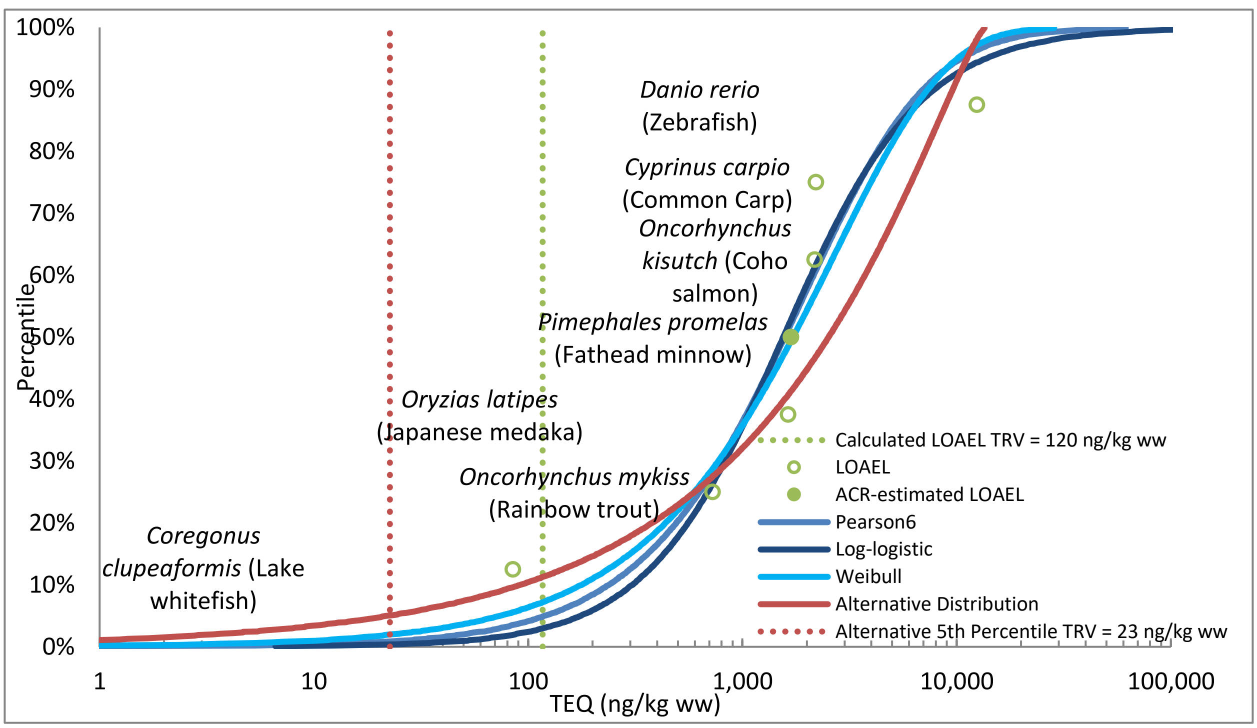


Figure 7-4.Fish chronic whole-body tissue 2,3,7,8-TCDD SSD toxicity data

The SSD-derived LOAEL of 120 ng/kg ww is within the range of measured LOAELs reported from the literature. However, the alternative SSD-derived LOAEL of

23 μg/kg ww is an order of magnitude less than the lowest measured LOAEL: a tissue residue of 85 ng/kg ww associated with reduced growth in lake whitefish after 30 days of exposure to dietary 2,3,7,8-TCDD (Fisk et al. 1997) (Appendix E). Thus, the alternative SSD-derived LOAEL of 23 ng/kg ww represents a conservatively extrapolated value that is much less than those empirically measured in the reviewed toxicity studies. The alternative NOAEL TRV (2.3 μg/kg ww) was extrapolated from the LOAEL TRV using an uncertainty factor of 10. There is uncertainty associated with the derivation of a NOAEL based on an extrapolation factor.

A NOAEL and LOAEL of 0.89 and 1.8 ng/kg ww, respectively, were also selected for TEQ - fish (Louis Berger et al. 2014) based on data presented in Couillard et al. (2011). Couillard et al. (2011) reported impacts on prey capture behavior in newly hatched mummichog following topical exposure of eggs to PCB 126 at doses of 50 kg/L, but not at doses of 25 kg/L. Larvae tissue concentrations were not reported by Couillard et al. (2011). Instead, they were estimated using the ratio of larval tissue concentration to topical dose (7.1) based on one empirical data point measured in a previous study (Couillard et al. 2008). PCDD/PCDF TEQs were derived by multiplying the estimated larval tissue concentrations (178 and 355 ng/kg ww) by the fish TEF for PCB 126 (0.005), resulting in the NOAEL and LOAEL of 0.89 and 1.8 ng/kg ww, respectively.

There is uncertainty associated with the selected TRVs based on several considerations. First, there is uncertainty in TEFs (see Section 4.3.2), which results in uncertainty in converting the effect level based on exposure to PCB 126 to an effect level based on exposure to the sum of toxic dioxin congeners and dioxin-like PCBs. In addition, concentrations in mummichog larvae were not measured in the same study that recorded impacts on behavior. Instead, the larval tissue concentrations were estimated from the topical dose using the ratio of larval tissue concentration to topical dose obtained from another study, and based on only one empirical data point. Finally, these TRVs are based on estimated tissue residues in larvae, but are being compared to adult tissue concentrations in the LPRSA. In comparison to other TRVs, the larval tissue TEQs of 1.8 and 0.89 ng/kg ww are an order of magnitude less than the egg tissue TEQ TRVs (established in Section 7.4.3) of 86 and 7.2 ng/kg ww.

Dieldrin

Two studies examining mortality following dieldrin exposure were found to meet TRV acceptability criteria. LOAELs were available for two species (rainbow trout and sheepshead minnow) and were therefore insufficient to develop an SSD. At the lowest LOAEL, Shubat and Curtis (1986) reported reduced growth of rainbow trout exposed for 16 weeks to aqueous dieldrin with an associated average tissue burden of 200 μg/kg ww. No effect on growth was detected in fish with higher tissue burdens when subjected to dietary or combined dietary and aqueous exposures. No adverse effects on growth were observed in fish at the next lower aqueous exposure level associated with a tissue burden of 120 μg/kg ww. These thresholds of 200 and 120 μg/kg ww were selected as the LOAEL and NOAEL values for dieldrin, respectively.

There is uncertainty associated with these TRVs, as there are very limited toxicity data available for dieldrin (two studies). It was also assumed that these values were reported as wet weight concentrations.

A NOAEL and LOAEL of 8.0 and 40 µg/kg ww, respectively, were also selected for dieldrin (Louis Berger et al. 2014) based on Shubat and Curtis (1986), but it appears that residues were assumed to be reported as dry weight and were therefore converted to wet weight. In addition, an uncertainty factor of two was applied to the TRVs (Louis Berger et al. 2014); there is uncertainty associated with the use of extrapolation factors.

Endosulfan

One study examining effects on survival was found to meet TRV acceptability criteria. Three LOAELs were available for three fish species (spot, pinfish, and mullet), so data were insufficient to develop an SSD. This study reported LOAEL values that ranged from 31 to 360 µg/kg ww (Appendix E). Schimmel et al. (1977) reported the lowest LOAEL of 31 µg/kg ww associated with a 15% increase in mortality relative to controls in spot croaker exposed to aqueous endosulfan for 96 hrs. A NOAEL was not identified within this study, so a NOAEL TRV of 3.1 µg/kg ww was estimated as the LOAEL divided by 10. There is uncertainty associated with the derivation of a NOAEL based on an extrapolation factor. There is also uncertainty associated with these TRVs, as there are very limited toxicity data available for dieldrin (one study). Because the magnitude of effect associated with the LOAEL was low, the effects on fish populations are uncertain, adding additional uncertainty to the extrapolated NOAEL.

No TRVs were available from the draft FFS (Malcolm Pirnie 2007b) or revised FFS (Louis Berger et al. 2014) for endosulfan in fish tissue.

Regulated metals

Arsenic, cadmium, chromium, copper, lead, silver, and zinc tissue residue effects data were evaluated to derive a TRV.

Arsenic

Three studies examining growth, behavior, and mortality were found to meet TRV acceptability criteria. Three LOAEL values were available for one fish species (rainbow trout and were therefore insufficient to develop an SSD. These studies reported LOAEL values that ranged from 2.5 to 8.1 mg/kg ww. Erickson et al. (2011) exposed juvenile rainbow trout to aqueous arsenic for 28 days and reported the lowest LOAEL value, which was selected for TRV derivation. A LOAEL of 2.5 mg/kg ww (assuming 80% moisture) associated with a ≥ 25% reduction in growth and a NOAEL of 1.3 mg/kg ww (assuming 80% moisture) were identified and selected as TRVs. McGeachy and Dixon (1990) found a similarly significant reduction in rainbow trout body weights (24 and 33%) associated with tissue concentrations of 2.5 and 3.5 mg/kg ww, respectively. However, growth measured as a condition index was not significantly affected, and McGeachy and Dixon (1990) stated that arsenic did not appreciably affect growth. Erickson et al. (2011) reported < 10% effects on growth of rainbow trout at concentrations < 1.3 mg/kg ww. There is uncertainty associated with the selected TRVs due to the paucity of toxicity data available for arsenic.

Both studies indicated a high degree of variability in arsenic tissue concentrations associated with mortality. McGeachy and Dixon (1992) reported that critical arsenic body burdens in rainbow trout ranged from 4 to 12 mg/kg ww, depending on temperature and exposure duration. Erickson et al. (2011) reported that rainbow trout critical arsenic body burdens ranged up to 15 mg/kg dw (3.0 mg/kg ww, assuming 80% moisture) in surviving fish, and that mortality did not correlate well with total arsenic accumulation. For these reasons, the selected TRVs are very conservative, and subsequent HQs may overestimate risk to fish within the LPRSA.

No TRVs for arsenic were available in the revised FFS (Louis Berger et al. 2014) or draft FFS ***(Malcolm Pirnie 2007b)***.

Cadmium

Eighteen studies examining effects on growth, reproduction, and survival were found to meet TRV acceptability criteria. Eighteen LOAELs were identified for 13 fish species (zebra fish [*D. rerio*], mummichog, three-spined stickle back [*Gasterosteus aculeatus*], gudgeon [*Gobio gobio*], American flagfish [*Jordanella floridae*], seabass [*Lates calcarifer*], spot, bluegill, striped bass, stone loach [*Nomacheilus barbatulus*], rainbow trout, Atlantic salmon, and brook trout [*Salvelinus fontinalis*]), ranging from 0.12 to 144 mg/kg ww (Figure 7-5). An ACR of 9.1 was applied to five acute LOAEL values to derive chronic LOAEL TRVs (USEPA 2001). Seventeen of these studies used water-based cadmium exposures while the remaining study was a diet-based exposures. Study duration ranged from 5 days to 3.5 years. An SSD was derived from this data and the distribution of final species LOAELs was best described by an inverse Gaussian distribution. The 5th percentile of the SSD (0.16 mg/kg ww) was selected as the LOAEL TRV (Figure 7-5). This SSD-derived LOAEL (0.16 mg/kg ww) is within the range of measured LOAELs reported from the literature (Appendix E). The NOAEL TRV (0.016 µg/kg ww) was extrapolated from the LOAEL TRV using an uncertainty factor of 10. There is uncertainty associated with the derivation of a NOAEL based on an extrapolation factor.

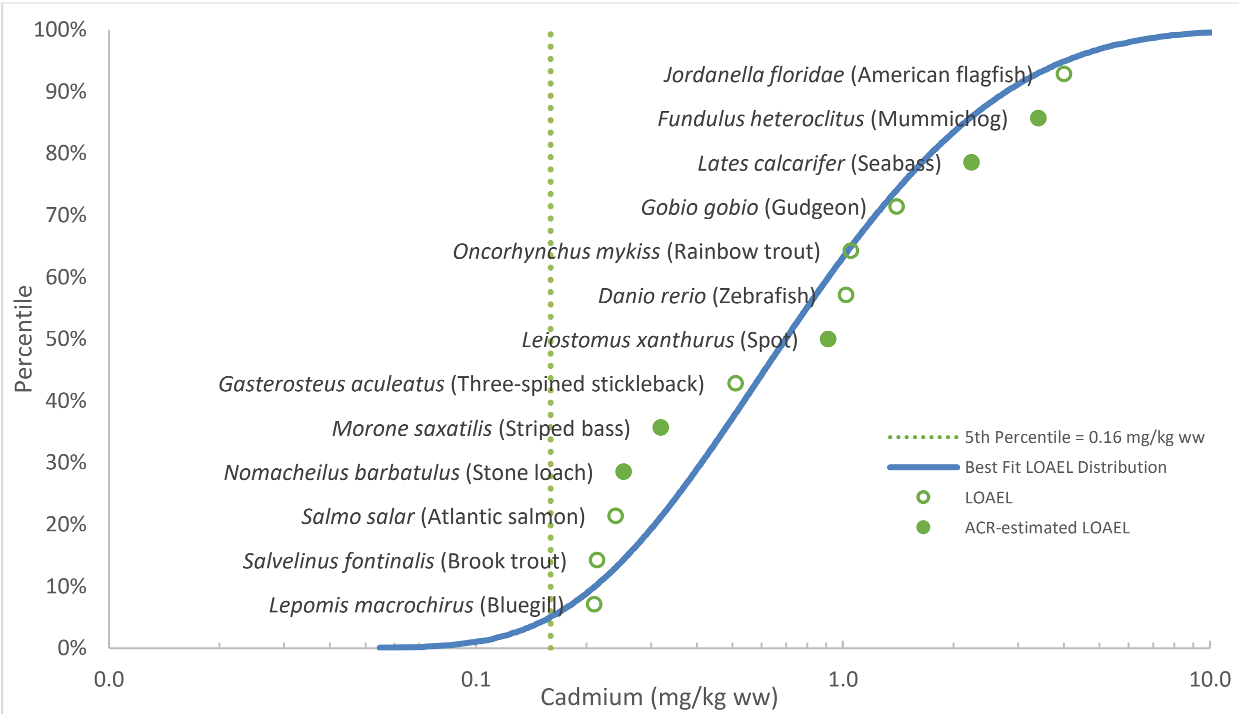


Figure 7-5.Fish chronic whole-body tissue cadmium SSD toxicity data

Cadmium can be sequestered in detoxified forms away from the site of action, contributing to uncertainty about the relationship of whole-body concentrations to adverse toxic effects (Amiard et al. 1987). Exposure conditions such as metal bioavailability, exposure route, and exposure time contribute to the regulation and bioaccumulation of cadmium, as well as the fraction of metabolically active cadmium causing toxicity. There is, therefore, a high degree of uncertainty in tissue residue effects thresholds (Adams et al. 2011).

No TRVs for cadmium were available in the revised FFS (Louis Berger et al. 2014) or draft FFS (Malcolm Pirnie 2007b).

Chromium

No appropriate LOAELs were identified for the derivation of a chromium TRV. Three studies that examined growth and survival endpoints (Attachment E) for three fish species (Chinook salmon, rainbow trout, and mummichog) were identified. These studies were reviewed but were eliminated from consideration for TRV derivation for the reasons described in the following paragraphs.

Farag et al. (2006) reported the lowest LOAEL of 1.30 mg/kg ww for the survival of juvenile Chinook salmon. However, this study was excluded from evaluation due to high uncertainty associated with the study design. An inconsistent dosing regimen was employed, wherein test organisms were exposed to 54 µg/L of aqueous chromium for 105 days with no significant effect on survival or growth metrics (weight and length); this exposure was associated with a tissue concentration of 1.80

mg/kg ww. This exposure was followed by an additional 29 days of exposure to 266 µg/L, during which an 18% reduction in the survival of juvenile Chinook salmon, associated with a tissue concentration of 1.30 mg/kg ww, was observed. It appears that the large increase in waterborne chromium for the last 29 days of the test was probably responsible for the reduced survival observed, making the effect independent from the concentrations measured in the tissue.

Two other studies were identified: Roling et al. (2006) reported a LOAEL of 44.1 mg/kg ww for the growth of larval mummichog over an exposure duration of 30 days, and Van der Putte et al. (1981) reported a LOAEL of 8.7 mg/kg ww for the survival of rainbow trout over an exposure duration of 4 days. Whereas the authors specified the use of hexavalent chromium in their toxicology studies, only total chromium was measured in LPRSA fish tissue, and no site-specific evaluation of the ratio between hexavalent chromium and total chromium was conducted. Therefore, there is significant uncertainty involved in the comparison of hexavalent chromium TRVs to total chromium fish tissue exposure point concentrations (EPCs).

No TRVs for chromium were available in the revised FFS (Louis Berger et al. 2014) or draft FFS (Malcolm Pirnie 2007b).

Copper

Studies have shown that fish acclimate to elevated aqueous copper levels, sequestering excess copper in metallothionein proteins within organs such as the liver, muscle, or gill, and then mobilizing copper when concentrations are low, thus maintaining stable, metabolically available copper concentrations (Marr et al. 1996). Based on a survey of national tissue datasets, Meador (2015) reported that background levels of copper in fish tissue were generally 0.99 mg/kg ww or less, and that salmonids generally had lower copper tissue concentrations than other fish. Studies of copper nutritional sufficiency have shown that optimal growth in some species occurs at substantially higher tissue burdens. Tan et al. (2011) reported an optimal dietary copper concentration of 3.13 to 4.24 mg/kg ww for yellow catfish, with an associated whole-body tissue concentration of approximately 4 mg/kg ww. Similarly, Lin et al. (2008) reported that in grouper, a whole-body copper concentration of 3.4 mg/kg ww was associated with nutritionally optimal dietary copper concentrations; this value was selected as the optimal nutritional threshold.

No appropriate LOAELs were identified for the derivation of a copper TRV. Five studies examining growth and mortality endpoints for five fish species (Nile tilapia [*Oreochromis niloticus*], rainbow trout, grey mullet [*Mugil cephalus*], grouper [*Epinephelinae* sp.], and mummichog) were identified. Lethal copper tissue burdens in fish have been shown to be a function of duration and route of exposure, among other factors (Adams et al. 2011; Zyadah and Abdel-Baky 2000). These studies, which reported tissue body burdens greater than the nutritionally optimal copper body burden of 3.4 mg/kg ww, were reviewed but rejected. A summary of CPG's reasoning for the exclusion of these studies is provided in the following paragraphs.

Ali et al. (2003) reported reduced growth in Nile tilapia exposed to aqueous copper resulting in a whole-body tissue concentration of 3.7 mg/kg ww for seven weeks. Baker et al. (1998) reported a LOAEL tissue burden of 4.6 mg/kg ww for reduced growth in grey mullet over an exposure duration of 10 weeks. Decreased food uptake was observed in both of these studies, which were excluded from TRV development because the observed effects could have been confounded with decreased food uptake, independent of copper tissue burden concentrations. CPG does not recommend the use of the following studies for TRV derivation:

- Mount et al. (1994) reported a LOAEL tissue burden of 4.5 mg/kg ww for the mortality of rainbow trout over an exposure duration of 60 days. Mortality and whole-body copper concentrations were measured at two points during the study, day 35 and day 60. Although no significant mortality was observed at day 35, whole-body copper concentrations were 45 to 55% higher at day 35 than at day 60, when significant mortality was observed in the two highest treatments. Therefore, mortality was likely due to elevated copper concentrations in the water, rather than elevated whole-body copper concentrations. CPG does not recommend the use of this study for TRV derivation.
- Lin et al. (2008) reported a LOAEL tissue burden of 6.1 mg/kg ww for the increased growth of grouper over an eight-week exposure period. A 16% increase in weight was associated with this tissue burden, which was the result of a 4.37 mg/kg dietary copper treatment. However, treatments greater than this level resulted in no significant increases in weight compared to controls, indicating that the increased weight was not the result of whole-body copper concentrations. CPG does not recommend the use of this study for TRV derivation.
- Eisler and Gardner (1973) reported a LOAEL tissue burden of 13 mg/kg ww for the mortality of mummichog over a 96-hr exposure period. Both copper treatments (1 and 8 mg/L) resulted in a significant increase in cumulative mortality compared to the control. However, tissue residue for the 1-mg/L treatment (13 mg/kg ww) was less than for the control (19 mg/kg ww), while the 8-mg/L treatment had a significantly higher whole-body tissue residue (26 mg/kg ww). CPG does not recommend the use of this study for TRV derivation because mortality was not dose-responsive in regards to tissue burden.

A NOAEL and LOAEL of 0.32 and 1.5 mg/kg ww, respectively, were selected for copper (Louis Berger et al. 2014) based on a mortality response in striped mullet from a series of acute (up to 168-hr) copper toxicity tests (Zyadah and Abdel-Baky 2000). These values were less than the nutritionally optimal levels for fish, and an increase in tissue concentrations in fish did not correlate with an increased adverse effect (Zyadah and Abdel-Baky 2000). The LOAEL of 1.5 mg/kg ww was derived from the reported tissue residue of 7.5 mg/kg ww at 24 hrs of exposure associated the 10 mg/L aqueous copper exposure level, at which point the LC50 was 6.3 mg/L. After 168 hrs, the LC50 was reduced to 1.8 mg/L. A subchronic-to-chronic uncertainty factor of 5 was used to determine the LOAEL of 1.5 mg/kg ww from the 24-hr LC50 value of 7.5 mg/kg ww. However, these LC50 values were not consistent with the other concentrations observed at other exposures. Tissue copper concentrations in control and in 0.5-, 2-, and 5-mg/L copper treatments were not dose responsive, but rather ranged up to 3.9 mg/L at the 0.5-mg/L treatment level. The NOAEL TRV of 0.32 mg/kg ww was derived from the 24-hr tissue concentration of 1.6 mg/kg ww, for which 30% mortality was observed in striped mullet exposed to 5 mg/L aqueous copper. A subchronic-to-chronic uncertainty factor of 5 was used to determine the NOAEL of 0.32 mg/kg ww from the 24-hr LC50 value of 1.6 mg/kg ww. There is uncertainty in the use of TRVs that appear to be less than nutritionally optimal threshold concentrations. There is also uncertainty associated with the use of an extrapolation factor to derive the TRV.

Lead

Two studies examining effects on growth and behavior were found to meet TRV acceptability criteria. LOAELs were available for two species of fish (fathead minnow and brook trout), ranging from 4.02 to 26.2 mg/kg ww. Holcombe et al. (1976) exposed juvenile brook trout to water-based lead concentrations for 3 generations over 3 years, resulting in the lowest reported LOAEL; this study was selected for TRV derivation. A LOAEL of 4.02 mg/kg ww (assuming 80% moisture) was identified and associated with decreased egg hatchability of the third generation of fish. No effects on survival, growth, or reproduction were observed at this exposure level in the preceding two generations; however, there was a clear dose-response relationship for increased scoliosis that affected spawning behavior. A NOAEL of 2.5 mg/kg ww (assuming 80% moisture) was also identified from the same study; these values were selected as TRVs. The paucity of data increased the uncertainty of the selected TRVs.

A NOAEL and LOAEL of 0.4 and 4.0 mg/kg ww, respectively, were also selected for lead (Louis Berger et al. 2014) based on data from Holcombe et al. (1976). The endpoint noted in the revised FFS (Louis Berger et al. 2014) is reproductive (i.e., deformed spines in third-generation fish). A NOAEL of 0.4 mg/kg ww (Louis Berger et al. 2014) was extrapolated from the LOAEL using an uncertainty factor of 10. There is uncertainty associated with the use of an extrapolation factor to derive the NOAEL.

Silver

One study examining effects on survival was found to meet TRV acceptability criteria. One LOAEL was available for one fish species (rainbow trout) and data were therefore insufficient to develop an SSD. Guadagnolo et al. (2001) exposed embryonic rainbow trout to aqueous silver for 32 days, identifying a LOAEL of 0.24 mg/kg ww (associated with increased mortality) and a NOAEL of 0.11 mg/kg ww; these values were selected as TRVs. Adverse effects observed in this study were not tissue concentration dependent, as tissue burdens did not consistently correlate with mortality. While this LOAEL was selected, it is a highly uncertain value given that the adverse effects observed were not concentration dependent. Furthermore, there is uncertainty associated with these selected TRVs due to the paucity of toxicity data available for silver.

No TRVs for silver were available in the revised FFS (Louis Berger et al. 2014) or draft FFS (Malcolm Pirnie 2007b).

Zinc

Zinc is an essential nutrient for fish and is actively regulated within tissue via a variety of mechanisms. Therefore, it is important to consider these two factors when determining potential effects for the evaluation of risks to fish. Fish are able to regulate their tissue burdens of zinc. For example, as the dietary zinc load increases, the proportion of zinc absorbed from the diet decreases. Additionally, alteration in zinc uptake across the gills prevents increased uptake during increased aqueous zinc exposures (Bury et al. 2003). Sun and Jeng (1998) reported that the tissues of aquatic organisms typically contained 10 to 100 mg/kg ww zinc, with little variation among freshwater and brackish water fish, marine fish, and invertebrates. A national survey conducted by USFWS collected whole-fish samples from more than 100 sites and found the mean zinc body burden to be 21.7 mg/kg ww (Schmitt and Brumbaugh 1990).

Pierson (1981) reported that reproduction in guppy was highest at an aqueous zinc concentration of 0.173 mg/L, which was associated with a tissue burden of 112 mg/kg ww; this was selected as the zinc nutritional threshold for fish. A study by Sun and Jeng (1998) and a national survey conducted by USFWS (Schmitt and Brumbaugh 1990) confirmed this value.

Four studies examining effects on growth and survival were found to meet TRV acceptability criteria. Four LOAELs were available for three fish species (American flagfish, guppy, and mummichog) and data was therefore insufficient to develop an SSD. The lowest LOAEL greater than the zinc nutritional threshold was selected for TRV derivation. Pierson (1981) exposed immature guppy to aqueous zinc for 134 days. A LOAEL of 403 mg/kg ww was associated with reduced growth and reproduction, and a NOAEL of 287 mg/kg ww was identified from the same study; these values were selected as TRVs.

No TRVs for zinc were available in the revised FFS (Louis Berger et al. 2014) or draft FFS (Malcolm Pirnie 2007b).

7.1.4 Risk characterization

This section presents the comparison of fish tissue EPCs to TRVs to calculate HQs and EFs for all COPECs. Uncertainties associated with the risk estimates are discussed.

7.1.4.1 Tissue HQs

Fish tissue LOAEL and NOAEL HQs are presented in Tables 7-8 and 7-9, respectively, for all fish species. Appendix G lists EPCs, TRVs, and calculated HQs for the fish tissue COPECs in a single table (Table G3). LOAEL HQs were ≥ 1.0 for at least one fish species for methylmercury, total PCBs, PCB TEQ - fish, 2,3,7,8-TCDD, PCDD/PCDF TEQ - fish, total TEQ - fish, dieldrin, and total DDx.

Table 7-8. Fish tissue LOAEL HQs

COPEC	Range of LOAEL HQs[a]																							
	HQ based on TRV-A[b]												HQs based on TRV-B[c]											
	Benthic Omnivores			Invertivore					Piscivore				Benthic Omnivores			Invertivore					Piscivore			
	Mummichog	Other Forage Fish	Common Carp	White Perch	Channel Catfish	Brown Bullhead	White Sucker	White Catfish	American Eel	Largemouth Bass	Smallmouth Bass	Northern Pike	Mummichog	Other Forage Fish	Common Carp	White Perch	Channel Catfish	Brown Bullhead	White Sucker	White Catfish	American Eel	Largemouth Bass	Smallmouth Bass	Northern Pike
Metals																								
Mercury	0.18	0.24	0.23	0.57	0.43	0.31	0.40[d]	0.80	0.74	**1.9[d]**	0.86[d]	0.63[d]	0.24	0.32	0.31	0.77	0.58	0.42	0.54[d]	**1.1**	**1.0**	**2.6[d]**	**1.2[d]**	0.85[d]
Methylmercury	0.15	0.20	0.18	0.49	0.40	0.26	0.37[d]	0.71	0.80	**1.5[d]**	0.63[d]	0.51[d]	0.20	0.27	0.24	0.65	0.54	0.35	0.50[d]	0.96	**1.1**	**2.0[d]**	0.85[d]	0.69[d]
Selenium	0.45	0.44	0.51	0.88	0.19	0.48	0.29[d]	0.24	0.48	0.37[d]	0.43[d]	0.34[d]	nc	nc	nc	nc	nc	nc	Nc	nc	nc	nc	nc	nc
PCBs																								
Total PCBs	0.16	0.14	**1.4**	0.66	0.45	0.37	0.76[d]	0.89	0.53	**2.1[d]**	0.37[d]	0.53[d]	**1.1**	**1.0**	**9.8**	**4.7**	**3.2**	**2.6**	**5.5[d]**	**6.4**	**3.8**	**15[d]**	**2.6[d]**	**3.8[d]**
PCB TEQ - fish	0.0052 (0.027[e])	0.0052 (0.027[e])	0.037 (0.19[e])	0.018 (0.091[e])	0.015 (0.078[e])	0.011 (0.057[e])	0.027[d] (0.14[d,e])	0.029 (0.15[e])	0.010 (0.052[e])	0.14[d] (0.74[d,e])	0.012[d] (0.061[d,e])	0.019[d] (0.010[d,e])	0.34	0.34	**2.4**	**1.2**	**1.0**	0.72	**1.8[d]**	**1.9**	0.67	**9.4[d]**	0.78[d]	**1.3[d]**
PCDDs/PCDFs																								
2,3,7,8-TCDD	0.41 (**2.1[e]**)	0.38 (**2.0[e]**)	**5.1 (27[e])**	**1.6 (8.3[e])**	0.80 (**4.2[e]**)	**1.3 (6.5[e])**	**1.1[d] (5.7[d,e])**	**1.8 (9.1[e])**	0.19 (**1.0[e]**)	**1.5[d] (7.8[d,e])**	0.63[d] (**3.3[d,e]**)	0.79[d] (**4.1[d,e]**)	**27**	**26**	**340**	**110**	**53**	**83**	**72[d]**	**120**	**13**	**100[d]**	**42[d]**	**53[d]**
PCDD/PCDF TEQ - fish	0.43 (**2.2[e]**)	0.41 (**2.1[e]**)	**5.2 (27[e])**	**1.7 (8.7[e])**	0.83 (**4.3[e]**)	**1.3 (7.0[e])**	**1.1[d] (5.7[d,e])**	**1.8 (9.6[e])**	0.20 (**1.0[e]**)	**1.5[d] (7.8[d,e])**	0.63[d] (**3.3[d,e]**)	0.83[d] (**4.3[d,e]**)	**28**	**27**	**340**	**110**	**56**	**89**	**72[d]**	**120**	**13**	**100[d]**	**42[d]**	**56[d]**
Total TEQ - fish	0.43 (**2.2[e]**)	0.41 (**2.1[e]**)	**5.2 (27[e])**	**1.7 (8.7[e])**	0.83 (**4.3[e]**)	**1.3 (7.0[e])**	**1.1 (5.7[d,e])**	**1.9 (10[e])**	0.21 (**1.1[e]**)	**1.5[d] (7.8[d,e])**	0.68[d] (**3.6[d,e]**)	0.92[d] (**4.8[d,e]**)	**28**	**27**	**340**	**110**	**56**	**89**	**72[d]**	**130**	**14**	**100[d]**	**46[d]**	**61[d]**
Organochlorine Pesticides																								
Dieldrin	0.055	0.080	0.28	0.16	0.24	0.15	0.13[d]	0.14	0.27	0.20[d]	0.10[d]	0.22[d]	0.28	0.40	**1.4**	0.78	**1.2**	0.75	0.63[d]	0.68	**1.4**	**1.0[d]**	0.50[d]	**1.1[d]**
Endosulfan I	0.026	0.048	0.13	0.0071	0.065	0.035	0.015[d]	0.055	0.019	0.32[d]	0.094[d]	0.068[d]	nc	nc	nc	nc	nc	nc	nc	nc	nc	nc	nc	nc
Total DDx	0.13	0.14	1.3	0.46	0.54	0.31	0.29[d]	0.67	0.50	0.31[d]	0.44[d]	0.54[d]	0.17	0.19	**1.7**	0.62	0.72	0.41	0.38[d]	0.90	0.67	0.41[d]	0.59[d]	0.72[d]

Bold identifies HQs ≥ 1.0.

Shaded cells identify HQs ≥ 1.0 based on a LOAEL TRV.

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

[b] TRVs were derived from the primary literature review.

[c] TRVs were based on USEPA's revised draft of the LPR restoration project FFS (Louis Berger et al. 2014) or first draft of the LPR restoration project FFS (Malcolm Pirnie 2007b).

[d] Fewer than six detected concentrations were available, so HQ was based on a maximum concentration rather than a UCL.

[e] HQs in parenthesis were based on additional alternative SSD-derived LOAELs evaluated (see text in Section 7.1.3 for details).

COPEC – chemical of potential ecological concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
FFS – focused feasibility study
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
LPR – Lower Passaic River
nc – not calculated
PCB – polychlorinated biphenyl
PCDD– polychlorinated dibenzo-*p*-dioxin
PCDF –polychlorinated dibenzofuran
SSD – species sensitivity distribution
TCDD – tetrachlorodibenzo-*p*-dioxin
TEQ – toxic equivalents
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT and 4,4′-DDT)
TRV – toxicity reference value
UCL – upper confidence limit on the mean
USEPA – US Environmental Protection Agency

Table 7-9. Fish tissue NOAEL HQs

COPEC	Range of NOAEL HQs[a]																							
	HQs based on TRV-A[b]												HQs based on TRV-B[c]											
	Benthic Omnivores			Invertivore					Piscivore				Benthic Omnivores			Invertivore					Piscivore			
	Mummichog	Other Forage Fish	Common Carp	White Perch	Channel Catfish	Brown Bullhead	White Sucker	White Catfish	American Eel	Largemouth Bass	Smallmouth Bass	Northern Pike	Mummichog	Other Forage Fish	Common Carp	White Perch	Channel Catfish	Brown Bullhead	White Sucker	White Catfish	American Eel	Largemouth Bass	Smallmouth Bass	Northern Pike
Metals																								
Mercury	**1.8**	**2.4**	**2.3**	**5.7**	**4.3**	**3.1**	**4.0**[d]	**8.0**	**7.4**	**19**[d]	**8.6**[d]	**6.3**[d]	**1.2**	**1.6**	**1.5**	**3.8**	**2.9**	**2.1**	**2.7**[d]	**5.4**	**5.0**	**13**[d]	**5.8**[d]	**4.2**[d]
Methylmercury	**1.5**	**2.0**	**1.8**	**4.9**	**4.0**	**2.6**	**3.7**[d]	**7.1**	**8.0**	**15**[d]	**6.3**[d]	**5.1**[d]	**1.0**	**1.3**	**1.2**	**3.3**	**2.7**	**1.8**	**2.5**[d]	**4.8**	**5.4**	**10**[d]	**4.2**[d]	**3.5**[d]
Selenium	nc[e]	nc[e]	nc[e]	nc[e]	nc[e]	nc[e]	nc[e]	nc[e]	nc[e]	nc[e]	nc[e]	nc[e]	nc	nc	nc	nc	nc	nc	nc	nc	nc	nc	nc	nc
PCBs																								
Total PCBs	**1.6**	**1.4**	**14**	**6.6**	**4.5**	**3.7**	**7.6**[d]	**8.9**	**5.3**	**21**[d]	**3.7**[d]	**5.3**[d]	**3.5**	**3.2**	**31**	**15**	**10**	**8.2**	**17**[d]	**20**	**12**	**46**[d]	**8.2**[d]	**12**[d]
PCB TEQ - fish	0.052 (0.27)	0.052 (0.27)	0.37 (**1.9**)	0.18 (0.91)	0.15 (0.78)	0.11 (0.57)	0.27[d] (**1.4**[d])	0.29 (**1.5**)	0.10 (0.52)	**1.4**[d] **(7.4**[d]**)**	0.12[d] (0.61[d])	0.19[d] (**1.0**[d])	0.70	0.70	**4.9**	**2.4**	**2.0**	**1.5**	**3.6**[d]	**3.9**	**1.3**	**19**[d]	**1.6**[d]	**2.6**[d]
PCDDs/PCDFs																								
2,3,7,8-TCDD	**4.1 (21)**	**3.8 (20)**	**51 (270)**	**16 (83)**	**8.0 (42)**	**13 (65)**	**11**[d] **(57)**[d]	**18 (91)**	**1.9 (10)**	**15**[d] **(78)**[d]	**6.3**[d] **(33)**[d]	**7.9**[d] **(41)**[d]	**55**	**52**	**690**	**210**	**110**	**170**	**150**[d]	**240**	**26**	**200**[d]	**85**[d]	**110**[d]
PCDD/PCDF TEQ - fish	**4.3 (22)**	**4.1 (21)**	**52 (270)**	**17 (87)**	**8.3 (43)**	**13 (70)**	**11**[d] **(57)**[d]	**18 (96)**	**2.0 (10)**	**15**[d] **(78)**[d]	**6.3**[d] **(33)**[d]	**8.3**[d] **(43)**[d]	**57**	**55**	**700**	**230**	**110**	**180**	**150**[d]	**250**	**27**	**200**[d]	**85**[d]	**110**[d]
Total TEQ - fish	**4.3 (22)**	**4.1 (21)**	**52 (270)**	**17 (87)**	**8.3 (43)**	**13 (70)**	**11**[d] **(57)**[d]	**19 (100)**	**2.1 (11)**	**15**[d] **(78)**[d]	**6.8**[d] **(36)**	**9.2**[d] **(48)**[d]	**57**	**55**	**700**	**230**	**110**	**180**	**150**[d]	**260**	**28**	**200**[d]	**92**[d]	**120**[d]
Organochlorine Pesticides																								
Dieldrin	0.092	0.13	0.46	0.26	0.39	0.25	0.21[d]	0.23	0.45	0.33[d]	0.17[d]	0.36[d]	**1.4**	**2.0**	**6.9**	**3.9**	**5.9**	**3.8**	**3.1**[d]	**3.4**	**6.8**	**5.0**[d]	**2.5**[d]	**5.4**[d]
Endosulfan I	0.26	0.48	**1.3**	0.071	0.65	0.35	0.15[d]	0.55	0.19	**3.2**[d]	0.94[d]	0.68[d]	nc	nc	nc	nc	nc	nc	nc	nc	nc	nc	nc	nc
Total DDx	**1.3**	**1.4**	**13**	**4.6**	**5.4**	**3.1**	**2.9**[d]	**6.7**	**5.0**	**3.1**[d]	**4.4**[d]	**5.4**[d]	0.85	0.96	**8.3**	**3.1**	**3.6**	**2.1**	**1.9**[d]	**4.5**	**3.3**	**2.1**[d]	**2.9**[d]	**3.6**[d]

Bold identifies HQs ≥ 1.0.

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

[b] TRVs were derived from the primary literature review.

[c] TRVs were based on USEPA's revised draft of the LPR restoration project FFS (Louis Berger et al. 2014) or first draft of the LPR restoration project FFS.

[d] Fewer than six detected concentrations available, so HQ based on a maximum concentration rather than a UCL.

[e] No NOAEL was selected because the LOAEL was based on ED10 value for the most sensitive species evaluated, below which adverse effects are not expected.

COPEC – chemical of potential ecological concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
ED10 – dose that corresponds to a 10% increase in an adverse effect of an exposed population
FFS – focused feasibility study
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
LPR – Lower Passaic River
nc – not calculated
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
PCDD– polychlorinated dibenzo-*p*-dioxin
PCDF –polychlorinated dibenzofuran
TCDD – tetrachlorodibenzo-*p*-dioxin
TEQ – toxic equivalent
TRV – toxicity reference value
UCL – upper confidence limit on the mean
USEPA – US Environmental Protection Agency
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT and 4,4′-DDT)

7.1.4.2 Uncertainties in risk characterization

This section discusses uncertainties associated with exposure assumptions and EPCs that could affect HQ calculations for fish tissue. Uncertainties associated with the TEQ methodology are presented in Section 4.1, and general TRV uncertainties are discussed in Sections 6.3.3.1 and 7.1.3.1. The uncertainties addressed in this section are as follows:

- **Treatment of non-detects for EPCs –** The concentrations of sum components (e.g., PCB congeners) that were not detected were assumed to be zero when calculating totals. The effect on non-TEQ sum HQs of using one-half the DL or the full DL was evaluated. For TEQ sums, EPCs were derived using USEPA's TEQ calculator (USEPA 2014) using the Kaplan-Meier method. The effect on TEQ-HQs of using zero, one-half DL, or the full DL was also evaluated.
- **Use of maximum concentrations as EPCs –** Maximum concentrations were used to represent EPCs for several species (i.e., largemouth bass, smallmouth bass, northern pike, and white sucker) with small sample sizes. This uncertainty was not empirically evaluated because too few samples were available for the calculation of a UCL. Risk estimates based on the maximum concentrations of limited samples are uncertain. If instead, a measure of central tendency (i.e., a mean concentration) was used as the EPC for these fish species, calculated HQs would be lower, and some HQs would be < 1.0 for methylmercury and total PCBs.
- **Use of UCLs as EPCs –** Tissue EPCs for most fish species (i.e., mummichog, other forage fish, common carp, channel catfish, brown bullhead, white catfish, white perch, and American eel) were based on a conservative upper-bound estimate of the mean concentration (i.e., UCL concentration). If instead, a measure of central tendency (i.e., a mean concentration) was used as the EPC for these fish species, calculated HQs would be lower.

The effects of the first uncertainty (i.e., treatment of non-detects for EPCs) on HQ calculations for fish tissue are presented in Table 7-10. The treatment of non-detected values in sums generally has no effect on the HQ.

Table 7-10. Fish tissue LOAEL HQs based on uncertainties in exposure assumptions and EPCs

Uncertainty	Parameter Values/Assumptions		Range of LOAEL HQs[a]											
			Total PCBs				PCDD/PCDF TEQ - Fish				Total TEQ - Fish			
			HQ Based on TRV-A[b]		HQ Based on TRV-B[c]		HQ Based on TRV-A[c]		HQ Based on TRV-B[c]		HQ Based on TRV-A[b]		HQ Based on TRV-B[c]	
	Original	Adjusted	Orig.	Adj.	Orig.	Adj.	Orig.	Adj.	Orig.	Adj.	Orig.	Adj.	Orig.	Adj.
Benthic omnivore (mummichog, other forage fish)														
Treatment of non-detects	DL = 0 for non-detects	use of one-half the DL or the full DL for non-detects[d]	0.14–0.16	0.14–0.16	**1.0–1.1**	**1.0–1.1**	0.41–0.43	0.41–0.43	**27–28**	**27–28**	0.41–0.43	0.41–0.43	**27–28**	**27–28**
		use of Kaplan-Meier method in USEPA's TEQ calculator (USEPA 2014)		na		na		0.41–0.43		ne		0.41–0.43		**27–28**
Invertivore (white perch, channel catfish, brown bullhead)														
Treatment of non-detects	DL = 0 for non-detects	use of one-half the DL or the full DL for non-detects[d]	0.37–0.66	0.37–0.66	2.6–4.7	**2.6–4.7**	0.83–**1.7**	0.83–**1.7**	**56–110**	**56–110**	0.83–**1.7**	0.83–**1.7**	**56–110**	**56–110**
		use of Kaplan-Meier method in USEPA's TEQ calculator (USEPA 2014)		na		na		0.83–**1.7**		ne		0.83–**1.7**		**56–110**

Uncertainty	Parameter Values/Assumptions		Range of LOAEL HQs[a]											
			Total PCBs				PCDD/PCDF TEQ - Fish				Total TEQ - Fish			
			HQ Based on TRV-A[b]		HQ Based on TRV-B[c]		HQ Based on TRV-A[c]		HQ Based on TRV-B[c]		HQ Based on TRV-A[b]		HQ Based on TRV-B[c]	
	Original	Adjusted	Orig.	Adj.	Orig.	Adj.	Orig.	Adj.	Orig.	Adj.	Orig.	Adj.	Orig.	Adj.
Piscivore (American eel, largemouth bass)														
Treatment of non-detects	DL = 0 for non-detects	use of one-half the DL or the full DL for non-detects[d]	0.53–**2.1**	0.53–**2.1**	**3.8–15**	**3.8–15**	0.20–**1.5**	0.21–**1.5**	**13–100**	**14–100**	0.21–**1.5**	0.22–**1.5**	**14–100**	**14–100**
		use of Kaplan-Meier method in USEPA's TEQ calculator (USEPA 2014)		na		na		0.21–**1.5**		ne		0.21–**1.5**		**14–100**

Bold identifies HQs ≥ 1.0.

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

[b] TRVs were derived from the primary literature review.

[c] TRVs were based on USEPA's revised draft of the LPR restoration project FFS (Louis Berger et al. 2014) or first draft of the LPR restoration project FFS (Malcolm Pirnie 2007b).

[d] LOAEL HQs are the same, regardless of treatment of non-detected values as one-half the DL or as the full DL.

DL – detection limit
EPC – exposure point concentration
FFS – focused feasibility study
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
LPR – Lower Passaic River
na – not applicable
PCB – polychlorinated biphenyl
PCDD – polychlorinated dibenzo-*p*-dioxin
PCDF – polychlorinated dibenzofuran
TEQ – toxic equivalent
USEPA – US Environmental Protection Agency

7.1.4.3 Tissue EFs for regulated metals

Tissue NOAEL and LOAEL EFs for regulated metals are presented in Table 7-11 for all fish species (mummichog, other forage fish, common carp, white perch, channel catfish, brown bullhead, white sucker, white catfish, American eel, largemouth bass, smallmouth bass, and northern pike).

Table 7-11. Fish tissue LOAEL and NOAEL EFs for regulated metals

COPEC	Range of EFs[a]																							
	EF based on TRV-A[b]												EF based on TRV-B[c]											
	Benthic Omnivores			Invertivore					Piscivore				Benthic Omnivores			Invertivore					Piscivore			
	Mummichog	Other Forage Fish	Common Carp	White Perch	Channel Catfish	Brown Bullhead	White Sucker	White Catfish	American Eel	Largemouth Bass	Smallmouth Bass	Northern Pike	Mummichog	Other Forage Fish	Common Carp	White Perch	Channel Catfish	Brown Bullhead	White Sucker	White Catfish	American Eel	Largemouth Bass	Smallmouth Bass	Northern Pike
LOAEL EF																								
Arsenic	0.15	0.14	0.060	0.088	0.022	0.052	0.044[d]	0.034	0.11	0.027[d]	0.10[d]	0.048[d]	nc	nc	nc	nc	nc	nc	nc	nc	nc	nc	nc	nc
Cadmium	0.28	0.36	0.20	0.088	0.088	0.17	0.081[d]	0.088	0.55	0.23[d]	0.056[d]	0.022[d]	nc	nc	nc	nc	nc	nc	nc	nc	nc	nc	nc	nc
Copper	nc[e]	nc[e]	nc[e]	nc[e]	nc[e]	nc[e]	nc[e]	nc[e]	nc[e]	nc[e]	nc[e]	nc[e]	**2.1**	**2.7**	0.73	**9.3**	0.87	0.57	0.73[d]	0.45	**1.7**	0.39[d]	0.53[d]	0.38[d]
Lead	0.60	0.75	0.20	0.11	0.075	0.20	0.07 5[d]	0.19	0.22	0.030[d]	0.025[d]	0.0083[d]	0.60	0.75	0.20	0.11	0.075	0.20	0.075[d]	0.19	0.22	0.030[d]	0.025[d]	0.0083[d]
Silver	0.18	0.19	0.063[d]	0.83	0.058[d]	0.033[d]	0.021[d]	0.014	0.10	0.011[d]	0.012[d]	0.012[d]	nc	nc	nc	nc	nc	nc	nc	nc	nc	nc	nc	nc
Zinc	0.11	0.090	0.19	0.065	0.050	0.073	0.052[d]	0.042	0.077	0.040[d]	0.046[d]	0.084[d]	nc	nc	nc	nc	nc	nc	nc	nc	nc	nc	nc	nc
NOAEL EF																								
Arsenic	0.29	0.26	0.12	0.17	0.042	0.10	0.085[d]	0.066	0.22	0.052[d]	0.19[d]	0.092[d]	nc	nc	nc	nc	nc	nc	nc	nc	nc	nc	nc	nc
Cadmium	**2.8**	**3.6**	**2.0**	0.88	0.88	**1.7**	0.81[d]	0.88	**5.5**	**2.3**[d]	0.56[d]	0.22[d]	nc	nc	nc	nc	nc	nc	nc	nc	nc	nc	nc	nc
Copper	nc[e]	nc[e]	nc[e]	nc[e]	nc[e]	nc[e]	nc[e]	nc[e]	nc[e]	nc[e]	nc[e]	nc[e]	**9.7**	**13**	**3.4**	**44**	**4.1**	**2.7**	**3.4**[d]	**2.1**	**8.1**	**1.8**[d]	**2.5**[d]	**1.8**[d]
Lead	0.96	**1.2**	0.32	0.18	0.12	0.32	0.12[d]	0.30	0.35	0.048[d]	0.039[d]	0.013[d]	**6.0**	**7.5**	**2.0**	**1.1**	0.75	**2.0**	0.75[d]	**1.9**	**2.2**	0.30[d]	0.25[d]	0.083[d]
Silver	0.40	0.42	0.14[d]	**1.8**	0.13[d]	0.073[d]	0.045[d]	0.030	0.23	0.024[d]	0.025[d]	0.025[d]	nc	nc	nc	nc	nc	nc	nc	nc	nc	nc	nc	nc
Zinc	0.16	0.13	0.26	0.091	0.070	0.10	0.073[d]	0.059	0.11	0.056[d]	0.064[d]	0.12[d]	nc	nc	nc	nc	nc	nc	nc	nc	nc	nc	nc	nc

Bold identifies EFs ≥ 1.0.

Shaded cells identify EFs ≥ 1.0 based on a LOAEL TRV.

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

[b] TRVs were derived from the primary literature review.

[c] TRVs were based on USEPA's revised draft of the LPR restoration project FFS (Louis Berger et al. 2014) or first draft of the LPR restoration project FFS (Malcolm Pirnie 2007b).

[d] Fewer than six detected samples were available, so the EF was based on a maximum concentration rather than a UCL concentration.

[e] A TRV was not derived; see Section 7.1.3.2.

COPEC – chemical of potential ecological concern
EF – exceedance factor
FFS – focused feasibility study
LOAEL – lowest-observed-adverse-effect level
LPR – Lower Passaic River
nc – not calculated
NOAEL – no-observed-adverse-effect level
TRV – toxicity reference value
UCL – upper confidence limit on the mean
USEPA – US Environmental Protection Agency

7.1.4.4 Comparison to background

Consistent with USEPA guidance (USEPA 2002c), this section presents background concentrations and risk estimates calculated for fish species-COPEC pairs with LOAEL HQs ≥ 1.0. Three background datasets were developed for use in this BERA using available data from the following areas: 1) upstream of Dundee Dam, to represent freshwater urban habitat, 2) Jamaica Bay/Lower Harbor, to represent estuarine urban habitat, and 3) Mullica River/Great Bay, to represent estuarine rural habitat. These datasets are summarized in Section 4.2, and details on how background values were determined from these datasets are presented in Appendix J. Table 7-12 presents a comparison of LPRSA fish tissue concentrations to background area concentrations, as available, for fish COPECs with LOAEL HQs ≥ 1.0. Background data from above Dundee Dam were available for comparison to LPRSA data for 10 species. Background data from Jamaica Bay/Lower Harbor and Mullica River/Great Bay were available for only a subset of COPECs for mummichog; these data were compared to LPRSA mummichog data.

Table 7-12. LPRSA fish tissue compared to background tissue for fish COPECs with LOAEL HQs ≥ 1.0

Species by COPEC[a]	Units (ww)	LPRSA				Above Dundee Dam				Jamaica Bay/Lower Harbor				Mullica River/Great Bay				LOAEL TRV	
		N	EPC	Min. Detect	Max. Detect	N	UCL	Min. Detect	Max. Detect	N	UCL	Min. Detect	Max. Detect	N	UCL	Min. Detect	Max. Detect	TRV- A[b]	TRV-B[c]
Mercury																			
Mummichog/killifish	µg/kg	18	63	36	71	1	na	40.4	40.4	7	64.2	14.6	76.7	10	24	6.8	38	350	260
Other forage fish	µg/kg	10	83	30	150	2	na	71.9	125	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]		
Common carp	µg/kg	12	80	42	110	10	110	43.4	133	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]		
White perch	µg/kg	22	200	33	**530**	8	**300**	139	**390**	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]		
Brown bullhead	µg/kg	6	110	48	140	6	189	29.7	254	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]		
Channel catfish	µg/kg	11	150	32	230	4	na	146	**555**	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]		
White sucker	µg/kg	5	140	77	140	5	na	60.9	229	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]		
American eel	µg/kg	21	**260**	74	**390**	16	250	148	**324**	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]		
Northern pike	µg/kg	1	220	220	220	1	na	**364**	**364**	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]		
Smallmouth bass	µg/kg	3	**300**	180	**300**	3	na	198	236	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]		
Methylmercury																			
Mummichog/killifish	µg/kg	18	53	19	69	1	na	34.5	34.5	2	71.4	69.2	71.4	na[e]	na[e]	na[e]	na[e]	350	260
Other forage fish	µg/kg	10	70	14	150	2	na	61.7	110	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]		
Common carp	µg/kg	12	62	39	90	10	110	47.5	131	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]		
White perch	µg/kg	22	170	25	330	8	**270**	120	**373**	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]		
Brown bullhead	µg/kg	6	92	39	120	6	203	29.7	**276**	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]		
Channel catfish	µg/kg	11	140	30	230	4	na	140	**559**	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]		
White sucker	µg/kg	5	130	71	130	5	na	51.3	196	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]		
American eel	µg/kg	21	**280**	92	470	16	190	121	255	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]		
Northern pike	µg/kg	1	180	180	180	1	na	316	**316**	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]		
Smallmouth bass	µg/kg	3	220	140	220	3	na	139	162	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]		
Total PCBs																			
Mummichog/killifish	µg/kg	18	**600**	240	930	1	na	219	219	7	1,900	55	**3,200**	na[e]	na[e]	na[e]	na[e]	3,800	530
Other forage fish	µg/kg	10	**550**	170	870	2	na	107	**853**	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]		
Common carp	µg/kg	12	**5,200**	1,500	7,900	10	**2,100**	**755**	**2,560**	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]		
White perch	µg/kg	22	**2,500**	290	5,100	8	**834**	408	**1,130**	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]		
Brown bullhead	µg/kg	6	**1,400**	260	1,700	6	519	183	**614**	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]		
Channel catfish	µg/kg	11	**1,700**	350	2,700	4	na	**948**	**2,130**	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]		
White sucker	µg/kg	5	**2,900**	540	2,900	5	na	327	**872**	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]		
American eel	µg/kg	21	**2,000**	420	5,700	16	**1,080**	206	**1,880**	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]		
Northern pike	µg/kg	1	**2,000**	2,000	2,000	1	na	**1,880**	**1,880**	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]		
Smallmouth bass	µg/kg	3	**1,400**	630	1,400	3	na	**1,000**	**1,310**	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]		

Table 7-12. LPRSA fish tissue compared to background tissue for fish COPECs with LOAEL HQs ≥ 1.0

Species by COPEC[a]	Units (ww)	LPRSA				Above Dundee Dam				Jamaica Bay/Lower Harbor				Mullica River/Great Bay				LOAEL TRV	
		N	EPC	Min. Detect	Max. Detect	N	UCL	Min. Detect	Max. Detect	N	UCL	Min. Detect	Max. Detect	N	UCL	Min. Detect	Max. Detect	TRV- A[b]	TRV-B[c]
PCB TEQ - fish																			
Mummichog/killifish	ng/kg	18	0.62	0.27	0.89	1	na	0.275	0.275	7	6.7	0.0079	**13**	10	0.18	0.0061	0.18	120 (23[d])	1.8
Other forage fish	ng/kg	10	0.62	0.19	0.9	2	na	0.147	0.968	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]		
Common carp	ng/kg	12	**4.4**	1.4	6.5	10	**2.56**	0.688	**4.68**	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]		
White perch	ng/kg	22	**2.1**	0.28	3.4	8	0.924	0.519	1.04	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]		
Brown bullhead	ng/kg	6	1.3	0.42	1.6	6	0.583	0.261	0.676	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]		
Channel catfish	ng/kg	11	**1.8**	0.23	2.8	4	na	1.28	**3.37**	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]		
White sucker	ng/kg	5	**3.2**	0.81	3.2	5	na	0.318	1.15	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]		
American eel	ng/kg	21	1.2	0.31	2.4	16	0.963	0.143	1.23	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]		
Northern pike	ng/kg	1	**2.3**	**2.3**	2.3	1	na	**2.56**	**2.56**	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]		
Smallmouth bass	ng/kg	3	1.4	0.69	1.4	3	na	1.17	1.47	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]		
PCDD/PCDF TEQ - fish																			
Mummichog/killifish	ng/kg	18	**51**	11	100	1	na	0.4	0.4	7	**20**	0.036	**25**	12	0.43	0.0073	0.47	120 (23[d])	1.8
Other forage fish	ng/kg	10	**49**	3.7	96	2	na	0.11	**2.5**	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]		
Common carp	ng/kg	12	**620**	8.5	1400	10	**5.94**	3.13	**7.27**	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]		
White perch	ng/kg	22	**200**	19	260	8	**2.52**	1.42	**3.09**	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]		
Brown bullhead	ng/kg	6	**160**	8.4	200	6	**2.24**	1.09	**2.64**	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]		
Channel catfish	ng/kg	11	**100**	23	170	4	na	3.3	**8.83**	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]		
White sucker	ng/kg	5	**130**	4.1	130	5	na	0.619	**2.55**	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]		
American eel	ng/kg	21	**24**	0.79	49	16	**1.44**	0.135	**2.52**	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]		
Northern pike	ng/kg	1	**100**	100	100	1	na	4.74	**4.74**	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]		
Smallmouth bass	ng/kg	3	**76**	8.6	76	3	na	1.72	**2.00**	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]		
Total TEQ - fish																			
Mummichog/killifish	ng/kg	18	**51**	12	100	1	0.676	0.676	0.676	7	**20**	0.044	**25**	10	0.35	0.013	0.49	120 (23[d])	1.8
Other forage fish	ng/kg	10	**49**	4.3	97	2	**3.47**	0.265	**3.47**	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]		
Common carp	ng/kg	12	**620**	9.9	1400	10	8.23	3.97	**9.18**	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]		
White perch	ng/kg	22	**200**	19	270	8	3.45	1.94	**4.14**	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]		
Brown bullhead	ng/kg	6	**160**	8.8	200	6	2.79	1.35	**3.32**	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]		
Channel catfish	ng/kg	11	**100**	23	170	4	**12.1**	4.57	**12.1**	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]		
White sucker	ng/kg	5	**130**	4.9	130	5	**3.7**	0.937	**3.7**	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]		
American eel	ng/kg	21	**25**	1.2	50	16	2.37	0.285	**3.74**	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]		
Northern pike	ng/kg	1	**110**	110	110	1	**7.3**	7.3	**7.3**	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]		
Smallmouth bass	ng/kg	3	**82**	9.8	82	3	**3.48**	2.98	**3.48**	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]		

Table 7-12. LPRSA fish tissue compared to background tissue for fish COPECs with LOAEL HQs ≥ 1.0

Species by COPEC[a]	Units (ww)	LPRSA				Above Dundee Dam				Jamaica Bay/Lower Harbor				Mullica River/Great Bay				LOAEL TRV	
		N	EPC	Min. Detect	Max. Detect	N	UCL	Min. Detect	Max. Detect	N	UCL	Min. Detect	Max. Detect	N	UCL	Min. Detect	Max. Detect	TRV- A[b]	TRV-B[c]
Dieldrin																			
Mummichog/killifish	µg/kg	18	11	3.5	28	1	na	18.9	18.9	7	21.5	2.16	34.3	10	0.92	0.92	0.92	200	40
Other forage fish	µg/kg	10	16	8.4	22	2	na	9.4	22	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]		
Common carp	µg/kg	12	**55**	29	72	10	31.8	12.7	34	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]		
White perch	µg/kg	22	31	7.8	47	8	22	16	25	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]		
Brown bullhead	µg/kg	6	30	9.7	34	6	18	2.54	25.8	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]		
Channel catfish	µg/kg	11	**47**	18	70	4	na	12.4	27.7	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]		
White sucker	µg/kg	5	25	16	25	5	na	4.7	**43.9**	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]		
American eel	µg/kg	21	**54**	7.6	110	16	**74**	3.1	**127**	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]		
Northern pike	µg/kg	1	**43**	43	43	1	na	38	38	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]		
Smallmouth bass	µg/kg	3	20	16	20	3	na	18	21	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]		
Total DDx																			
Mummichog/killifish	µg/kg	18	66	26	100	1	na	45	45	7	180	10	240	nd	nd	nd	nd	520	390
Other forage fish	µg/kg	10	75	22	140	2	na	30	120	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]		
Common carp	µg/kg	12	**650**	110	1100	10	220	87	280	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]		
White perch	µg/kg	22	240	38	490	8	150	85	170	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]		
Brown bullhead	µg/kg	6	160	20	200	6	67	27	76	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]		
Channel catfish	µg/kg	11	280	48	490	4	na	120	340	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]		
White sucker	µg/kg	5	150	63	150	5	na	33	170	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]		
American eel	µg/kg	21	260	32	470	16	270	62	**490**	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]		
Northern pike	µg/kg	1	280	280	280	1	na	230	230	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]		
Smallmouth bass	µg/kg	3	230	100	230	3	na	140	150	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]	na[e]		

Note: The maximum detected concentration for background areas exclude outlier concentrations as described in Appendix J.

Bold identifies LOAEL HQs ≥ 1.0.

a Only COPECs with HQs ≥ 1.0 based on LOAEL TRV included in table.

b TRVs were derived from the primary literature review based on the process identified in Section 7.1.3.1.

c TRVs were based on USEPA's revised draft of the LPR restoration project FFS (Louis Berger et al. 2014) or first draft of the LPR restoration project FFS (Malcolm Pirnie 2007b).

d HQs in parenthesis were based on additional alternative SSD-derived LOAELs evaluated (see text in Section 7.1.3 for details).

e Data were not available.

COPEC – chemical of potential ecological concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
EPC – exposure point concentration
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
LPR – Lower Passaic River
LPRSA – Lower Passaic River Study Area
na – not applicable
PCB – polychlorinated biphenyl
PCDD– polychlorinated dibenzo-*p*-dioxin
PCDF – polychlorinated dibenzofuran
SSD – species sensitivity distribution
TEQ – toxic equivalent
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT and 4,4′-DDT)
TRV – toxicity reference value
UCL – upper confidence limit on the mean
USEPA – US Environmental Protection Agency
ww – wet weight

LPRSA fish tissue EPCs for American eel, brown bullhead, common carp, channel catfish, mummichog, northern pike, other forage fish, smallmouth bass, white perch, and white sucker compared to those above Dundee Dam by COPEC are summarized as follows:

- For methylmercury, LPRSA tissue EPCs were less than maximum concentrations for 7 of 10 fish species above Dundee Dam, and less than UCLs for 3 of 4 fish species for which UCLs above Dundee Dam could be calculated. For mercury, LPRSA tissue EPCs were less than maximum concentrations for 8 of 10 fish species above Dundee Dam, and less than UCLs for 3 of 4 fish species for which UCLs above Dundee Dam could be calculated.
- For total PCBs, LPRSA tissue EPCs were less than maximum concentrations for 2 of 10 fish species above Dundee Dam, but greater than UCLs of all 4 fish species for which UCLs above Dundee Dam could be calculated.
- For PCB TEQ - fish, LPRSA tissue EPCs were less than maximum concentrations for 6 of 10 fish species above Dundee Dam, but greater than UCLs of all 4 fish species for which UCLs above Dundee Dam could be calculated.
- For PCDD/PCDF TEQ - fish and total TEQ - fish, LPRSA tissue EPCs were greater than maximum concentrations for all 10 fish species above Dundee Dam, and greater than UCLs of all 4 fish species for which UCLs above Dundee Dam could be calculated.
- For dieldrin, LPRSA tissue EPCs were less than maximum concentrations for 5 of 10 fish species above Dundee Dam, but greater than UCLs of all 4 fish species for which UCLs above Dundee Dam could be calculated.
- For total DDx, LPRSA tissue EPCs were less than maximum concentrations for 4 of 10 fish species above Dundee Dam, and less than the UCL for 1 of the 4 fish species for which UCLs above Dundee Dam could be calculated.

The comparison of LPRSA mummichog EPCs with HQ ≥ 1.0 to EPCs from Jamaica Bay/Lower Harbor and Mullica River/Great Bay is summarized as follows:

- For total PCBs, LPRSA mummichog tissue concentrations were less than Jamaica Bay/Lower Harbor maximum concentrations and UCLs. No Mullica River/Great Bay data were available for total PCB congeners.
- For PCDD/PCDFs TEQ - fish and total TEQ – fish, LPRSA mummichog tissue concentrations were greater than Jamaica Bay/Lower Harbor and Mullica River/Great Bay maximum concentrations and UCLs.

7.1.5 Summary of key uncertainties

The primary uncertainty associated with the fish tissue risk characterization is the use of the tissue LOE for inorganic metals, as discussed in Section 6.3.3.1. Whole-body tissue concentrations of total metals (other than organometals, including methylmercury, organo-selemium, and butyltins) are poorly predictive of adverse effects for several reasons: toxicity is caused by specific metal species, fish store excess metals in non-toxic compartments, and toxicity is strongly dependent on the rate and exposure pathway. For these reasons, USEPA risk assessment guidance for metals and recent expert guidance conclude that comparison of whole-body metals tissue concentrations to literature-reported whole-body effects thresholds is not sufficiently robust for drawing risk conclusions (Adams et al. 2011; USEPA 2007e). Further details on key uncertainties for individual metals are presented in Section 7.1.3.2.

An additional important uncertainty associated with the tissue LOE is the use of EPCs based on maximum concentrations for species with limited samples (i.e., largemouth bass [n = 3], smallmouth bass [n = 3], northern pike [n = 1], and white sucker [n = 5]). There is a large amount of uncertainty associated with these risk results because of the small sample size. Other uncertainties in the fish tissue assessment, such as the TEQ methodology and the use of laboratory toxicity studies to predict effects, could either under- or overestimate risks. However, the HQs are more likely to overestimate risk because of a number of conservative assumptions used in the risk evaluation, such as the use of the 5th percentile LOAEL among all species or endpoints as the TRV, and the use of an upper exposure value (i.e., UCL) as the EPC.

Specific uncertainties associated with TRVs, including the derivation of TRVs using SSDs, are discussed in Sections 7.1.3.2 and 6.3.3.1. For the COPECs with TRVs based on 5th percentile LOAELs determined from SSDs (i.e., mercury/methylmercury, total PCBs, TEQ - fish, total DDx, and cadmium), the range of the empirical LOAELs and number of data points (i.e., number of species included in the SSD) are shown in Table 7-13 to provide context of uncertainty for SSD-derived values.

Table 7-13. Uncertainty evaluation of fish tissue TRVs based on SSDs

COPEC	TRV			No. of species (count of LOAELs in SSD)	No. ACR-adjusted LOAELs/No. LOAELs in SSD	Empirical LOAEL Range	Notes on Key Uncertainties
	Unit (ww)	NOAEL	LOAEL				
Methylmercury/ mercury	µg/kg	35	350	n = 12	2 / 12	470 to 22,000	• SSD-derived LOAEL < lowest measured LOAEL
Total PCBs	µg/kg	380	3,800	n = 11	3 / 11	9,300 – 645,000	• SSD-derived LOAEL < lowest measured LOAEL • removal of acute studies (ACR-extrapolated LOAELs) from SSDs results in twofold increase in TRV
TEQ - fish	ng/kg	12 (2.3)	120 (23[a])	n = 7	1 / 7	85 – 14,400	• Alternative SSD-derived LOAEL

COPEC	TRV			No. of species (count of LOAELs in SSD)	No. ACR-adjusted LOAELs/No. LOAELs in SSD	Empirical LOAEL Range	Notes on Key Uncertainties
	Unit (ww)	NOAEL	LOAEL				
							< lowest measured LOAEL
Total DDx	µg/kg	52[b]	520	n = 7	0 / 7	1,100 – 200,000	• SSD-derived LOAEL and alternate SSD-derived LOAEL both < lowest measured LOAEL
Cadmium	mg/kg	0.016	0.16	n = 13	5 / 13	0.12 – 144	• SSD-derived LOAEL is within range of measured LOAELs

Note: TRVs included in this table are based on SSDs that are based on TRVs derived from the general literature search.

ACR – acute-to-chronic ratio
COPEC – chemical of potential ecological concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
SSD – species sensitivity distribution
TEQ – toxic equivalent
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT and 4,4′-DDT)
TRV – toxicity reference value
ww – wet weight

7.1.6 Summary

Seventeen COPECs were evaluated in whole-body tissue for 12 fish species. HQs were ≥ 1.0 for one or more species for methylmercury/mercury, total PCBs, PCB TEQ - fish, 2,3,7,8-TCDD, PCDD/PCDF TEQ - fish, total TEQ - fish, total DDx, and dieldrin. HQs calculated for white sucker, largemouth bass, northern pike, and smallmouth bass were based on maximum concentrations because fewer than six samples were available for each species (white sucker [n = 5], largemouth bass [n = 3], northern pike [n = 1], and smallmouth bass [n = 3]). There is a large amount of uncertainty associated with these risk results because of the small sample size. A summary of the fish tissue LOAEL HQs is presented in Table 7-14.

Table 7-14. Summary of fish tissue LOAEL HQs

COPEC[b]	Range of LOAEL HQs[a]				Key Uncertainties
	HQ Based on TRV-A[c]		HQ Based on TRV-B[d]		
	Fish Species with HQs ≥ 1.0	LOAEL HQ Values ≥ 1.0	Fish Species with HQs ≥ 1.0	LOAEL HQ Values ≥ 1.0	
Organic metals					
Methylmercury/mercury	HQs ≥ 1.0 for largemouth bass only	**1.5–1.9**	HQs ≥ 1.0 for largemouth bass, American eel, smallmouth bass, and white catfish-	0.85**–2.6**	• TRV-A and TRV-B derived using SSDs • EPC for largemouth bass based on maximum tissue concentration (insufficient data for UCL derivation; n=3)
PCBs					
Total PCBs	HQs ≥ 1.0 for largemouth bass and common carp	**1.4–2.1**	HQs ≥ 1.0 for all fish species evaluated	**1.0–15**	• TRV-A based on SSD less than lowest measured LOAEL • TRV-B based on changes in smolt seawater preference in Atlantic salmon • EPC for largemouth bass based on maximum tissue concentration (insufficient data for UCL derivation; n=3)
PCB TEQ - fish	HQs < 1.0 for all fish species evaluated	all HQs < 1.0	HQs ≥ 1.0 for white perch, channel catfish, largemouth bass, common carp, white catfish, white sucker, and northern pike	**1.0–9.4**	• TRV-A based on SSD within range of measured LOAELs evaluated • Alternative TRV-A based on SSD with relatively poor visual and statistical fits to the empirical data, likely over-predicts risk; alternative SSD less than lowest measured LOAEL • TRV-B based on interpolated larvae concentration from egg tissue

COPEC[b]	Range of LOALEL HQs[a]				Key Uncertainties
	HQ Based on TRV-A[c]		HQ Based on TRV-B[d]		
	Fish Species with HQs ≥ 1.0	LOAEL HQ Values ≥ 1.0	Fish Species with HQs ≥ 1.0	LOAEL HQ Values ≥ 1.0	
PCDDs/PCDFs					
2,3,7,8-TCDD	HQs ≥ 1.0 for white perch, largemouth bass, white catfish, white sucker, brown bullhead, and common carp (HQs ≥ 1.0 for all fish species[d] evaluated)	**1.1–5.1 (1.0–9.1[e])**	HQs ≥ 1.0 for all fish species evaluated	**13–340**	• TRV-A based on SSD within range of measured LOAELs evaluated • Alternative TRV-A based on SSD with relatively poor visual and statistical fits to the empirical data, likely over-predicts risk; alternative SSD less than lowest measured LOAEL • TRV-B based on interpolated larvae concentration from egg tissue
PCDD/PCDF TEQ - fish	HQs ≥ 1.0 for white perch, brown bullhead, largemouth bass, white catfish, white sucker, and common carp (HQs ≥ 1.0 for all fish species[d] evaluated)	**1.1–5.2 (1.0–9.6[e])**	HQs ≥ 1.0 for all fish species evaluated	**13–340**	
Total TEQ - fish	HQs ≥ 1.0 for white perch, brown bullhead, largemouth bass, white sucker, white catfish, and common carp (HQs ≥ 1.0 for all fish species[d] evaluated)	**1.1–5.2 (1.1–10[e])**	HQs ≥ 1.0 for all fish species evaluated	**14–340**	
Organochlorine Pesticides					
Dieldrin	HQs < 1.0 for all fish species evaluated	all HQs < 1.0	HQs ≥ 1.0 for channel catfish, American eel, largemouth bass,	**1.0–1.4**	• Limited toxicity dataset for TRV derivation • TRV-B derived using extrapolation factors from 96-hr study

COPEC[b]	Range of LOAEL HQs[a] — HQ Based on TRV-A[c] — Fish Species with HQs ≥ 1.0	LOAEL HQ Values ≥ 1.0	HQ Based on TRV-B[d] — Fish Species with HQs ≥ 1.0	LOAEL HQ Values ≥ 1.0	Key Uncertainties
			northern pike, and common carp		
Total DDx	HQs ≥ 1.0 for common carp only	**1.3**	HQs ≥ 1.0 for common carp only	**1.7**	• TRV-A based on SSD less than lowest measured LOAEL evaluated • TRV-B based on SSD within range of measured LOAELs evaluated (including TRVs based on field-collected organisms)

Bold identified HQs ≥ 1.

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

b Only COPECs with HQs ≥ 1.0 based on a LOAEL TRV are included in the table.

c TRVs were derived from the primary literature review based on process identified in Section 7.1.3.1.

d TRVs were based on USEPA's revised draft of the LPR restoration project FFS (Louis Berger et al. 2014) or first draft of the LPR restoration project FFS (Malcolm Pirnie 2007b).

e HQs in parenthesis were based on additional alternative SSD-derived LOAEL evaluated (see text in Section 7.1.3 for details).

COPEC – chemical of potential ecological concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
EPC – exposure point concentration
FFS – focused feasibility study
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
LPR – Lower Passaic River
PCB – polychlorinated biphenyl
PCDD– polychlorinated dibenzo-*p*-dioxin
PCDF –polychlorinated dibenzofuran
SSD – species sensitivity distribution
TCDD – tetrachlorodibenzo-*p*-dioxin
TEQ – toxic equivalents
TRV – toxicity reference value
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT and 4,4′-DDT)
USEPA – US Environmental Protection Agency

Risk estimates for regulated metals were evaluated in the uncertainty section (Section 7.1.4.2), and fish tissue EFs for copper were ≥ 1.0 for four species using TRVs from the revised FFS (Louis Berger et al. 2014) (Table 7-15). There is high uncertainty associated with the risk estimates for inorganic metals due to the varying ways fish uptake, bioaccumulate, and regulate metals within tissues (Section 6.3.3.1).

Table 7-15. Summary of fish tissue LOAEL EFs for regulated metals

	Range of LOAEL HQs[a]				
	HQ Based on TRV-A[c]		HQ Based on TRV-B[d]		
COPEC[b]	Fish Species with EFs ≥ 1.0	LOAEL EFs Values ≥ 1.0	Fish Species with EFs ≥ 1.0	LOAEL EF Values ≥ 1.0	Key Uncertainties
Copper	nc	nc	EFs ≥ 1.0 for mummichog, other forage fish, white perch, and American eel	**1.7–9.3**	• Tissue-residue approach not recommended for regulated metals[e] • TRV-B less than range of nutritionally optimal threshold concentrations

Bold identified HQs ≥ 1.

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

b Only regulated metals with EFs ≥ 1.0 based on a LOAEL TRV are included in the table.

c TRVs were derived from the primary literature review based on process identified in Section 7.1.3.1.

d TRVs based on USEPA's revised draft of the LPR restoration project FFS (Louis Berger et al. 2014) or first draft of the LPR restoration project FFS (Malcolm Pirnie 2007b).

e USEPA framework for metals risk assessment (USEPA 2007e) recommends against the use of a tissue residue approach, stating that the CBR approach for metals "does not appear to be a robust indicator of toxic dose.

CBR – critical body residue
COPEC – chemical of potential ecological concern
EF – exceedance factor
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
LPR – Lower Passaic River
nc – not calculated
TRV – toxicity reference value
USEPA – US Environmental Protection Agency

7.2 Dietary Assessment

The dietary assessment was conducted for the following species from each of the three major feeding guilds: mummichog/other forage fish and common carp (benthic omnivores); white perch, channel catfish, white catfish, and white sucker (invertivores); and American eel, largemouth bass, northern pike, and smallmouth bass (piscivores). For each species, the assessment was conducted for the COPECs identified in the SLERA (Appendix A; Section 5).

This section summarizes the COPECs, describes how exposure and effects concentrations were derived, presents the HQs, and summarizes the uncertainties associated with the dietary assessment.

7.2.1 COPECs

The COPECs for each fish species were identified using a risk-based screening process in the SLERA, wherein doses calculated using maximum concentrations were compared to dietary screening-level TRVs (Appendix A). These are summarized in Section 5. Metals (cadmium, chromium, cobalt, copper, methylmercury/mercury, nickel, selenium, vanadium, and zinc), TBT, benzo(a)pyrene, total PAHs, total PCBs, TEQ (PCB TEQ - fish, PCDD/PDF TEQ - fish, total TEQ - fish), and total DDx were identified as COPECs for fish diet (Table 7-16). A number of COIs could not be screened as part of the SLERA (Appendix A) because no dietary screening levels were available. These COIs are presented in Section 5.5.2, as are the implications of not being able to evaluate these COIs.

Table 7-16. Fish dietary COPECs

COPEC	
Metals	
Cadmium	Nickel
Chromium	Selenium
Cobalt	Vanadium
Copper	Zinc
Methylmercury/mercury	
Butyltins	
TBT	
PAHs	
Benzo(a)pyrene	Total PAHs
PCBs	
Total PCBs	PCB TEQ - Fish
PCDD/PCDFs	
PCDD/PCDF TEQ - Fish	Total TEQ - Fish
Organochlorine Pesticides	
Total DDx	

Note: COPEC based on SLERA NOAEL HQ ≥ 1.0.

COPEC – chemical of potential ecological concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
HQ – hazard quotient
NOAEL – no-observed-adverse-effect level
PCDD – polychlorinated dibenzo-*p*-dioxin
PCDF – polychlorinated dibenzo-*p*-furan
TEQ – toxic equivalency
SLERA – screening-level ecological risk assessment
TBT – tributyltin

PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl

total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT and 4,4′-DDT)

7.2.2 Exposure

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

7.2.2.1 Methods

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

$$\text{Dose} = \frac{\left[(\text{FIR} \times \text{EPC}_{\text{prey}}) + (\text{SIR} \times \text{EPC}_{\text{sed}})\right]}{\text{BW}} \times \text{SUF}$$ **Equation 7-2**

Where:

Dose	=	daily ingested dose (mg/kg bw/day)
FIR	=	food ingestion rate (kg ww/day)
EPC_{prey}	=	exposure point concentration in prey tissue (mg/kg ww)
SIR	=	incidental sediment ingestion rate (kg dw/day)
EPC_{sed}	=	exposure point concentration in sediment (mg/kg dw)
BW	=	body weight (kg)
SUF	=	site use factor (unitless); proportion of time selected species spends foraging in the LPRSA

The body weights and ingestion rates were obtained from the literature for each species and are described in Section 7.2.2.2. The EPC in prey for each species was calculated from the fraction of the prey type in the species' diet and the chemical exposure concentration in that prey type, as follows:

$$EPC_{prey} = (EPC_1 \times F_1) + (EPC_2 \times F_2) + (EPC_3 \times F_3)$$ **Equation 7-3**

Where:

EPC_{prey}	=	exposure point concentration in prey items (mg COPEC/kg food dw)
$EPC_{1,2,3}$	=	exposure point concentration in each individual prey type (mg COPEC/kg tissue dw)
$F_{1,2,3}$	=	fraction ingested of each individual prey type (kg fish/kg food)

The dietary fraction (DF) of each component in each species' diet was based on information from the literature. The DFs assumed for each species and the assumptions used to derive them are described in detail in Section 7.2.2.3.

7.2.2.2 Body weights, ingestion rates, and site use factor

The exposure parameters used in the dietary dose calculations (i.e., body weights, ingestion rates, and site use factor [SUF]) are presented in Table 7-17, and were selected as follows:

- The body weight for each species was based on an average of all body weights for individuals of that particular species collected during 2009/2010 fish sampling events.
- For mummichog/other forage fish, food ingestion rates (FIRs) were based on the measured ingestion rate for mummichog. For the other species, FIRs were estimated as a function of body weight and temperature using an equation from Arnot and Gobas (2004).
- Incidental sediment ingestion rates (SIRs) were expressed as a percentage of the dry weight FIR. For American eel, the percentage of incidentally ingested sediment was based on species-specific data from the literature (Wenner and Musick 1975). For the other species, information on feeding habits and best professional judgment was used to estimate incidental SIRs.
- Dry weight FIRs (required for deriving incidental SIRs) were derived from wet weight ingestion rates assuming 80% moisture in prey (based on average moisture contents of 72, 79, and 88% for fish, invertebrates, and worms from the LPRSA, respectively).
- A SUF of 1 was used for all species, based on the assumption that they use 100% of their preferential foraging (exposure) areas. Some fish species (e.g., American eel and white perch) forage outside of the LPRSA, and therefore use the LPRSA as their exposure area less than 100% of the time. The use of an SUF of 1 provides conservative estimates of the potential risks in these cases; the effect on the HQs of using SUFs < 1 is discussed in Section 7.2.4.2.

Two general size classes were evaluated for American eel (i.e., American eel ≥ 50 cm in length and American eel < 50 cm in length), because of differences in their diet, as discussed in Section 7.2.2.3.

Table 7-17. Exposure parameter values for fish species

Species	BW (kg)[a]	Food Ingestion		Incidental Sediment Ingestion		
		FIR (kg ww/day)	Source	SI (%)[b]	SIR (kg dw/day)[c]	Source
Mummichog/ other forage fish	0.0032	0.00019	Weisberg and Lotrich (1982)[d]	10%	0.0000037	no empirical data available; based on feeding habits and best professional judgment
Common carp	2.7	0.11	Arnot and Gobas (2004)[f]	15%	0.0033	no empirical data available; based on feeding habits and best professional judgment

Species	BW (kg)[a]	Food Ingestion		Incidental Sediment Ingestion		
		FIR (kg ww/day)	Source	SI (%)[b]	SIR (kg dw/day)[c]	Source
White sucker	0.79	0.039	Arnot and Gobas (2004)[f]	10%	0.00077	no empirical data available; based on feeding habits and best professional judgment
White perch	0.057[e]	0.0041	Arnot and Gobas (2004)[f]	5%	0.000041	no empirical data available; based on feeding habits and best professional judgment
Channel catfish	0.74	0.036	Arnot and Gobas (2004)[f]	10%	0.00073	no empirical data available; based on feeding habits and best professional judgment
White catfish	0.75	0.037	Arnot and Gobas (2004)[f]	10%	0.00074	no empirical data available; based on feeding habits and best professional judgment
American eel < 50 cm	0.032	0.0025	Arnot and Gobas (2004)[f]	5%	0.000025	Wenner and Musick (1975)
American eel ≥ 50 cm	0.45	0.024	Arnot and Gobas (2004)[f]	5%	0.00024	Wenner and Musick (1975)
Largemouth bass	0.078	0.0054	Arnot and Gobas (2004)[f]	1%	0.000011	no empirical data available; based on feeding habits and best professional judgment
Smallmouth bass	0.14	0.0089	Arnot and Gobas (2004)[f]	1%	0.000018	no empirical data available; based on feeding habits and best professional judgment
Northern pike	2.8	0.11	Arnot and Gobas (2004)[f]	1%	0.00023	no empirical data available; based on feeding habits and best professional judgment

a Body weight for each species was based on average of all body weights for that species collected during 2009/2010 fish sampling events.

b Based on percentage of the dry weight FIR that is incidentally ingested sediment.

c Wet weight FIR converted to dry weight FIR assuming 80% moisture in prey (based on average moisture contents of 72, 79, and 88% for fish, invertebrates, and worms, respectively) to determine SIR in kg/day.

d FIR is 0.0582 g dw/g dw bw/day; wet weight FIR assumes the same moisture content in mummichog tissue and in invertebrate prey (so FIR = 0.0582 g ww/g ww bw/day).

e Average body weight excludes small white perch (n = 452) collected during the late summer SFF sampling effort in 2010 (average weight was 0.004 g).

f FIR (kg ww/day) is a function of body weight based on the following equation: FIR = $(0.022 \times BW^{0.85}) \times \exp^{(0.06 \times T)}$, where T = 12.7ºC based on monthly average temperatures in the LPRSA between 1998 and 2009 (USEPA 2009) and body weight is in kg.

BW or bw – body weight
dw – dry weight
FIR – food ingestion rate
LPRSA – Lower Passaic River Study Area
SFF – small forage fish
SI – sediment ingestion
SIR – sediment ingestion rate
USEPA – US Environmental Protection Agency
ww – wet weight

7.2.2.3 Prey composition and exposure areas

For the dietary dose equation (Equation 7-2), prey ingested by fish species were limited to only those prey types for which empirical tissue chemistry data from the LPRSA were available. These tissue data include freshwater and estuarine worms (from the bioaccumulation study), blue crab, and fish. While fish and blue crab data were field collected, the worm tissue data were based on the laboratory bioaccumulation study in which worms were exposed to homogenized sediment collected from the 0- to 15-cm depth horizon. The available tissue data do not include other prey items that may be important components of the fish diets, such as amphipods, algae, zooplankton, or detritus; therefore, the representativeness of the dietary estimates for fish (based on available prey tissue data) of actual LPRSA fish diets is uncertain.

The proportions of worms, blue crab, and fish in the diets of fish species, as well as the size of fish each species would most likely feed upon, were based on a review of the literature as presented in Table 7-18. The rationale for the selection of these prey portions and sizes is presented in more detail for each species later in this section. A best estimate portion for each potential prey item (worms, blue crab, or fish) was selected; however, there is uncertainty in the assigned percentages, given the opportunistic feeding behavior of most fish species. In reality, it is likely that fish diets vary considerably depending on prey availability, and thus the season and specific location of a given fish may result in a significantly different diet.

Table 7-18. Prey composition used to estimate dietary dose for fish species

Species	Percentage of Prey Type in Diet[a]				
	Worm[b]	Blue Crab	Fish		
			≤ 11 cm[c]	≤ 13 cm[d]	≤ 20 cm[e]
Mummichog/other forage fish	100	0	0	0	0
Common carp	82	17	1	0	0
White sucker	90	10	0	0	0
White perch	70	15	15	0	0
Channel catfish	55	5	0	40	0
White catfish	55	5	0	40	0
American eel < 50 cm	80	10	0	10	0
American eel ≥ 50 cm	35	25	0	0	40
Largemouth bass	10	10	0	80	0
Smallmouth bass	10	10	0	80	0
Northern pike	0	10	0	0	90

a Fish diet is evaluated in both this BERA and the LPRSA FS bioaccumulation model (Windward 2015b) and estimated using data from the literature and regional studies. The fish diet information presented in this BERA and in the LPRSA FS bioaccumulation model differ: while this BERA relied on dietary items for which empirical LPRSA tissue concentrations were available, the FS bioaccumulation model used estimated tissue concentration ranges for a larger modeled prey base.

b Includes both freshwater and estuarine worms.

[c] Fish ≤ 11 cm long included the following species: gizzard shad, mummichog, silver shiner, spottail shiner, and white perch. For composite samples, length is based on the maximum of any fish in the sample.

[d] Fish ≤ 13 cm long included the following species: gizzard shad, mixed forage fish, mummichog, silver shiner, spottail shiner, and white perch. For composite samples, length is based on the maximum of any fish in the sample.

[e] Fish ≤ 20 cm long included the following species: gizzard shad, mixed forage fish, mummichog, pumpkinseed, silver shiner, spottail shiner, and white perch. For composite samples, length is based on the maximum of any fish in the sample.

BERA – baseline ecological risk assessment

FS – feasibility study

LPRSA – Lower Passaic River Study Area

The areas for the exposure of different fish species to sediment and prey are presented in Table 7-19, and the rationale for the selection of these exposure areas is presented in more detail for each species in the remainder of this section.

Table 7-19. LPRSA exposure areas for fish species

Species	Exposure Area	
	Prey	Sediment
Mummichog/other forage fish	site wide	site-wide mudflat areas[a]
Common carp	> RM 4 for worms and fish ≤ 11 cm; site wide for blue crabs	> RM 4
White sucker	> RM 4 for worms; site wide for blue crabs	> RM 4
White perch	site wide	site wide
Channel catfish	> RM 4 for worms and fish ≤ 13 cm; site wide for blue crabs	> RM 4
White catfish	site wide	site wide
American eel < 50 cm	site wide	site wide
American eel ≥ 50 cm	site wide	site wide
Largemouth bass	> RM 4 for worms and fish ≤ 13 cm; site wide for blue crabs	> RM 4
Smallmouth bass	> RM 4 for worms and fish ≤ 13 cm; site wide for blue crabs	> RM 4
Northern pike	> RM 4 for fish ≤ 20 cm; site wide for blue crabs	> RM 4

[a] Mudflats are defined as areas within -2 ft MLLW and < 6º slope and include all grain sizes.

LPRSA – Lower Passaic River Study Area

MLLW – mean lower low water

RM – river mile

In accordance with the PFD (Windward and AECOM 2009), stomach contents of selected fish species collected in 2009 and 2010 were examined. Fish stomach content prey taxonomy information samples could not be collected because of a lack of sufficient specimens for analysis. Of the 119 fish from the LPRSA that underwent the pathology evaluation in 2009 (n = 83) and 2010 (n = 36), 32% of the fish stomachs (n = 38) were observed to be empty, 53% (n = 63) were observed to contain food (mostly unidentifiable digested material), and 3% (n = 4) were observed to contain digestive fluid or mucous only.[102] In fish caught in the LPRSA, discernible stomach contents included amphipods in one white perch; an unidentifiable whole fish and other fish remains in one striped bass; and an unidentifiable whole fish in one redfin pickerel (*Esox americanus*). Of the 46 fish from the freshwater background area above Dundee Dam that underwent the pathology evaluation, 24% of the fish stomachs (n = 11) were observed to be empty, 59% (n = 27) were observed to contain food (mostly unidentifiable digested material), and 17% (n = 8) were observed to contain digestive fluid or mucous only. The only discernible stomach content in fish caught above Dundee Dam was a nematode worm in one pumpkinseed. Because the taxonomic prey composition could not be determined in the selected LPRSA fish examined, regional and general literature data were reviewed and used to determine the dietary composition of selected fish species in the evaluation of dietary exposure.

Mummichog/ Other Forage Fish

The dietary assumptions for mummichog/other forage fish are based on use of surrogate species for mummichog dietary habits.

Prey Composition

Several studies conducted in varied habitats have found that the mummichog diet consists of detritus, algae, small crustaceans (i.e., amphipods, tanaids, copepods, and ostracods), insects (adult and larvae), and polychaetes (Abraham 1985; Allen et al. 1994; James-Pirri et al. 2001; Kneib 1986; Currin et al. 2003). Since estuarine and freshwater worm (*Nereis virens* and *Lumbriculus variegates,* respectively) data were available from the LPRSA, these data were used to represent the benthic prey portion in the mummichog diet (i.e., 100% of the mummichog diet was based on worm tissue). Both freshwater and estuarine worm data are included in the mummichog diet because mummichog were found in most LPRSA river reaches.

Exposure Area

Mummichog tend to inhabit bays and tidally influenced rivers and creeks or estuaries, prefer shallow water near the shoreline, and typically do not go deeper than 3.7 m (12 ft) (Bigelow and Schroeder 1953). They are usually found within 110 m of shorelines along intertidal marshes and mudflats (Armstrong and Child 1965 as cited in Abraham 1985; Hardy 1978 as cited in Abraham 1985; Lotrich 1975). Mummichog and other SFF were collected throughout the LPRSA during 2009 and 2010 surveys/sampling (Windward 2018b, c). During field efforts conducted in the spring and summer of 2010 (Windward 2011c), it was observed that mummichog prefer shallow water habitats and

[102] There were no stomach contents observations recorded for 12% (n = 14) of the assessed fish.

mudflats, often with overhanging or shoreline vegetation. Based on this information, the exposure area selected for the risk calculations for mummichog/other forage fish includes only mudflat areas throughout the LPRSA.

Common Carp

Prey Composition

Common carp are omnivores and are considered highly opportunistic feeders with a variable diet, the majority of which is composed of detritus, algae/plants, and small benthic invertebrates, as well as insects, small fish, and zooplankton (Maryland DNR 2007a; Garcia-Berthou 2001; USGS 2010; Walburg and Nelson 1966). Carp are mainly bottom dwellers that feed by rooting in the bottom substrate with their snouts and eating the food they dislodge, along with fine sediment and detritus (Pennsylvania FBC 2011). Algae, detritus, pebbles, and sediment are commonly found in the stomach contents of common carp (Campos 2005). Common carp have also been reported to prey on the eggs of other fish species, decayed aquatic plants, and the stalks, leaves, and seeds of aquatic and terrestrial plants (USGS 2010). Quantitative ranges of prey portions vary widely for carp based on their opportunistic nature and the availability of prey. Prey portions of common carp for a Colorado river were as follows: 24 to 56% detritus, 22 to 60% plants and benthic algae, 0 to 2% zooplankton, 4 to 11% insects, 2 to 44% benthic invertebrates, and 0 to 2% fish (FishBase 2014). Based on the prey types available from the LPRSA for dietary modeling, the carp diet was modeled using worms, blue crabs, and fish. Only a small portion of their diet (1%) was assumed to be fish. Considering their method of feeding and using best professional judgement, carp were assumed to prey on only small fish ≤ 11 cm in length.[103] The remainder of the carp diet was represented with worms (82%) and blue crab (17%). Worms were used as a surrogate to represent zooplankton and insects and also accounted for the detritus and plants in their diet. Blue crab were used as a surrogate to represent macroinvertebrates such as crayfish. Diet composition for carp was uncertain given the large portion of dietary items with no empirical chemistry data from the LPRSA (e.g., detritus and plants).

Exposure Area

Carp are hardy and tolerant of a wide variety of conditions, and generally favor large bodies of slow-flowing or standing water and soft bottom sediments (Maryland DNR 2007a). However, carp can be found in waters that have any type of substrate, including mud, sand, or gravel (Pennsylvania FBC 2011). Common carp generally inhabit lakes, ponds, and the lower sections of rivers, but are also found in brackish-water estuaries, backwaters, bays, and the saline coastal waters of several states bordering the Atlantic and Pacific Oceans and the Gulf of Mexico (USGS 2010). In the LPRSA, common carp were collected from areas between RM 4 and Dundee Dam. Based on the results of the

[103] Fish ≤ 11 cm include the following species: gizzard shad, mummichog, silver shiner, spottail shiner, and white perch. For composite samples, length is based on the maximum of any fish in the sample.

LPRSA sampling, the exposure area selected for the risk calculations for carp was limited to the LPRSA above RM 4.

White Sucker

Prey Composition

White sucker are freshwater fish found in lacustrine and riverine environments (Twomey et al. 1984). Adults prey on benthic invertebrates (e.g., amphipods and gastropods) and insects. Prey portions of adult white sucker reported for Canada's Salt River were as follows: 29 to 42% detritus, 59 to 66% insects, 4 to 9% gastropods, 0 to 4% fish, and 0 to 3% ostracods (FishBase 2017). Based on the prey types available from the LPRSA for dietary modeling, the white sucker diet was modeled using worms and blue crabs. Fish were assumed to make up a negligible portion of the white sucker diet. The white sucker diet was represented primarily with worms (90% of their diet), which were used as a surrogate for insects and also accounted for the detritus portion in their diet. Blue crab (used as a surrogate to represent macroinvertebrates such as gastropods) were assumed to make up 10% of the white sucker diet. The diet composition for white sucker was uncertain given the large portion of dietary items with no empirical chemistry data from the LPRSA (e.g., detritus).

Exposure Area

White sucker are found in freshwater and brackish waters. White sucker collected during the 2009 and 2010 LPRSA sampling events were limited to freshwater portions of the LPRSA between RM 6 and Dundee Dam. Therefore, the exposure area selected for the risk calculations for white sucker was the lower-salinity and freshwater reaches of the LPRSA, from RM 4 to RM 17.4.

White Perch

Prey Composition

As an invertivorous benthic-feeding fish, the white perch's common dietary components include amphipods, shrimp, and copepods, based on regional studies for the Hudson and Hackensack Rivers (Bath and O'Connor 1985; Weis 2005). White perch diets vary depending on the time of year and the maturity of the individual fish. A greater proportion of the white perch's diet is fish in late summer and fall, while a greater proportion is invertebrates in winter and spring (Bath and O'Connor 1985; Weis 2005). An analysis of white perch stomach contents from fish caught in Lake Erie found high variability in white perch diets, which ranged from 0 to 96% cladocerans, 0 to 30% copepods, 0 to 14% chironomids, 0 to 77% gizzard shad, and 1 to 15% unidentified fish, depending on the season (Schaeffer and Margraf 1986).

In the Hudson River, New York, Bath and O'Connor (1985) identified amphipods as the most common food item in mature and immature white perch gut contents from the oligohaline zone from May through November, although the species' diet was quite variable. The average frequency of occurrence of the various prey items in white perch gut contents was as follows: 37.3% amphipods, 6.5% isopods, 5.2% insects, 4.4% annelids, 3.1% shrimp, 2.1% fish eggs, 1.8% fish, 1.2% cladocerans, 0.5% fish larvae, 8.5% plant matter, and 29.7% unidentified material. In a year-round study of the Hackensack River, New Jersey, the predominant prey items of white perch were amphipods, while fish and shrimp were less important items (Weis 2005). While prey contents varied by season, the yearly average dry weight percentages were as follows: 23% amphipods, 17% shrimp, 17% fish, < 1% plant matter, and 43% unidentified material (Weis 2005).

The data from the Hackensack River as reported by Weis (2005), based on a year-round study in the area most regionally applicable to the LPRSA, were used to estimate general prey portions used in risk calculations (Table 7-18). Blue crab (representing a surrogate prey species for shrimp) and small fish were assumed to each make up approximately 15% of the white perch diet based on the data from the Hackensack River. Amphipods and unidentified material made up the remaining white perch stomach content in the Hackensack data; LPRSA worms were used to represent this remaining portion (70%) of the white perch diet (the unidentified material observed in the white perch's diet was apportioned to the worm category). A high proportion of non-decapod invertebrates in the white perch diet was also reported in the Hudson River data (Bath and O'Connor 1985; Weis 2005). For the small fish prey portion, it was assumed that white perch would only consume fish < 11 cm[104] in length, because it was unlikely that a white perch caught in the LPRSA (the maximum specimen length in any one sample was 32 cm) could consume fish larger than about one-third its own size.

Exposure Area

White perch were collected throughout the LPRSA during 2009 and 2010 surveys/sampling (Windward 2018b, c). Habitat information from the literature suggests that white perch is an adaptable species that migrates between lower- and higher-salinity areas of rivers and estuaries, and that their preference for lower- or higher-salinity areas can vary depending on life stage (Stanley and Danie 1983). Based on this information, the exposure area selected for the risk calculations for white perch included all areas of the LPRSA.

Channel Catfish

Prey Composition

Channel catfish are opportunistic, omnivorous bottom feeders and have a variable diet that includes SFF, terrestrial and aquatic insects, detritus, plant material, crayfish, and mollusks (Fewlass 1980; Holtan 1998a; McMahon and Terrell 1982). Channel catfish tend to forage along the bottoms of water bodies (Pennsylvania FBC 2011).

[104] Fish ≤ 11 cm include the following species: gizzard shad, mummichog, silver shiner, spottail shiner, and white perch. For composite samples, length is based on the maximum of any fish in the sample.

Juvenile channel catfish feed primarily on insects, insect larvae, and small aquatic zooplankton (Wellborn 1988; Holtan 1998a; McMahon and Terrell 1982). In a study of juvenile channel catfish from the Lower Susquehanna River, Maryland, that were not much longer than 20 cm, caddis fly larvae were found to account for 40 to 60% of their diet, and midge larvae for 25 to 55% of their diet (Weisberg and Janicki 1990). As channel catfish grow, they begin to feed on snails, crayfish, and small fish, but still eat aquatic insects and occasionally plant matter (Holtan 1998a). Adult channel catfish feed primarily on fish and plant matter, and secondarily on insects and benthic invertebrates (Fewlass 1980). Adult catfish feed predominantly on fish, whereas juvenile catfish feed primarily on insects, insect larvae, and zooplankton (Wellborn 1988; Holtan 1998a; McMahon and Terrell 1982). The following gravimetric percentages were documented in channel catfish from the Susquehanna River in Maryland (Fewlass 1980): 43% fish, 1.6% mollusks (Pelecypoda), 3.2% insects, 2.2% crustacean (primarily *Callinectes sapidus*), 45% plants (generally intermingled with invertebrates), and 5.1% inorganic content (primarily small stones that were part of *Trichoptera* cases). When plants were present in channel catfish stomach contents, they were usually intermingled with invertebrates, suggesting incidental ingestion while catfish were feeding on invertebrates.

The 11 channel catfish caught in the LPRSA and analyzed for tissue chemistry ranged in length from 35 to 51 cm, a size range representing sexually mature channel catfish (Fewlass 1980). Therefore, the general prey composition for channel catfish used in dietary risk calculations (Table 7-18) was based on the adult data from Fewlass (1980). Blue crab (representing mollusks and crustaceans) and small fish were assumed to make up approximately 5 and 40%, respectively, of the channel catfish diet. The remaining portion of the channel catfish diet was primarily plants, but also included insects. Because plant chemistry data were not available, the remaining portion (55%) of the channel catfish diet was represented by LPRSA worms. Taking into account the size of catfish in the LPRSA, best professional judgment was used to assume that channel catfish would not consume fish > 13 cm in length.[105] The diet composition for channel catfish was uncertain given the large portion of dietary items with no empirical chemistry data from the LPRSA (e.g., plants).

Exposure Area

Channel catfish were collected from Reaches 5 through 8 (RM 8 through RM 17.4) of the LPRSA during 2009 and 2010 surveys, and more than half of those collected were collected from Reach 8 (RM 14 through RM 17.4) (Windward 2018b, c). Habitat information from the literature suggests that channel catfish prefer clear, slow-moving waters, seeking out deep pools for shelter during the day (Holtan 1998a). In general,

[105] Fish ≤ 13 cm long included the following species: gizzard shad, mixed forage fish, mummichog, silver shiner, spottail shiner, and white perch. For composite samples, length was based on the maximum of any fish in the sample.

channel catfish prefer salinities of < 8 ppth; growth effects may occur at greater salinities (McMahon and Terrell 1982). Based on this information—and assuming the potential for channel catfish to occur in lower reaches than those observed during surveys based on seasonal shifts in salinity in the LPRSA—the exposure area selected for the risk calculations for channel catfish included the LPRSA above RM 4.

White Catfish

Prey Composition

White catfish are bottom-feeding invertivorous fish widely distributed throughout New Jersey (Pennsylvania FBC 2011). As juveniles, white catfish feed on amphipods, shrimp, and insect larvae, as well as larger invertebrates and some fish. White catfish fish can be somewhat opportunistic and swim within the water column to feed of plantivorous fish; however, the majority of their diet is composed of benthic invertebrates and a small fraction of fish (California Fish Website 2013; Turner 1966). DFs for juveniles and adult white catfish from California rivers were reported as follows: 41% benthic invertebrates, 41% small fish, 6% birds/mammals, 2% bryozoans, 2% insects, and 9% other (FishBase 2014). Only a limited portion of these dietary items were available from the LPRSA for dietary modeling; the dietary composition for LPRSA white catfish was based on the following prey items: worms, blue crabs and small fish. The same portions were assigned to white catfish and channel catfish: 5% blue crab (representing benthic macroinvertebrates such as crayfish and shrimp), 40% small fish, and 55% worms (surrogate for benthic invertebrates and insects). Similar to the assumption made for channel catfish, white catfish were assumed to not consume fish > 13 cm in length.[106]

Exposure Area

White catfish are native to coastal Atlantic waters, inhabiting freshwater and brackish habitats along the Gulf of Mexico coast and the Atlantic coast from New York to Florida (Maryland DNR 2007b). Of all the catfish species, white catfish are the most tolerant of salt water (Pennsylvania FBC 2011). They can live in lakes and reservoirs, as well as brackish bays and estuaries. White catfish were collected from the LPRSA in Reaches 2 through 8 (RM 2 to RM 17.4) during the 2009 and 2010 sampling events. Their presence in the LPRSA supported information in the literature (Pennsylvania FBC 2011), which noted the white catfish's ability to survive in waters with higher salinities. Therefore, the exposure area selected for the risk calculations for white catfish included all areas of the LPRSA.

[106] Fish ≤ 13 cm long included the following species: gizzard shad, mixed forage fish, mummichog, silver shiner, spottail shiner, and white perch. For composite samples, length was based on the maximum of any fish in the sample.

American Eel

Prey Composition

American eel are opportunistic carnivores and have a diverse diet that includes annelids, polychaetes, insect larvae and nymphs, crustaceans, bivalves, gastropods, fish, frogs, and small mammal remains (Facey and Van Den Avyle 1987; Morrison 2001; Gray 1992; ASMFC 2000; Denoncourt and Stauffer 1993). American eel tend to feed near the water's bottom and will scavenge dead organisms (Facey and Van Den Avyle 1987), although not as a substantial portion of their diet (Denoncourt and Stauffer 1993). Eel larvae likely feed on plankton when living in a marine environment (Gray 1992). Juvenile life stage American eel (i.e., glass, elver, and yellow eel) consume fish and invertebrates (NJDEP 2001a).

Prey size tends to increase as eel size increases (Ogden 1970) and both small and large American eel were found in the LPRSA (Table 2-4). As eel grow, a clear shift occurs in the percent composition of prey in their diet; larger eel have a diet of mainly fish and crustaceans, while smaller eel mainly prey on insects (Ogden 1970). Accordingly, dietary risks were evaluated separately for small eel (< 50 cm in length) and large eel (≥ 50 cm length).

In a study of gut contents of American eel from New Jersey streams, Ogden (1970) found that eel < 50 cm in length consumed primarily insects (72–100%) and secondarily fish (0–22%) and crustaceans (0–19%), whereas American eel ≥ 50 cm in length consumed more fish (20–60%) and crustaceans (20–40%) and fewer insects (0–40%). Similarly, a study of prey items in the diet of American eel from the James River (a tributary to Chesapeake Bay) found that for eel < 25 cm in length, invertebrates comprised 95% of the diet, while crayfish comprised 5% (Lookabaugh and Angermeier 1992). The same study found that American eel ≥ 37 cm in length consumed < 5% invertebrates and > 95% crayfish and vertebrates.

Percentages of worms, crabs, and fishes in the diet of American eel in the LPRSA (Table 7-18) were estimated roughly based on data from New Jersey streams (Ogden 1970). Small eel (< 50 cm) were assumed to consume approximately 80% worms (surrogate for insect species), 10% blue crabs (surrogate for crustaceans), and 10% fish. Larger eel (≥ 50 cm) were assumed to consume approximately 35% worms (surrogate for insect species), 25% blue crab (surrogate for crustaceans) and 40% fish. Best professional judgment was used to select the maximum sizes of fish in the diet of American eel: 13 cm for eel < 50 cm in length and 20 cm for eel ≥ 50 cm in length.[107]

[107] Fish ≤ 13 cm long included the following species: gizzard shad, mixed forage fish, mummichog, silver shiner, spottail shiner, and white perch. Fish ≤ 20 cm long included the following species: gizzard shad, mixed forage fish, mummichog, pumpkinseed, silver shiner, spottail shiner, and white perch. For composite samples, length is based on the maximum of any fish in the sample.

Exposure Area

American eel were collected throughout the LPRSA during 2009 and 2010 surveys/sampling (Windward 2018b, c). Habitat information from the literature suggests that American eel are an adaptable species that migrates between lower- and higher-salinity areas of rivers and estuaries; migration patterns vary depending on the life stage (ASMFC 2000; Facey and Van Den Avyle 1987). Based on this information, the exposure area selected for the risk calculations for American eel includes all areas of the LPRSA.

Largemouth Bass

Prey Composition

Adult largemouth bass are predominately piscivorous and eat fish such as bluegills, minnows, perch, shiners, smelt, sculpin, suckers, and smaller centrachids (Scott and Crossman 1973). However, they are opportunistic and will also eat crayfish, frogs, insects, snakes, and even small mammals and birds that enter the water (Scott and Crossman 1973; FishBase 2007).

Largemouth bass > 5 cm in total length feed almost exclusively on other fish (Scott and Crossman 1973; TAMS and Menzie-Cura 2000). Data from a Hudson River, New York, study indicate that 75 to 90% of the largemouth bass diet consists of fish, and 10 to 25% consists of various invertebrates, including crayfish (TAMS and Menzie-Cura 2000). The invertebrates most commonly observed in the gut contents of largemouth bass include amphipods, isopods, cladocerans, cyclopoid copepods, ostracods, and some chironomid larvae (TAMS and Menzie-Cura 2000). Largemouth bass prey composition assumptions used in the risk calculations (Table 7-18) were estimated from the Hudson River data (TAMS and Menzie-Cura 2000): 10% worms (surrogate for insect species), 10% blue crab (surrogate for crustaceans), and 80% fish. Best professional judgment was used to assume that largemouth bass would not consume fish > 13 cm (approximately) in length.

Exposure Area

Largemouth bass were collected only in areas above RM 6 during the 2009 and 2010 surveys (Windward 2018b, c). Largemouth bass are typically a freshwater species and are usually found only in water with a salinity of < 4 ppth (Stuber et al. 1982b). They prefer abundant aquatic vegetation and overgrown banks (Curtis and Wehrly 2006; Page and Burr 1991), which tend to be found only in the upper reaches of the LPRSA, and have strong site fidelity and small home ranges, rarely larger than 100 m in length (Gatz and Adams 1994). Therefore, the exposure area selected for the risk calculations for largemouth bass included only the lower salinity and freshwater reaches of the LPRSA, from RM 4 to RM 17.4.

Smallmouth Bass

Prey Composition

Similar to largemouth bass, smallmouth bass adults are piscivorous fish considered top predators within their ecosystems. Common dietary items for adult smallmouth bass from Pennsylvania, Minnesota, and California rivers include the following: detritus (0 to 6%), insects (1 to 92%), decapods (2 to 21%), other benthic invertebrates (including crustaceans and oligochaetes) (0 to 9%), and fish (0 to 78%) (FishBase 2014). A study of fish in Lake Sammamish, Washington, reported ranges of prey found in smallmouth bass stomachs over the course of 5 years as follows: 0 to 19% aquatic insects, 15 to 42% crayfish, and 50 to 71% fish (Pflug and Pauley 1984). In the Willamette River in Portland, Oregon, smallmouth bass stomach contents contained the following prey (by wet weight): 90% fish, 5% crayfish, and 5% shrimp (Pribyl et al. 2005). Based on these studies, smallmouth bass were assigned the same prey portions as largemouth bass: 10% worms (surrogate for small benthic invertebrates and insects), 10% blue crab (surrogate for crayfish and other decapods) and 80% fish. Best professional judgment was used to assume that largemouth bass would not consume fish > 13 cm (approximately) in length.

Exposure Area

Smallmouth bass inhabit and prefer primarily freshwater environments. Smallmouth bass prefer rocky locations with more limited vegetation, deeper water, and faster currents than largemouth bass (Pflug and Pauley 1984; NJDEP 2011a). Smallmouth bass were collected only in areas above RM 6 during the 2009 and 2010 surveys (Windward 2018b, c). The exposure area selected for the risk calculations for smallmouth bass included only the lower salinity and freshwater reaches of the LPRSA, from RM 4 to RM 17.4.

Northern Pike

Prey Composition

Northern pike are large, piscivorous fish (FishBase 2007; Inskip 1982). Adult northern pike are large, aggressive predators; common dietary items for northern pike within Alberta and Ohio include fish (e.g., bass, bluegill, shad, and silversides), which make up 91 to 95% of their diet (FishBase 2014). Benthic invertebrates like crayfish and other crustaceans have also been found in the diet of northern pike, comprising 0 to 9% of their diet (FishBase 2014). The LPRSA northern pike diet was assumed to be comprised of 90% fish and 10% blue crabs (representing crayfish and other crustaceans). Best professional judgment was used to assume that northern pike could consume larger fish up to 20 cm (approximately) in length.

Exposure Area.

Northern pike are found in freshwater and brackish water. Few northern pike were caught during the 2009 and 2010 LPRSA sampling events; the two individuals caught were from freshwater portions of the LPRSA between RM 8 and RM 12. The exposure area selected for the risk calculations for northern pike included only the lower-salinity and freshwater reaches of the LPRSA, from RM 4 to RM 17.4.

7.2.2.4 Exposure point concentrations

EPCs were calculated for each of the two modeled media types (prey and sediment) to calculate dietary doses using Equation 7-2 (Section 7.2.2.1). For prey concentrations, EPCs were calculated separately for each of the prey types for each species (worms, blue crab, fish ≤ 11 cm, fish ≤ 13 cm, and fish ≤ 20 cm) as the UCL for each prey group (or the maximum concentration if there were fewer than six detected values) (Table 7-20). For sediment, the EPCs were equal to the UCLs using data from the relevant exposure areas for each fish species (Table 7-19). UCL concentrations were calculated using USEPA's ProUCL® statistical package (Version 5.0.00) (USEPA 2013d), as described in Section 4.3.7.[108] For each dataset with fewer than six samples, a UCL was not calculated; instead, the maximum concentration was used as the EPC. The UCLs used to calculate EPCs are presented in Appendix C.

[108] The UCL recommended by USEPA's ProUCL® software (typically the 95% UCL, but in some cases the 97.5% or even the 99% UCL) was used. The selected UCL statistic is presented in Appendix C1.

Table 7-20. Data groups for calculation of prey and sediment EPCs for fish diet

Species and Exposure Area	Prey Type and Exposure Area			Sediment Exposure Area
	Prey Type[a,b]	% in Diet	Exposure Area	
Mummichog/other forage fish	worms	100	site-wide	site-wide mudflats
Common carp	worms	82	RM ≥ 4	RM ≥ 4
	blue crab	17	site-wide	
	fish 0–11 cm[c]	1	RM ≥ 4	
White sucker	worms	90	RM ≥ 4	RM ≥ 4
	blue crab	10	site-wide	
White perch	worms	70	site-wide	site-wide
	blue crab	15	site-wide	
	fish 0–11 cm[c]	15	site-wide	
Channel catfish	worms	55	RM ≥ 4	RM ≥ 4
	blue crab	5	site-wide	
	fish 0–13 cm[d]	40	RM ≥ 4	
White catfish	worms	55	site-wide	site-wide
	blue crab	5	site-wide	
	fish 0–13 cm[d]	40	site-wide	
American eel < 50 cm	worms	80	site-wide	site-wide
	blue crab	10	site-wide	
	fish 0–13 cm[d]	10	site-wide	
American eel ≥ 50 cm	worms	35	site-wide	site-wide
	blue crab	25	site-wide	
	fish 0–20 cm[e]	40	site-wide	
Largemouth bass	worms	10	RM ≥ 4	RM ≥ 4
	blue crab	10	site-wide	
	fish 0–13 cm[d]	80	RM ≥ 4	
Smallmouth bass	Worms	10	RM ≥ 4	RM ≥ 4
	blue crab	10	site-wide	
	fish 0–13 cm[d]	80	RM ≥ 4	
Northern pike	blue crab	10	site-wide	RM ≥ 4
	fish 0–20 cm[e]	90	RM ≥ 4	

Note: If fewer than six samples were available to calculate a UCL, the maximum concentration was used.

a As represented by whole-body tissue concentrations.

b For composite fish samples, length was based on the maximum length of any fish in the sample.

c Fish ≤ 11 cm long included the following species: gizzard shad, mummichog, silver shiner, spottail shiner, and white perch. For composite samples, length was based on the maximum of any fish in the sample.

[d] Fish ≤ 13 cm long included the following species: gizzard shad, mixed forage fish, mummichog, silver shiner, spottail shiner, and white perch. For composite samples, length was based on the maximum of any fish in the sample.

[e] Fish ≤ 20 cm long included the following species: gizzard shad, mixed forage fish, mummichog, pumpkinseed, silver shiner, spottail shiner, and white perch. For composite samples, length was based on the maximum of any fish in the sample.

EPC – exposure point concentration

RM – river mile

UCL – upper confidence limit on the mean

7.2.2.5 Estimated doses

Dietary doses were calculated based on Equation 7-2 using the prey and sediment, ingestion rates, and species body weights from Table 7-17; the prey composition from Table 7-18; and the EPCs from Appendix C. These doses are presented in Table 7-21.

Table 7-21. Dietary doses for fish

COPEC	Units	Dietary Dose										
		Benthic Omnivore		Invertivore				Piscivore				
		Mummichog/ Other Forage Fish	Common Carp	White Perch	Channel Catfish	White Sucker	White Catfish	American Eel - Small (< 50 cm)	American Eel - Large (≥ 50 cm)	Largemouth Bass	Smallmouth Bass	Northern Pike
Metals												
Cadmium	mg/kg bw/day	0.013	0.012	0.011	0.010	0.011	0.0089	0.012	0.0070	0.0051	0.0047	0.0024
Chromium	mg/kg bw/day	1.5	1.2	1.3	1.2	1.5	0.95	1.5	0.75	0.96	0.88	0.59
Cobalt	mg/kg bw/day	0.050	0.038	0.043	0.035	0.046	0.030	0.051	0.022	0.021	0.019	0.0087
Copper	mg/kg bw/day	0.56	0.60	0.71	0.47	0.59	0.45	0.71	0.63	0.44	0.40	0.30
Mercury	µg/kg bw/day	7.6	6.1	7.2	5.5	5.5	6.0	7.5	6.4	4.7	4.3	4.1
Methyl mercury	µg/kg bw/day	0.18	0.95	2.1	1.0	0.73	1.5	1.6	3.3	2.6	2.4	4.0
Nickel	mg/kg bw/day	0.81	0.65	0.75	0.71	0.84	0.53	0.90	0.44	0.68	0.62	0.36
Selenium	mg/kg bw/day	0.033	0.027	0.044	0.032	0.032	0.031	0.046	0.038	0.047	0.043	0.033
Vanadium	mg/kg bw/day	0.067	0.060	0.058	0.065	0.061	0.057	0.065	0.039	0.059	0.054	0.031
Zinc	mg/kg bw/day	2.7	2.2	2.9	2.4	2.4	2.3	3.1	2.2	2.8	2.6	1.4
Organotin												
TBT	µg/kg bw/day	0.088	0.094	0.34	0.14	0.097	0.45	0.25	0.34	0.22	0.20	0.30
PAHs												
Benzo(a)pyrene	µg/kg bw/day	31	21	26	17	24	17	32	12	6.0	5.5	1.8
Total PAHs	µg/kg bw/day	190	210	170	180	230	140	200	89	68	62	23
PCBs												
Total PCBs	µg/kg bw/day	18	15	24	23	16	21	24	49	40	36	65

Table 7-21. Dietary doses for fish

COPEC	Units	Dietary Dose										
		Benthic Omnivore		Invertivore				Piscivore				
		Mummichog/ Other Forage Fish	Common Carp	White Perch	Channel Catfish	White Sucker	White Catfish	American Eel - Small (< 50 cm)	American Eel - Large (≥ 50 cm)	Largemouth Bass	Smallmouth Bass	Northern Pike
PCB TEQ - fish	ng/kg bw/day	0.016	0.020	0.027	0.026	0.020	0.022	0.026	0.048	0.044	0.040	0.056
PCDD/PCDFs												
PCDD/PCDF TEQ - fish	ng/kg bw/day	5.5	5.4	4.7	5.1	5.1	4.2	4.9	5.1	4.0	3.7	5.4
Total TEQ - fish	ng/kg bw/day	5.6	5.4	4.7	5.1	5.1	4.3	5.0	5.3	4.0	3.7	5.4
Organochlorine Pesticides												
Dieldrin	µg/kg bw/day	0.10	0.12	0.29	0.34	0.12	0.29	0.25	0.55	0.84	0.77	0.84
Total DDx	µg/kg bw/day	1.2	1.4	2.3	2.2	1.4	2.0	2.2	4.1	4.5	4.1	5.6

bw – body weight
COPEC – chemical of potential ecological concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
PCDD – polychlorinated dibenzo-*p*-dioxin
PCDF – polychlorinated dibenzo-*p*-furan
TBT – tributyltin
TEQ – toxic equivalency
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT and 4,4′-DDT)

7.2.3 Effects

This section presents the TRVs derived for the COPECs identified in the SLERA.

7.2.3.1 Methods for selecting TRVs

One set of fish dietary TRVs was used to derive HQs. TRVs were selected by first conducting a comprehensive literature search for relevant toxicological studies. These studies were then evaluated for acceptability of use. For those studies considered acceptable, NOAEL and LOAEL daily doses were derived. For those studies considered acceptable, NOAEL and LOAEL daily doses were derived as described in Appendix E. Details regarding the literature search and acceptability of the studies are presented in Appendix E. The TRV derivation and selection processes, along with general uncertainties in the use of TRVs to estimate risk, are described in more detail below.

TRV Derivation

Dietary TRVs for fish were expressed as a daily dose in mg/kg bw/day. However, many studies reported toxicity results as the chemical concentration in food associated with adverse effects, rather than as a daily dose. If the daily exposure dose was not presented in a study, it was derived using the reported concentration in food, the fish body weight (kg), either the ingestion rate (kg/day) reported in the study or a published value, and the following equation:

$$TRV = \frac{EPC_{diet} \times IR}{BW} \qquad \textbf{Equation 7-4}$$

Where:

TRV	=	toxicity reference value (mg/kg bw/day)
EPC_{diet}	=	exposure point concentration in diet (mg/kg)
IR	=	ingestion rate (kg/day)
BW	=	body weight (kg)

Detailed information regarding the conversion of dietary concentrations reported in the literature to body weight-normalized TRVs is presented in Appendix E.

TRV Selection Process

When sufficient data were available (i.e., data for at least five species), TRVs were generated using an SSD approach. The LOAEL TRV was selected as the 5th percentile of the SSD, and the NOAEL TRV was extrapolated from the LOAEL TRV using an uncertainty factor of 10.. When data were insufficient to generate an SSD, the lowest acceptable LOAEL and highest NOAEL below the LOAEL from the same study were selected as TRVs. Additional details on the SSD approach are presented in Section 7.1.3.1. Details regarding the literature search and acceptability of the studies are presented in Appendix E.

TRV Uncertainty

The dietary approach for inorganic metals is uncertain because the uptake and toxicity of inorganic metals to fish can vary widely depending on digestive physiology (e.g., gut residence time), food nutritional quality, distribution and chemical form (of metals) in prey tissue, and environmental conditions under which toxicity is evaluated (e.g., temperature) (USEPA 2007e). Metals are ubiquitous in the environment and most aquatic organisms have specific mechanisms for metals uptake, internal transport, sequestration, and depuration (Meyer et al. 2005). Essential metals are actively regulated by many aquatic organisms, because such metals are necessary for normal metabolic function; other non-essential metals may be regulated because they mimic essential elements and are transported by the same mechanisms (Bury et al. 2003). Two recent review papers by Meyer et al. (2005) and Wang (2013) summarize the state of scientific knowledge concerning dietary metals toxicity. Both these papers and USEPA's framework for metals risk assessment (USEPA 2007e) indicate that the current understanding of dietary metals toxicology is insufficient to accurately predict site-specific risks based on laboratory toxicity studies.

The dietary toxicity of metals to fish is dependent upon a number of factors, including the type of food, the potential for contaminated food to leach into water and sediment, the bioavailability of metals (e.g., distribution in prey and the metal speciation), and the mixture of metals in food (Meyer et al. 2005; Wang 2013; Clearwater et al. 2002). These factors are discussed below.

- **Type of food** – The type of food used in toxicity tests and in the natural environment is an important factor in determining dietary toxicity. For laboratory toxicity tests, natural prey, as opposed to formulated diets, are generally considered more environmentally realistic. Within laboratory-prepared formulated diets, fish or wheat meal may contain ingredients such as fatty acids, minerals, flavor enhancers, or dietary supplements, all of which can influence the bioavailability of metals or otherwise complicate the interpretation of dietary studies (Clearwater et al. 2002). However, laboratory-exposed natural prey can also be problematic. For example, to study the effects of metals contamination in the Clark Fork River, Montana, Mount et al. (1994) made a comprehensive effort to create realistic metals-contaminated diets in the laboratory using live invertebrates. However, chemical analysis showed that the metals distribution in and digestion of these invertebrates were different from those of metals-contaminated invertebrates obtained from the wild (Farag et al. 2000; Suedkamp 1999).
- **Potential for contaminated food to leach into water and sediment** – In the natural environment, concentrations of metals in contaminated food are equal to, or will come into equilibrium with, those in water and sediment. However, in laboratory toxicity tests, metals in contaminated food may leach into the water; the degree of leaching depends on the suborganismal location of each metal

(e.g., intracellular vs. extracellular), its different partitioning coefficient, and the particle load (Wang 2013). Simple partitioning calculations indicate that any change in food concentration can cause a redistribution of metals between the particles and the dissolved phase (Wang and Fisher 1998). Additionally, metals may be released from live food as a result of direct exudation (Zhang and Wang 2004). Thus, the release kinetics of the metals concerned and the contribution of waterborne uptake to overall metals accumulation and toxicity are critical factors in understanding dietary toxicity.

- **Bioavailability of metals** – Within live prey, the bioavailability of metals can vary based on their distribution and speciation within prey organisms. Metals can be sequestered in sub-cellular granules, metal-binding proteins (e.g., metallothionein), or the carapaces of invertebrates. Organically incorporated metals (e.g., in metallothionein) are generally more bioavailable, whereas those in granules or other storage mechanisms are not (Wang 2013). Several studies have demonstrated that a variety of essential and non-essential metals (e.g., silver, cadmium, selenium, lead, and zinc) in algae cytoplasm are bioavailable to consumers, whereas metals bound to the algae's cell wall are unavailable (Reinfelder and Fisher 1991; Hutchins et al. 1995; Stewart and Fisher 2003).
- **Mixture of metals in food** – Distribution of metals within prey organisms has also been shown to affect their toxicity to fish. For example, when zebrafish were fed cadmium in diets consisting of either different fractions of the crustacean *Gammarus pulex* (i.e., bioavailable metallothionein-like protein [MTLP]) or less bioavailable metal-rich granules (MRGs) and exoskeleton (i.e., MRG + exoskeleton), the MRG + exoskeleton fraction caused more oxidative damage to zebrafish than did the MTLP fraction (Khan et al. 2010a). In a similar study conducted with zebrafish exposed to zinc or copper in *G. puplex* fractions, copper in the MRG + exoskeleton fraction caused greater oxidative stress than did zinc (Khan et al. 2010b). Additionally, the biotransformation of metals within an organism can affect their toxicity: juvenile grunt (*Terapon jaruba*) were found to convert dietary doses of inorganic arsenic(III) and arsenic(IV) to non-toxic arsenobetaine (Zhang et al. 2011). It has been demonstrated that in organisms with more complex digestive tracts, gut transit time can affect the bioavailability of metals in different cell fractions (Wang and Fisher 1999; Roditi and Fisher 1999), further complicating accurate prediction of toxic doses across species.

In addition to the factors discussed above, fish species differ in their handling of food (e.g., whether the carapace is ingested), organ-specific assimilation, and storage/detoxification mechanisms (Wang 2013). Furthermore, as Clearwater et al. (2002) demonstrated for copper and zinc, the feeding rate is important in determining the effective dose of metal. However, many toxicity studies do not report the doses administered to experimental animals. In addition to feeding rate, Wang (2013) suggests that the bioavailable fraction of metal should be noted when reporting dietary toxicity testing results.

Multiple factors affect the bioavailability and toxicity of metals from the fish dietary pathway in ways that cannot be determined based on site-specific data. Therefore, the dietary doses used to characterize exposure are affected by uncertainty, and may over- or under-predict the potential for unacceptable risk. In addition to the general uncertainties discussed above for fish dietary TRVs, there is uncertainty associated with calculating daily exposure doses when doses were not reported for a study. Herein, if a feeding rate was not reported, an ingestion rate of 2% of the body weight was assumed as a conservative estimate based on the FIRs commonly reported for laboratory toxicity studies. If the type of food was not reported, it was assumed that dried commercial or pelleted feed was used, as was the case for most fish studies. For wet food, a moisture content of 80% was assumed if not specified.

7.2.3.2 Selected TRVs for fish diet

Selected fish dietary TRVs are presented in Table 7-22 and are described for each COPEC in the following subsections.

Table 7-22. Fish dietary TRVs

COPEC	Units (ww)	TRV		Endpoint	Source
		NOAEL	LOAEL		
Metals					
Cadmium	mg/kg bw/day	0.0010[a]	0.010	growth (rockfish)	Kim et al. (2004); Kang et al. (2005)
Chromium	mg/kg bw/day	0.19	na	growth (grey mullet)	Walsh et al. (1994)
Cobalt	mg/kg bw/day	0.14[a]	1.4	growth (white carp)	Javed (2013)
Copper	mg/kg bw/day	1.0	2.0	growth (rockfish)	Kang et al. (2005)
Methylmercury/ mercury	µg/kg bw/day	0.56[a]	5.6	growth, reproduction, mortality, and behavior (10 species)	SSD-derived 5th percentile value
Nickel	mg/kg bw/day	0.14[a]	1.4	growth (Indian carp)	Javed (2013)
Selenium	mg/kg bw/day	0.011[a]	0.11	growth (rainbow trout) (NOAEL); growth, reproduction, and mortality (7 species) (LOAEL)	Knight et al. (2016) (NOAEL); SSD-derived 5th percentile value (LOAEL)
Vanadium	mg/kg bw/day	0.019[a]	0.19	growth (rainbow trout)	Hilton and Bettger (1988)
Zinc	mg/kg bw/day	19	38	growth (rainbow trout)	Takeda and Shimma (1977)

COPEC	Units (ww)	TRV		Endpoint	Source
		NOAEL	LOAEL		
Butyltins					
TBT	µg/kg bw/day	1.2[a]	12	growth (zebrafish)	Lima et al. (2015)
PAHs					
Benzo(a)pyrene	µg/kg bw/day	30	40	growth (rockfish)	Kim et al. (2008)
Total PAHs	µg/kg bw/day	6,200	18,000	growth (Chinook salmon)	Meador et al. (2006)
PCBs					
Total PCBs	µg/kg bw/day	5.0[a]	50	reproduction (barbel)	Hugla and Thome (1999)
PCB TEQ - Fish	ng/kg bw/day	0.0027[a]	0.027	mortality (rainbow trout)	Giesy et al. (2001)
PCDDs/PCDFs					
PCDD/PCDF TEQ - Fish	ng/kg bw/day	0.0027[a]	0.027	mortality (rainbow trout)	Giesy et al. (2001)
Total TEQ - Fish	ng/kg bw/day	0.0027[a]	0.027	mortality (rainbow trout)	Giesy et al. (2001)
Organochlorine pesticides					
Total DDx	µg/kg bw/day	14.3[a]	143	reproduction (brook trout)	Macek (1968)

[a] NOAEL extrapolated from LOAEL by a factor of 10.

bw – body weight
COPEC – chemical of potential ecological concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
LOAEL – lowest-observed-adverse-effect level
na – not applicable; no data were available
NOAEL – no-observed-adverse-effect level
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
PCDD – polychlorinated dibenzo-*p*-dioxin
PCDF – polychlorinated dibenzo-*p*-furan
SSD – species sensitivity distribution
TBT – tributyltin
TEQ – toxic equivalent
TRV – toxicity reference value
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT and 4,4′-DDT)
ww – wet weight

Cadmium

Ten studies examining effects on growth, reproduction, and mortality due to dietary cadmium were reviewed. Five LOAEL values and nine NOAEL values were available for five fish species (Atlantic salmon, goldfish, guppy, rainbow trout, and rockfish [*Sebastes* sp.]). These studies reported LOAEL values that ranged from 0.010 to 200 mg/kg bw/day. The lowest LOAEL of 0.01 mg/kg bw/day for decreased rockfish growth (Kim et al. 2004; Kang et al. 2005) was selected. This LOAEL was two to three orders of magnitude less than both the NOAELs and LOAELs reported in other toxicological studies. No lower NOAEL was identified, so a NOAEL of

0.0010 mg/kg bw/day was selected as the LOAEL TRV divided by 10. There is uncertainty associated with the use of an extrapolation factor to derive the NOAEL.

Szebedinsky et al. (2001) observed that physiological mechanisms and responses to chronic cadmium exposure, whether via diet or water, were complex. The relative concentrations of cadmium in water and diet, the presence of metallothionein in various organs, and the alkalinity of water all play a role in the bioaccumulation and toxicity of cadmium. Furthermore, Chowdhury et al. (2004) showed that acclimation to cadmium over time may impact cadmium uptake and toxicology. Based on these studies, uncertainty exists regarding the effects of chronic cadmium exposure.

Chromium

Two studies examining the effects of dietary chromium on the growth of two fish species (tilapia [*Oreochromis* sp.] and grey mullet) were found to meet TRV acceptability criteria. No LOAEL values were identified for chromium from these studies. The highest NOAEL (0.19 mg/kg bw/day) was selected as the NOAEL TRV based on Walsh et al. (1994), wherein grey mullet was simultaneously exposed to chromium in both sediment and diet. At 0.19 mg/kg bw/day, the chromium-exposed fish showed a significant increase in growth, which is not considered an adverse effect. There is uncertainty associated with the use of an unbounded NOAEL, as it may over-predict the potential for a no-adverse-effect level. There is also uncertainty due to the limited toxicity dataset available for dietary chromium and fish.

Cobalt

Three studies examining the effects of cobalt on growth and mortality were found to meet the TRV acceptability criteria. Six LOAELs were reported from these studies, ranging from 1.4 to 3.4 mg/kg bw/day for three fish species (white carp [*Cirrhina mrigala*], Indian carp [*Catla catla*], and rohu [*Labeo rohita*]). The FIR was estimated assuming a default feeding rate of 2% bw/day, because the reported feeding rate (0.18–0.19% bw/day) was excessively low. A NOAEL of 83.2 mg/kg bw/day was reported in one study. The lowest LOAEL of 1.4 mg/kg bw/day, associated with decreased growth in white carp after 12 weeks of dietary exposure, was selected as the LOAEL TRV (Javed 2013). No NOAEL was reported by Javed (2013), so the NOAEL TRV of 0.14 mg/kg bw/day was estimated as the LOAEL TRV divided by 10. There is uncertainty associated with the use of an extrapolation factor to derive the NOAEL.

Copper

Thirteen acceptable toxicity studies were identified that evaluated the effects of dietary copper. Seven LOAELs were reported for four species of fish (Atlantic salmon, grey mullet, rainbow trout, and rockfish), ranging from 2.0 to 60 mg/kg bw/day. Kang et al. (2005) reported the lowest LOAEL of 2.0 mg/kg bw/day associated with a 50% reduction in rockfish growth after 60 days of exposure to dietary copper in the form of a pelletized diet; this was selected as the LOAEL TRV. The NOAEL of 1.0 mg/kg bw/day from this same study was selected as the NOAEL TRV. Based on a comprehensive review of data available at the time, Clearwater et al. (2002) indicated that daily doses of copper that caused adverse effects appeared to be fairly consistent within species for a given life stage; however, the diet type (e.g., purified, practical, or live diet) affected toxic doses because copper chelated to organic compounds, altering bioavailability. Additionally, other chemicals present in diet (e.g., zinc), specific copper compounds present in diet, and water quality (especially temperature, and possibly salinity) appeared to affect the doses at which toxic effects were observed. These factors contribute to the uncertainty associated with the TRV for dietary copper in fish.

Mercury and Methylmercury

Thirteen studies examining growth, reproduction, mortality, and behavior were found to meet acceptability criteria. These studies reported 15 LOAELs for methylmercury ranging from 1.5 to 2,500 µg/kg bw/day in 10 species of fish (Atlantic croaker [*Micropogonias undulatus*], mummichog, fathead minnow, European sturgeon [*Huso huso*], walleye [*Sander vitreus*], zebrafish, blackfish [*Tautoga onitis*], rainbow trout, green sturgeon [*Acipenser medirostris*], and white sturgeon). An SSD was developed and the distribution of the final species LOAELs was best described by a log-logistic distribution. The 5th percentile LOAEL TRV based on the SSD was 5.6 µg/kg bw/day (Figure 7-6). The SSD-derived LOAEL is within the range of measured LOAELs derived from the literature. The NOAEL TRV (0.56 µg/kg ww) was extrapolated from the LOAEL TRV using an uncertainty factor of 10. There is uncertainty associated with the use of an extrapolation factor to derive the NOAEL.

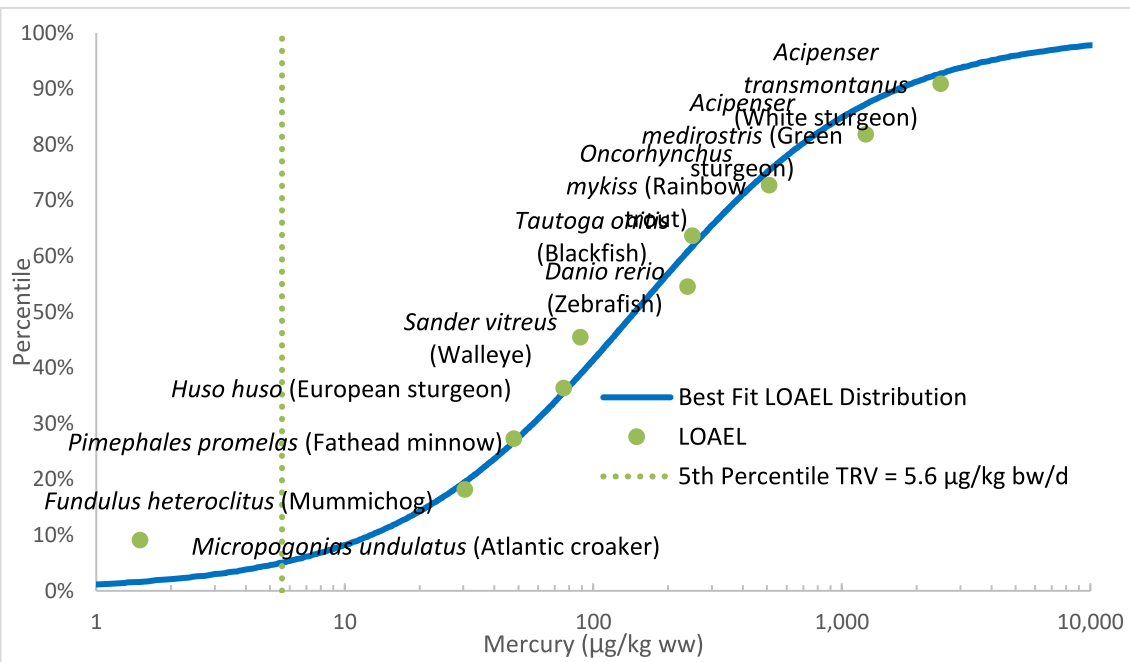


Figure 7-6.Fish diet methylmercury SSD toxicity data

Nickel

Three acceptable studies were identified that evaluated the growth effects of nickel. Three LOAELs and five NOAELs were identified, ranged from 1.4 to 1.6 and 2.9 to 27 mg/kg bw/day, respectively. The lowest LOAEL of 1.4 mg/kg bw/day, associated with decreased growth in Indian carp over 12 weeks, was selected (Javed 2013). The FIR was estimated assuming a default feeding rate of 2% bw/day, because the reported feeding rate (0.18–0.19% bw/day) was excessively low. No NOAEL was reported by Javed (2013), so a NOAEL of 0.14 mg/kg bw/day was estimated as the LOAEL divided by 10. There is uncertainty associated with the use of an extrapolation factor to derive a NOAEL.

Selenium

Eight studies examining growth, reproduction, and mortality were identified from the literature. These studies reported nine LOAELs for selenium ranging from 0.19 to 1.2 mg/kg bw/day in seven species of fish (Chinook salmon, rainbow trout, bluegill, Sacramento splittail [*Pogonichthys macrolepidotus*], striped bass, white sturgeon, and fathead minnow). An SSD was developed using LOAELs (Figure 7-7).

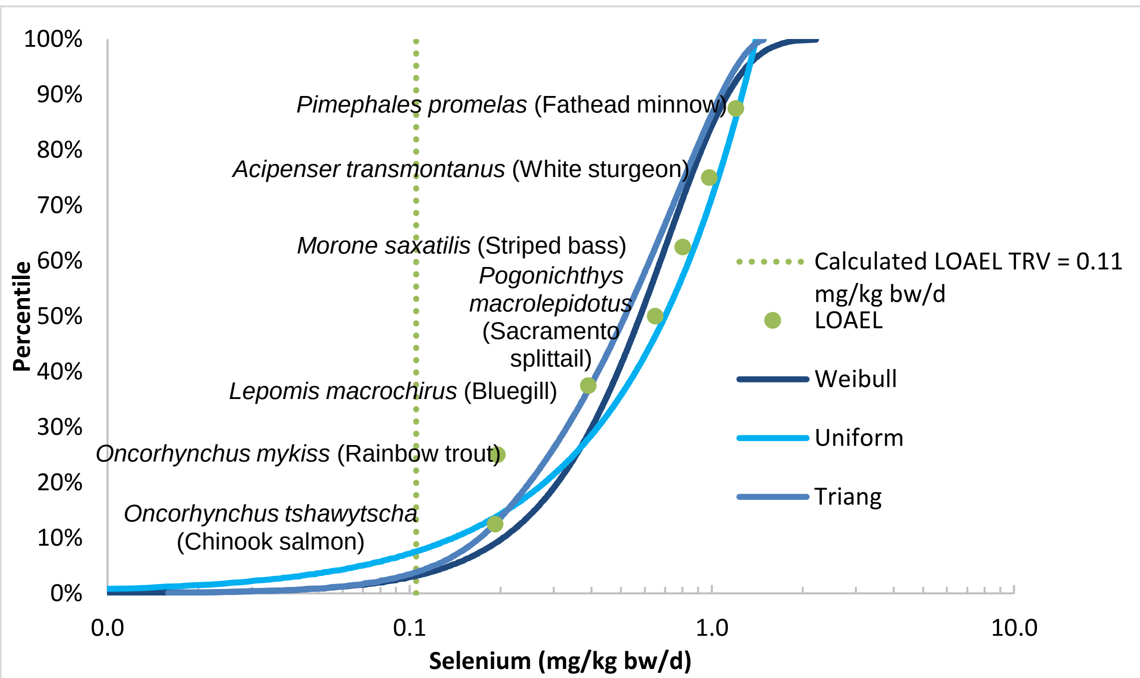


Figure 7-7.Fish diet selenium SSD toxicity data

The uniform, triangular, and Weibull distributions are the highest-ranking distributions according to the A-D statistic. Because these distributions fit the lower end of the data differently, all three distributions were selected, and the geometric mean of the 5th percentile values (0.11 mg/kg bw/day) for the uniform (0.07 mg/kg bw/day), triangular (0.12 mg/kg bw/day), and Weibull (0.14 mg/kg bw/day) distributions was selected as the LOAEL. This SSD-derived LOAEL (0.11 mg/kg bw/d) is less than the lowest measured LOAEL reported in the literature: a dose of 0.19 mg/kg bw/day associated with reduced survival in Chinook salmon after 90 days of exposure (Hamilton et al. 1990) (Appendix E). Thus, the SSD-derived LOAEL represents a conservatively extrapolated value that is less than those empirically measured in the reviewed toxicity studies. The NOAEL TRV (0.011 mg/kg ww) was extrapolated from the LOAEL TRV using an uncertainty factor of 10. There is uncertainty associated with the use of an extrapolation factor to derive the NOAEL.

Vanadium

One acceptable toxicity study was identified that evaluated the effects of dietary vanadium on fish. Hilton and Bettger (1988) reported a LOAEL of 0.19 mg/kg bw/day associated with a 260% reduction in the growth of rainbow trout relative to controls. The reported FIR was estimated assuming an average rainbow trout feeding rate of 1.9% bw/day, because the reported feeding rate (0.17% bw/day) was excessively low. The NOAEL TRV (0.019 mg/kg ww) was extrapolated from the LOAEL TRV using an uncertainty factor of 10. There is uncertainty due to the limited toxicity dataset available for dietary vanadium and fish (one study) and the use of an extrapolation factor to derive the NOAEL.

Zinc

One acceptable toxicity study was identified that evaluated the effects of dietary zinc on fish. Takeda and Shimma (1977) reported a LOAEL of 38 mg/kg bw/day associated with a 20% reduction in the growth of rainbow trout relative to controls. A NOAEL of 19 mg/kg bw/day was identified from the same study. These doses were selected as the LOAEL and NOAEL TRVs, respectively. However, the FIR was not reported, so these doses were estimated assuming an average rainbow trout feeding rate of 1.9% bw/day. Based on a comprehensive review of data available at the time, Clearwater et al. (2002) found no relationship between zinc toxicity to fish exposed to laboratory-prepared diets and factors such as diet type, supplemented metal compound (e.g., zinc sulfate or zinc carbonate), life stage, exposure duration, or water quality. Clearwater et al. (2002) indicated that the lack of any clear relationship was due, in part, to the lack of sufficient information to make valid comparisons between studies. There is also uncertainty due to the limited toxicity dataset available for dietary zinc and fish.

Tributyltin

Six studies examining the effects of TBT on growth, reproduction, and mortality were identified that met the TRV acceptability criteria. Five LOAELs and three NOAELs were identified, ranging from 0.012 to 10 and 0.012 to 1.0 mg/kg bw/day respectively. The six studies examined three species of fish (Japanese medaka, Japanese whiting [*Sillago japonica*], and zebrafish). A LOAEL of 0.012 mg/kg bw/day was associated with a decrease in female growth after 115 days of exposure (Lima et al. 2015). No growth NOAEL was reported by Lima et al. (2015), so a NOAEL of 0.0012 mg/kg bw/day was estimated as the LOAEL divided by 10. There is uncertainty associated with the use of an extrapolation factor to derive the NOAEL.

Benzo(a)pyrene

Seven studies examining effects on growth and mortality due to dietary benzo(a)pyrene were found to meet TRV acceptability criteria. Five LOAEL values and seven NOAEL values were available for five fish species (grouper [*Epinephelus malabaricus*], English sole [*Parophrys vetulus*], rainbow trout, rockfish [*Sebastes schlegelii*], and zebrafish). These studies reported LOAEL values that ranged from 40 to 19,000 µg/kg bw/day. Kim et al. (2008) exposed juvenile rockfish to dietary benzo(a)pyrene for 30 days, resulting in the lowest LOAEL of 40 µg/kg bw/day, associated with a 70% reduction in growth of juvenile rockfish. This LOAEL was selected as the LOAEL TRV. A NOAEL of 30 µg/kg bw/day from the same study was selected as the NOAEL TRV.

Total PAHs

Two studies examining effects of dietary PAHs on growth and the immune system (associated with increased mortality) were reviewed. One LOAEL and three NOAELs were available for one fish species (Chinook salmon). The only LOAEL was 18,000 µg/kg bw/day (Meador et al. 2006), associated with a 9% reduction in the dry weight of fish after 53 days of exposure. Meador et al. (2006) exposed juvenile Chinook salmon to a dietary PAH mixture designed to resemble a field PAH mixture from the Duwamish River in Seattle, Washington. However, the specific PAH mixture used in this study may not represent PAH concentrations found within the LPRSA; therefore, uncertainty in the applicability of these TRVs should be considered. A NOAEL of 6,200 µg/kg bw/day was identified and selected from the same study. There is uncertainty due to the limited toxicity dataset available for dietary PAHs and fish (only two studies, only one of which reported a LOAEL).

TEQ - fish

Four studies examining the effects on mortality and growth were found to meet TRV acceptability criteria. Three LOAELs and five NOAELs were identified, ranging from 0.027 to 6.2 and 1.4 to 15 ng/kg bw/day, respectively. These studies examined two fish species (rainbow trout and lake whitefish [*Coregonus clupaformis*]). The lowest LOAEL of 0.027 ng/kg bw/day, associated with increased mortality in rainbow trout following exposure to 1.8 ng/kg, was selected (Jones et al. 2001). There is uncertainty associated with an unbounded LOAEL based on a lethal effect (mortality). No NOAEL was identified by Jones et al. (2001), so a NOAEL of 0.0027 ng/kg bw/day was estimated as the LOAEL divided by 10. There is uncertainty associated with the use of an extrapolation factor to derive the NOAEL. This LOAEL (0.027 ng/kg bw/day) is two orders of magnitude less than the other two LOAELs derived from dietary 2,3,7,8-TCDD exposure (Appendix E).

Total PCBs

Eight studies examining the effects of dietary PCBs on fish growth, reproduction, and mortality met TRV acceptability criteria. Five LOAELs and four NOAELs were identified, ranging from 50 to 3,800 and 1.0 to 1,600 µg/kg bw/day, respectively. These studies examined seven fish species (channel catfish, Atlantic croaker, rainbow trout, Chinook salmon, barbel [*Barbus barbus*], mummichog, and tilapia [*Oreochromis mossambicus*]). The lowest LOAEL of 50 µg/kg bw/day, associated with a significant reduction in barbel fecundity (i.e., number of eggs per female), was selected (Hugla and Thome 1999). While fecundity was reduced at this dose, there was no significant effect on egg weight or hatching rate, thus, the relationship between the selected LOAEL of 50 µg/kg bw/day and the adverse effects at this dose is uncertain. A NOAEL of 5.0 µg/kg bw/day was estimated by dividing the LOAEL by 10. There is uncertainty associated with the use of an extrapolation factor to derive the TRV.

Total DDx

Nine studies examining the effects on mortality, growth, and reproduction were found to meet the TRV acceptability criteria. Eight LOAELs were identified, ranging from 140 to 6,000 µg/kg bw/day; ten NOAELs were identified, ranging from 2.3 to 1,500 µg/kg bw/day. These studies examined nine species of fish (brook trout [*Salvelinus fontinalis*], Chinook salmon, coho salmon, pinfish [*Lagodon rhomboides*], rainbow trout [*Salma gairdneri*], fathead minnow, largemouth bass, goldfish, and Atlantic menhaden). The lowest LOAEL of 143 µg/kg bw/day was selected as the LOAEL. At this LOAEL, embryo survival was reduced in brook trout following 156 days of exposure (Macek 1968). No NOAEL was identified by Macek (1968), so the NOAEL was estimated as the LOAEL divided by 10. There is uncertainty associated with the use of an extrapolation factor to derive the NOAEL.

7.2.4 Risk characterization

This section presents the dietary HQs for fish (Section 7.2.4.1), as well as uncertainties associated with the HQ calculations (Section 7.2.4.2).

7.2.4.1 Dietary HQs

Dietary HQs were calculated using the calculated doses presented in Table 7-21 (based on UCLs or maximum concentrations if there were fewer than six detected values) and the TRVs identified in Table 7-22. HQs are presented in Tables 7-23 and 7-24. Appendix G provides dietary doses, TRVs, and calculated HQs for the fish dietary COPECs in a single table (Table G4). LOAEL HQs were ≥ 1.0 for six COPECs:

- **Cadmium**: five species (mummichog [1.3], common carp [1.2], white sucker [1.1], white perch [1.1], American eel < 50 cm [1.2])
- **Mercury**: six species (mummichog [1.1], common carp [1.1], white catfish [1.1], white perch [1.3], American eel < 50 cm [1.3], American eel ≥ 50 cm [1.1])

- **Total PCBs**: one species (northern pike [1.3])
- **PCB TEQ - fish**: four species (American eel ≥ 50 cm [1.8], largemouth bass [1.6], smallmouth bass [1.5], northern pike [2.1])
- **PCDD/PCDF TEQ - fish**: all species (mummichog [200], common carp [200], white perch [170], channel catfish [190], white sucker [190], white catfish [160], American eel < 50 cm [180], American eel ≥ 50 cm [190], largemouth bass [150], smallmouth bass [140], northern pike [200])
- **Total TEQ** - fish: all species (mummichog [210], common carp [200], white perch [170], channel catfish [190], white sucker [190], white catfish [160], American eel < 50 cm [190], American eel ≥ 50 cm [200], largemouth bass[150], smallmouth bass [140], northern pike [200])

Table 7-23. Fish dietary LOAEL HQs

COPEC	LOAEL HQs[a]										
	Benthic Omnivores		Invertivore				Piscivore				
	Mummichog	Common Carp	White Perch	Channel Catfish	White Sucker	White Catfish	American Eel - Small (< 50 cm)	American Eel - Large (≥ 50 cm)	Largemouth Bass	Smallmouth Bass	Northern Pike
Metals											
Cadmium	**1.3**	**1.2**	**1.1**	0.99	**1.1**	0.89	**1.2**	0.70	0.51	0.47	0.24
Chromium	nc	nc	nc	nc	nc	nc	nc	nc	nc	nc	nc
Cobalt	0.036	0.027	0.031	0.025	0.033	0.022	0.037	0.016	0.015	0.014	0.0062
Copper	0.28	0.30	0.36	0.24	0.3	0.22	0.36	0.31	0.22	0.20	0.15
Mercury	**1.3**	**1.1**	**1.3**	0.98	0.99	**1.1**	**1.3**	**1.1**	0.84	0.77	0.74
Methyl mercury	0.032	0.17	0.37	0.18	0.13	0.26	0.28	0.59	0.47	0.43	0.72
Nickel	0.58	0.46	0.54	0.50	0.60	0.38	0.64	0.31	0.48	0.45	0.26
Selenium	0.30	0.25	0.40	0.29	0.29	0.28	0.42	0.35	0.43	0.39	0.30
Vanadium	0.35	0.32	0.30	0.34	0.32	0.30	0.34	0.21	0.31	0.28	0.16
Zinc	0.071	0.057	0.077	0.063	0.063	0.061	0.082	0.057	0.075	0.068	0.037
Butyltins											
TBT	0.0073	0.0078	0.028	0.011	0.0081	0.037	0.021	0.028	0.018	0.017	0.025
PAHs											
Benzo(a)pyrene	0.77	0.52	0.65	0.43	0.61	0.43	0.79	0.29	0.15	0.14	0.044
Total PAHs	0.011	0.011	0.0096	0.0098	0.013	0.0076	0.011	0.0050	0.0038	0.0035	0.0013
PCBs											
Total PCBs	0.35	0.30	0.48	0.46	0.33	0.42	0.48	0.98	0.79	0.73	**1.3**
PCB TEQ - fish	0.60	0.74	1.0	0.96	0.76	0.82	0.95	**1.8**	**1.6**	**1.5**	**2.1**

Table 7-23. Fish dietary LOAEL HQs

COPEC	LOAEL HQs[a]										
	Benthic Omnivores		Invertivore				Piscivore				
	Mummichog	Common Carp	White Perch	Channel Catfish	White Sucker	White Catfish	American Eel - Small (< 50 cm)	American Eel - Large (≥ 50 cm)	Largemouth Bass	Smallmouth Bass	Northern Pike
PCDDs/PCDFs											
PCDD/PCDF TEQ - fish	**200**	**200**	**170**	**190**	**190**	**160**	**180**	**190**	**150**	**140**	**200**
Total TEQ - fish	**210**	**200**	**170**	**190**	**190**	**160**	**190**	**200**	**150**	**140**	**200**
Organochlorine pesticides											
Dieldrin	nc	nc	nc	nc	nc	nc	nc	nc	nc	nc	nc
Total DDx	0.0085	0.0096	0.016	0.016	0.0096	0.014	0.015	0.029	0.032	0.029	0.039

Bold identifies HQs ≥ 1.0.

Shaded cells identify HQs ≥ 1.0 based on a LOAEL TRV.

[a] HQs were based on dietary doses presented in Table 7-21 and NOAEL and LOAEL TRVs derived from the primary literature review presented in Table 7-22.

COPEC – chemical of potential ecological concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
nc – not calculated
NOAEL – no-observed-adverse-effect level
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
PCDD – polychlorinated dibenzo-*p*-dioxin
PCDF – polychlorinated dibenzo-*p*-furan
TBT – tributyltin
TEQ – toxic equivalent
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT and 4,4′-DDT)
TRV – toxicity reference value

Table 7-24. Fish dietary NOAEL HQs

COPEC	NOAEL HQs[a]										
	Benthic Omnivores		Invertivore				Piscivore				
	Mummichog	Common Carp	White Perch	Channel Catfish	White Sucker	White Catfish	American Eel - Small (< 50 cm)	American Eel - Large (≥ 50 cm)	Largemouth Bass	Smallmouth Bass	Northern Pike
Metals											
Cadmium	**13**	**12**	**11**	**9.9**	**11**	**8.9**	**12**	**7.0**	**5.1**	**4.7**	**2.4**
Chromium	**7.7**	**6.4**	**6.8**	**6.5**	**7.9**	**5.0**	**8.1**	**4.0**	**5.1**	**4.6**	**3.1**
Cobalt	0.36	0.27	0.31	0.25	0.33	0.22	0.37	0.16	0.15	0.14	0.062
Copper	0.56	0.60	0.71	0.47	0.59	0.45	0.71	0.63	0.44	0.40	0.30
Mercury	**13**	**11**	**13**	**9.8**	**9.9**	**11**	**13**	**11**	**8.4**	**7.7**	**7.4**
Methyl mercury	0.32	**1.7**	**3.7**	**1.80**	**1.3**	**2.6**	**2.8**	**5.9**	**4.7**	**4.3**	**7.2**
Nickel	**5.8**	**4.6**	**5.4**	**5.0**	**6.0**	**3.8**	**6.4**	**3.1**	**4.8**	**4.5**	**2.6**
Selenium	**3.0**	**2.5**	**4.0**	**2.9**	**2.9**	**2.8**	**4.2**	**3.5**	**4.3**	**3.9**	**3.0**
Vanadium	**3.5**	**3.2**	**3.0**	**3.4**	**3.2**	**3.0**	**3.4**	**2.1**	**3.1**	**2.8**	**1.6**
Zinc	0.14	0.11	0.15	0.13	0.13	0.12	0.16	0.11	0.15	0.14	0.073
Butyltins											
TBT	0.073	0.078	0.28	0.11	0.081	0.37	0.21	0.28	0.18	0.17	0.25
PAHs											
Benzo(a)pyrene	**1.0**	0.69	0.86	0.58	0.81	0.57	**1.1**	0.38	0.20	0.18	0.059
Total PAHs	0.031	0.033	0.028	0.028	0.037	0.022	0.033	0.014	0.011	0.010	0.0037
PCBs											
Total PCBs	**3.5**	**3.0**	**4.8**	**4.6**	**3.3**	**4.2**	**4.8**	**9.8**	**7.9**	**7.3**	**13**
PCB TEQ - fish	**6.0**	**7.4**	**10**	**9.6**	**7.6**	**8.2**	**9.5**	**18**	**16**	**15**	**21**
PCDDs/PCDFs											
PCDD/PCDF TEQ - fish	**2,000**	**2,000**	**1,700**	**1,900**	**1,900**	**1,600**	**1,800**	**1,900**	**1,500**	**1,400**	**2,000**

COPEC	NOAEL HQs[a]										
	Benthic Omnivores		Invertivore				Piscivore				
	Mummichog	Common Carp	White Perch	Channel Catfish	White Sucker	White Catfish	American Eel - Small (< 50 cm)	American Eel - Large (≥ 50 cm)	Largemouth Bass	Smallmouth Bass	Northern Pike
Total TEQ - fish	**2,100**	**2,000**	**1,700**	**1,900**	**1,900**	**1,600**	**1,900**	**2,000**	**1,500**	**1,400**	**2,000**
Organochlorine pesticides											
Dieldrin	0.012	0.015	0.036	0.042	0.015	0.035	0.030	0.068	0.10	0.095	0.10
Total DDx	0.085	0.096	0.16	0.16	0.096	0.14	0.15	0.29	0.32	0.29	0.39

Bold identifies HQs ≥ 1.0.

[a] HQs were based on dietary doses presented in Table 7-21 and NOAEL and LOAEL TRVs presented in Table 7-22.

COPEC – chemical of potential ecological concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
nc – not calculated
NOAEL – no-observed-adverse-effect level
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
PCDD – polychlorinated dibenzo-*p*-dioxin
PCDF – polychlorinated dibenzo-*p*-furan
TBT – tributyltin
TEQ – toxic equivalent
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT and 4,4′-DDT)
TRV – toxicity reference value

7.2.4.2 Uncertainties in risk characterization

As noted in the introduction to Section 7.2.3.1, metals dietary toxicology is uncertain and has been questioned as a means to quantify dietary risk, specifically with regard to the high variability in uptake and toxicity of inorganic metals in fish. USEPA has recommended that this dietary approach for fish be used "only for conservatively screening for exposure and potential risks to consumers (i.e., in cases where whole-body residues in prey are below dietary toxic thresholds)" (USEPA 2007e). For more definitive assessments, USEPA suggests that further research is needed to quantify the bioavailability and effects of inorganic dietary metals (USEPA 2007e).

Because of the high uncertainty associated with the dietary assessment discussed above, uncertainties were identified but not quantitatively evaluated. These uncertainties include the following:

- **Modeled diet –** As discussed in Section 7.2.2.3, dietary items were limited to prey species with tissue chemistry data available from the LPRSA, including bioaccumulation worm, blue crab, and fish tissue, and did not include other prey items that may be important components of fish diets, such as amphipods, algae, zooplankton, or detritus.[109] The uncertainty was particularly high for species with a large portion of prey items not available for modeling (carp and white sucker). In addition to the limited types of prey used in the fish dietary model, the selection of explicit prey portions did not reflect the largely opportunistic feeding behavior of most fish species. Therefore, the representativeness of the dietary estimates (based on available prey tissue data) for actual LPRSA fish diets is highly uncertain.
- **Food ingestion rate –** Measured FIRs for all fish species other than mummichog were not available. FIRs for these species were estimated as a function of body weight and temperature using an equation from Arnot and Gobas (2004). It is unknown whether the modeled FIRs reflect actual ingestion rates of LPRSA fish.
- **Sediment ingestion rate –** Measured incidental SIRs for all fish species other than American eel were not available. SIRs for these species were based on best professional judgment. It is unknown whether the estimated SIRs reflect actual ingestion rates of LPRSA fish.

[109] It should also be noted that bioaccumulation model fish diets presented in the LPRSA FS bioaccumulation model (Windward 2015b) differ from those in this BERA for several reasons. First, the estimated fish diets in the FS bioaccumulation model are not limited to prey types with empirical tissue chemistry data. The model includes a wider range of potential prey items (e.g., detritus, algae, and zooplankton) with estimated concentrations, whereas the BERA fish diets are only based on only prey items for which tissue concentration data are available. Second, while the FS bioaccumulation model uses point estimates (single values) for prey portions, the model also estimates fish dietary proportions based on iterations of Monte Carlo simulations using a range of prey portion values. This allows for the refinement of fish diets based on the available site-specific information.

- **Site use factor –** An SUF of 1 was used for all fish species. This SUF assumes there is no movement or foraging that occurs outside of the LPRSA. However, fish that may (white perch) or are known to (American eel) go outside the LPRSA seasonally or during specific life stages may have SUFs less than 1, which would reduce HQs. However, LOAEL HQs for white perch and American eel are already less than or nearly equal to one[110] based on a SUF of 1.
- **Exposure area –** Mudflats from the entire LPRSA were used as the exposure area for mummichog. Mummichog are known to have strong site fidelity with a home range that is dependent on site-specific factors. Historically, mummichog home ranges have been considered small, 36 to 38 m for adults (Lotrich 1975). More recent studies report ranges varying from 10s to 100s of meters (Currin et al. 2003); a recent recapture study that looked at mummichog site fidelity in areas with greater tidal fluctuations reported a home range of up to 650 m (Sweeney et al. 1998). Thus, the exposure area assumed for mummichog (all mudflats from the LPRSA) may overestimate actual mummichog exposure areas.
- **Treatment of non-detects for EPCs –** The concentrations of congeners that were not detected were assumed to be zero when calculating total PCBs and TEQs. Based on the evaluation of non-detects for other fish LOEs (see Section 7.1.4.2 [fish tissue uncertainty] and Section 7.3.4.2 [fish surface water uncertainty]) and other receptor groups (see Sections 8.1.4.2 [bird diet uncertainty] and Section 9.1.4.2 [mammal diet uncertainty]), the treatment of non-detects as zero is not expected to affect HQ calculations.

7.2.5 Summary of key uncertainties

The primary uncertainty associated with the dietary approach is that the uptake by and toxicity of inorganic metals to fish can vary widely. USEPA recommends a dietary assessment of metals only for conservative screening purposes, because the uptake by and toxicity of inorganic metals to fish can vary widely depending upon a number of factors, including (but not limited to) digestive physiology (e.g., gut residence time), food nutritional quality, distribution and chemical form of metals in prey tissue, and environmental conditions under which toxicity is evaluated (e.g., temperature) (USEPA 2007e). In addition, the representativeness of the dietary estimates (based on available prey tissue data that does not include prey items that may be important components of fish diets, such as amphipods, algae, zooplankton, or detritus) for actual LPRSA fish diets is highly uncertain.

Specific uncertainties associated with TRVs, including the derivation of TRVs using SSDs, are discussed in Sections 7.1.3.2 and 6.3.3.1. For the COPECs with TRVs based on 5th percentile LOAELs determined from SSDs (i.e., mercury and selenium), the range of

[110] The only LOAEL HQs ≥ 1.0 for white perch and American eel is for American eel ≤ 50 cm long and cadmium (LOAEL HQ is 1.2).

the empirical LOAELs and number of data points (i.e., number of species included in the SSD) are shown in Table 7-25 to provide context of uncertainty for SSD-derived values.

Table 7-25. Uncertainty evaluation of fish diet TRVs based on SSDs

COPEC	TRV Unit	NOAEL	LOAEL	No. of Species (count of LOAELs in SSD)	Empirical LOAEL Range	Notes on Key Uncertainties
Mercury	µg/kg bw/day	0.56	5.6	n = 10	1.5– 2,500	SSD-derived LOAEL within range of measured LOAELs
Selenium	mg/kg bw/day	0.107	0.11	n = 9	0.19 – 1.2	SSD-derived LOAEL < lowest measured LOAEL

Note: TRVs included in this table are based on SSDs that are based on TRVs derived from the general literature search.

bw – body weight
COPEC – chemical of potential ecological concern
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
SSD – species sensitivity distribution
TRV – toxicity reference value

7.2.6 Summary

Twenty dietary COPECs were evaluated for all fish species. A summary of LOAEL HQs ≥ 1.0 for fish diet are summarized in Table 7-26.

Table 7-26. Summary of fish dietary LOAEL HQs

COPEC	HQ		Key uncertainties
	Fish Species with HQs ≥ 1.0	LOAEL HQ Values ≥ 1.0	
Cadmium	HQs ≥ 1.0 for mummichog, carp, white perch, white sucker, American eel - small (< 50 cm)	1.1–1.3	• LOAEL TRV 2–3 orders of magnitude less than both NOAELs and LOAELs reported in other toxicological studies
Mercury	HQs ≥ 1.0 for mummichog, carp, white perch, white catfish, American eel - small (< 50 cm), American eel - large (≥ 50 cm)	1.1–1.3	• TRV based on SSD within range of measured LOAELs evaluated • LOAEL HQs for methylmercury < 1.0 for all species
Total PCBs	HQ ≥ 1.0 for northern pike	1.3	• LOAEL based on fecundity (number of eggs per female), but no significant reduction on egg weight or hatching rate was reported.
PCB TEQ - fish	HQs ≥ 1.0 for white perch, American eel - large (≥ 50 cm), largemouth bass, smallmouth bass, northern pike	1.0–2.1	• LOAEL TRV 2 orders of magnitude less than LOAELs reported for 2 other species
PCDD/PCDF TEQ - fish	HQs ≥ 1.0 for all fish species evaluated	140–200	
Total TEQ - fish	HQs ≥ 1.0 for all fish species evaluated	140–210	

COPEC – chemical of potential ecological concern
HQ – hazard quotient
PCB – polychlorinated biphenyl
LOAEL – lowest-observed-adverse-effect level
PCDD – polychlorinated dibenzo-p-dioxin
PCDF – polychlorinated dibenzo-p-furan
SSD – species sensitivity distribution
TEQ – toxic equivalency
TRV – toxicity reference value

There is a high uncertainty associated with the dietary approach for metals and fish and the exposure assumptions used; therefore, consistent with USEPA's recommendation for metals (USEPA 2007e), fish dietary metal HQs should not be used for the purposes of risk management conclusions and decisions. This is further discussed in Section 13.

7.3 Surface Water Assessment

The surface water assessment was conducted for fish to evaluate the effect of direct exposure to COPECs in surface water. Risk estimates are expressed as HQs, which were derived by comparing the surface water EPCs with the TRVs.

7.3.1 COPECs

Surface water COPECs for fish were identified in the SLERA as COIs with maximum concentrations equal to or exceeding their respective screening thresholds. Surface water COPECs for fish are presented in Table 7-27. The COPECs for fish are the same as those for benthic invertebrates (Section 6.2), except TEQs in surface water were also evaluated for fish. A number of COIs could not be screened as part of the SLERA (Appendix A) because no surface water screening levels were available. These COIs are presented in Section 5.5.2, as are the implications of not being able to evaluate these COIs.

Table 7-27. Surface water COPECs evaluated for fish

COPECs	Estuarine (RM 0–RM 13)	Freshwater (RM 4–RM 17.4)
Metals[a]		
Cadmium	X	X
Chromium	X	X
Copper	X	X
Lead	X	X
Mercury	X	X
Selenium	X	X
Silver	X	X
Zinc	X	X
Butyltin		
TBT	X	
PAHs		
Anthracene	X	X
Benzo(a)anthracene	X	X
Benzo(a)pyrene	X	X
Fluoranthene	X	X
Pyrene	X	X

Table 7-27. Surface water COPECs evaluated for fish

COPECs	Estuarine (RM 0–RM 13)	Freshwater (RM 4–RM 17.4)
SVOCs		
BEHP	X	X
BBP	X	X
PCBs		
Total PCBs	X	X
PCB TEQ - fish[b]	X	X
PCDDs/PCDFs		
2,3,7,8-TCDD	X	X
PCDD/PCDF TEQ - fish[b]	X	X
Total TEQ - fish[b]	X	X
Pesticides		
4,4′-DDE	X	X
4,4′-DDT	X	X
Dieldrin	X	
Hexachlorobenzene	X	X
Total chlordane	X	X
Total DDx	X	X
Other		
Cyanide	X	X

Note: X indicates COPEC based on SLERA NOAEL and/or LOAEL HQ ≥ 1.0. The same COPECs were also evaluated for zooplankton, except for TEQs.

a All metals were identified as COPECs based on the total concentrations.

b TEQs - fish in surface water were only evaluated in addition to 2,3,7,8-TCDD for the assessment of fish and surface water, not for benthic invertebrates or zooplankton.

BBP – butyl benzyl phthalate
BEHP – bis(2-ethylhexyl) phthalate
COPEC – chemical of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
PCDD – polychlorinated dibenzo-*p*-dioxin
PCDF –polychlorinated dibenzofuran
RM – river mile
SLERA – screening-level ecological risk assessment
SVOC – semivolatile organic compound
TBT – tributyltin
TCDD – tetrachlorodibenzo-*p*-dioxin
TEQ – toxic equivalent
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT and 4,4′-DDT)

A number of COIs could not be screened as part of the SLERA (Appendix A) because no freshwater or estuarine screening levels were available. These COIs are presented in Section 5.5.2, as are the implications of not being able to evaluate these COIs.

7.3.2 Exposure

The surface water EPCs for fish were calculated separately for two areas: between RM 0 and RM 13 for comparison to estuarine thresholds, and between RM 4 and 17.4 for comparison to freshwater thresholds. Both near-bottom (3 ft [0.9 m] above the bottom) and near-surface (3 ft [0.9 m] below the surface) samples collected throughout the LPRSA during various flow events in 2011, 2012, and 2013 (see Table 4-4) were used in EPC calculations. Surface water EPCs were based on a conservative upper-bound estimate of the mean concentration (i.e., UCL concentration). UCLs were calculated using USEPA's ProUCL® statistical package (Version 5.0.00) (USEPA 2013d) as described in Section 4.3.7.[111] UCL concentrations could not be derived for one COPEC (i.e., TBT) because of the limited number of detected concentrations; therefore, the maximum concentration was used as the EPC. Concentrations of individual surface water samples were also presented to determine the range of surface water concentrations over smaller areas of the LPRSA and over seasonal flow events.

Summary concentrations of all LPRSA surface water samples are presented in Table 7-28.

[111] The UCL recommended by USEPA's ProUCL® software (typically the 95% UCL, but in some cases the 97.5% or even the 99% UCL) was used. The selected UCL statistic is presented in Appendix C1.

Table 7-28. COPEC summary statistics for LPRSA site-wide surface water samples

COPEC	Estuarine (RM 0–RM 13)						Freshwater (RM 4–RM 17.4)					
	No. Detects/ No. Samples	%	Concentration				No. Detects/ No. Samples	%	Concentration			
			Min.	Max.	Mean	UCL			Min.	Max.	Mean	UCL
Metals (µg/L)												
Cadmium (dissolved)	272/320	85	0.004	0.149	0.046	0.043	113/154	73.4	0.004	0.149	0.038	0.035
Chromium (dissolved)	194/200	97	0.18	5.46	0.83	0.89	98/98	100	0.18	5.46	1.1	1.2
Copper (dissolved)	320/320	100	1.11	9.26	2.51	2.61	154/154	100	1.36	9.26	2.94	3.1
Lead (dissolved)	320/320	100	0.07	9.97	0.85	1.2	154/154	100	0.098	9.97	1.2	1.8
Mercury (dissolved) (ng/L)	319/320	99.7	0.26	91.5	6.4	9	154/154	100	0.29	91.5	8.1	13
Selenium (dissolved)	56/200	28	0.2	3.2	0.67	0.54	48/98	49	0.2	3.2	0.66	0.62
Silver (dissolved)	119/200	59.5	0.004	0.119	0.019	0.017	48/98	49	0.004	0.119	0.032	0.026
Zinc (dissolved)	200/200	100	1.54	18.5	7.1	7.5	98/98	100	2.1	18.5	6.8	8.5
Butyltin (µg/L)												
TBT	2/200	1	0.013	0.026	0.02	nc[a]	1/98	1	0.026	0.026[b]	na	na
PAHs (ng/L)												
Anthracene	190/200	95	1.81	140	13.6	15	98/98	100	2.41	120	15.6	17.2
Benzo(a)anthracene	193/200	96.5	3.76	316	37.6	41.1	98/98	100	6.65	316	49.9	56.6
Benzo(a)pyrene	181/200	90.5	7.14	560	61.9	65.7	96/98	98	9.67	560	82	95.3
Fluoranthene	200/200	100	14.9	583	109	120	98/98	100	25.5	583	145	161
Pyrene	200/200	100	19.2	587	112	123	98/98	100	23.2	587	146	165
SVOCs (µg/L)												
BEHP	18/167	10.8	1.2	6	2.5	1.7	14/90	15.6	1.2	6	2.4	1.8
BBP	48/168	28.6	0.14	25	0.84	0.74	26/91	28.6	0.14	25	1.2	1.1
PCBs (ng/L)												

Table 7-28. COPEC summary statistics for LPRSA site-wide surface water samples

COPEC	Estuarine (RM 0–RM 13)						Freshwater (RM 4–RM 17.4)					
	No. Detects/ No. Samples	%	Concentration				No. Detects/ No. Samples	%	Concentration			
			Min.	Max.	Mean	UCL			Min.	Max.	Mean	UCL
Total PCBs	320/320	100	0.0485	183	20.8	25.5	154/154	100	1.96	183	25.8	34
PCB TEQ - fish	320/320	100	1.39×10^{-8}	0.000335	2.51×10^{-5}	3.35×10^{-5}	154/154	100	3.72×10^{-6}	0.000335	3.47×10^{-5}	5.08×10^{-5}
PCDDs/PCDFs (ng/L)												
2,3,7,8-TCDD	273/320	85.3	0.000617	1.87	0.0215	0.0541	139/154	90.3	0.000996	1.87	0.0375	0.108
PCDD/PCDF TEQ - fish	316/320	98.8	6.35×10^{-7}	1.88	0.0203	0.0713	154/154	100	9.46×10^{-7}	1.88	0.0357	0.11
Total TEQ - fish	320/320	100	0.000718	1.88	0.0201	0.0559	154/154	100	0.000718	1.88	0.0357	0.11
Pesticides (ng/L)												
4,4′-DDE	184/200	92	0.22	8.26	1.1	1.2	93/98	94.9	0.29	8.26	1.5	1.7
4,4′-DDT	148/200	74	0.0509	3.82	0.45	0.41	87/98	88.8	0.0619	3.82	0.61	0.66
Dieldrin	179/200	89.5	0.16	3.18	1.1	1.1	98/98	100	0.412	3.18	1.4	1.5
Hexachlorobenzene	46/200	23	0.0836	2.57	0.403	0.19	20/98	20.4	0.119	1.74	0.441	0.2
Total chlordane	200/200	100	0.0967	15.9	2.52	3.01	98/98	100	0.875	15.9	3.85	4.32
Total DDx	199/200	99.5	0.216	21.1	2.93	3.25	98/98	100	0.26	21.1	4.02	4.78
Other (mg/L)												
Cyanide	11/200	5.5	0.003	0.031	0.009	0.01	10/98	10.2	0.003	0.014	0.0068	0.0068
Hardness as calcium carbonate[c]	na	na	na	na	na	nc[c]	98/98	100	1.1`	3,510	490	nc

Note: The UCL was selected as the EPCs, except where otherwise noted.

[a] Fewer than six samples were analyzed, so the maximum concentration was used as the EPC.

[b] The maximum concentration is the DL.

[c] Hardness as calcium carbonate is not a COPEC; summary statistics are presented for freshwater only because hardness data were used in the derivation of site-specific surface water TRVs (e.g., cadmium freshwater TRVs). A UCL for hardness was not calculated.

BBP – butyl benzyl phthalate
BEHP – bis(2-ethylhexyl) phthalate
COPEC – chemical of potential ecological concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
DL – detection limit
EPC – exposure point concentration
LPRSA – Lower Passaic River Study Area
na – not applicable (not detected)
nc – not calculated (insufficient number of detected values)
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
PCDD– polychlorinated dibenzo-p-dioxin
PCDF –polychlorinated dibenzofuran
RM – river mile
SVOC – semivolatile organic compound
TBT – tributyltin
TCDD – tetrachlorodibenzo-p-dioxin
TEQ – toxic equivalent
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT, and 4,4′-DDT)
TRV – toxicity reference value
UCL – upper confidence limit on the mean

7.3.3 Effects

For each surface water COPEC, chronic TRVs based on up-to-date toxicological data relevant to aquatic species were derived to predict risk to benthic invertebrates and fish (Appendix D).[112] TRVs used to estimate risk to the fish community are the same as those used to estimate risk to the benthic invertebrate community. Details on the methods used to derive surface water TRVs, including an overview of the selection process for each surface water COPEC, and general uncertainties associated with these TRVs are presented in Section 6.2.3. Additional details on the derivation process of surface water TRVs are presented in Appendix D. General uncertainty associated with surface water TRVs, specifically those based on SSDs or the BLM, are discussed in Section 6.2.3.1.

Table 7-29 presents the selected surface water TRVs and summarizes the general representativeness of the selected TRVs of fish toxicity.

[112] Some screening levels (e.g., for total PCBs, 2,3,7,8-TCDD, and other organic COIs) used in the SLERA were protective of wildlife or human health (i.e., a 304(a) aquatic life criterion using the FRV procedure issued in 1980 or 1986, which is no longer used by USEPA to derive chronic criteria). Such screening levels were not used in the evaluation of aquatic invertebrates and fish exposure to surface water.

Table 7-29. Surface water TRVs used in the evaluation of fish

COPEC	TRV Type	TRV (µg/L)[a]		TRV Derivation Method	Fish Toxicity Relative to Selected TRV
		Acute	Chronic		
Metals[b]					
Cadmium	estuarine	33	7.9	dissolved saltwater CMC and CCC (USEPA 2016a)	TRVs may be overly conservative for fish. Acute toxicity data included 16 fish and 78 invertebrate species, showing a wide range of sensitivity; the most sensitive species were invertebrates. The chronic TRV was derived from the FAV using an ACR (USEPA 2016a).
	freshwater	1.4–6.5	0.59–2.0	freshwater CMC and CCC (USEPA 2016a); TRV ranges reflect range of mean sample-specific hardness values	Acute TRV is expected to be protective of fish. Acute toxicity data included 33 fish species, showing a wide range of sensitivities; the six most sensitive genera were fish (including rainbow trout, a commercially and recreationally important species). Chronic TRV may be overly protective of fish, as the two most sensitive genera included in the chronic SSD dataset were invertebrates (USEPA 2016a).
Chromium	estuarine	1,100	50	saltwater AWQC for dissolved chromium(VI) USEPA (2017c)	Representativeness of the estuarine TRVs is unclear because USEPA (2017c) only indicates 1995 as the publication year of updated criteria. Freshwater chromium(VI) criteria were updated in 1995 (USEPA 1996), but the source of updated saltwater criteria is unclear.
	freshwater	16	11	freshwater CMC and CCC from USEPA (1996), converted to dissolved chromium using USEPA-recommended CF (USEPA 2017c)	TRVs may be overly conservative for fish. Acute toxicity data in USEPA (1996) included 17 fish and 17 invertebrate species; invertebrate species were generally the most sensitive to cadmium (e.g., 10 most sensitive genera were invertebrates).

Table 7-29. Surface water TRVs used in the evaluation of fish

COPEC	TRV Type	TRV (µg/L)[a]		TRV Derivation Method	Fish Toxicity Relative to Selected TRV
		Acute	Chronic		
Copper	estuarine	0.80–11.2[c,d]	0.80–11.2[c,d,e]	Sample-specific CMC based on saltwater BLM developed for most sensitive species (Chadwick et al. 2008); CMC assumed to be protective of chronic toxicity	The acute TRV is expected to be overly protective of acute and chronic toxicity to fish. The acute TRV for copper in saltwater is based upon the sensitivity of the invertebrate *Mytilus galloprovincialis*, which represents the most sensitive genus considered in USEPA (2003a). The fish *Paralichthys dentatus* (Summer flounder), 3rd most sensitive genus in the acute SSD, has similar sensitivity; however, this species is not closely related to LPRSA species. The LPRSA species winter flounder (of the same taxonomic order) is 10-fold less sensitive. Chronic data for saltwater organisms are limited, and evaluation of potential ACRs indicate that acute criteria or TRVs based on early life stages of sensitive invertebrates would be protective of chronic toxicity.
	freshwater	14.3–100[c,d]	8.9–62.1[c,d]	Sample-specific CMC and CCC (using ACR) based on freshwater BLM from (USEPA 2007f)	The acute TRV is expected to be conservative for fish. Acute toxicity data were considered for 38 species, with the 9 most sensitive species represented by cladocerans, snails, amphipods, and freshwater mussels. The acute TRV is driven by the sensitivity of invertebrates, with the most sensitive fish being about 10-fold less sensitive than the most sensitive invertebrate. The chronic TRV was based on applying an ACR of 3.22 to the acute TRV. Given the relative acute sensitivities of fish compared to invertebrates, and the range in ACRs for sensitive to moderately insensitive fish (i.e., 2.88 to 11.4 (USEPA 2007f)), the chronic TRV is expected to be overly protective of fish (Appendix D).

Table 7-29. Surface water TRVs used in the evaluation of fish

COPEC	TRV Type	TRV (µg/L)[a]		TRV Derivation Method	Fish Toxicity Relative to Selected TRV
		Acute	Chronic		
Lead	estuarine	100	9.7	proposed acute and chronic saltwater criteria (Church et al. 2017) based on acute and chronic SSDs	TRVs are expected to be protective of fish. Acute toxicity data are available for 54 species, and the 18 most acutely sensitive species are all invertebrates. Therefore, saltwater fish are relatively insensitive to lead. Chronic TRVs may be conservative for fish, but toxicity data are also limited for fish. Chronic toxicity data are available for 21 species, 19 of which are invertebrates. The chronic TRV is driven by the sensitivity of an invertebrate (a mysid), which is about 5 times more sensitive than the most sensitive fish species tested to date (Appendix D).
	freshwater	192–890[c,d]	7.4–42.3[c,d]	sample-specific CMC and CCC (using ACR) based on freshwater BLM (DeForest et al. 2017)	TRVs are conservative for fish. Acute toxicity data are available for 32 species, 11 of which are fish. The 4 most acutely sensitive species are invertebrates; the most acutely sensitive fish species are about 1 order of magnitude less sensitive than the acute TRV. Chronic toxicity data are available for 15 species, 11 of which are invertebrates. TRV is driven by the sensitivity of an invertebrate (a snail) and the 7 most sensitive species tested to date are invertebrates. The most sensitive fish species tested to date is about 30-fold less sensitive than the most sensitive invertebrate (Appendix D).
Mercury	estuarine	1.8	0.94	saltwater CMC and CCC from USEPA (1984) converted to dissolved mercury using USEPA's metals-specific CF (USEPA 2017c)	TRVs are expected to be protective of fish. Acute toxicity data from USEPA (1984) for fish and invertebrates showed a wide range of sensitivities to mercury, with the most sensitive species being invertebrates (Appendix D).
	freshwater	1.4	0.21	acute TRV is freshwater CMC from USEPA (1996) converted to dissolved mercury using USEPA's metals-specific CF (USEPA 2017c); chronic TRV is lowest LOEC from USEPA (2016c)	Acute TRV may be overly conservative for fish, as the CMC is based on the most sensitive invertebrate species. Chronic TRV is expected to be protective of fish, as it is the lowest chronic toxicity value for a fish species (Appendix D).

Table 7-29. Surface water TRVs used in the evaluation of fish

COPEC	TRV Type	TRV (µg/L)[a]		TRV Derivation Method	Fish Toxicity Relative to Selected TRV
		Acute	Chronic		
Selenium	estuarine	290	71	saltwater AWQC (USEPA 2017c)	The representativeness of the estuarine TRVs cannot be evaluated because the source of the criteria does not indicate how the values were derived.
	freshwater	na	3.1	chronic TRV is dissolved selenium CCC in lotic waters (USEPA 2016b); no acute TRV selected for selenium	Chronic TRV is expected to be protective of fish and other aquatic species; focus of CCC derivation was on fish species, which are particularly sensitive to selenium.
Silver	estuarine	5.54	2.0	5th percentile of saltwater SSD based on acute toxicity data divided by 2; chronic value derived using an ACR of 5.536	TRVs are expected to be protective of fish. Toxicity data included 12 fish and 11 invertebrate species; invertebrate species were generally more sensitive than fish species (Appendix D).
	freshwater	1.8	0.69	acute and chronic values based on a proposed freshwater BLM from an unpublished report (HydroQual et al. 2007)	TRV is expected to be protective of fish; BLM is based on both invertebrates and fish toxicity and accounts for influence of water quality characteristics (Appendix D).
Zinc	estuarine	75	19	acute TRV is the 5th percentile of an acute saltwater SSD divided by 2; chronic TRV is the 5th percentile of a chronic saltwater SSD	TRV may be overly conservative for fish. Toxicity data included 18 fish and 107 invertebrate species, showing a wide range of sensitivity among species (Appendix D).
	freshwater	195–1,660[c,dc]	44.8–229[c,d]	sample-specific CCC and CMC based on freshwater BLM from DeForest and Van Genderen (2012)	TRVs are expected to be protective of fish. Acute toxicity data were considered for 96 species, with the 10 most sensitive species representing cladocerans, fish, amphipods, and mussels. The 2nd most sensitive species was a fish. Chronic toxicity data were considered for 20 species, 10 of which were invertebrates. The most sensitive organism was an invertebrate (a water flea), and the 2nd and 3rd most sensitive species were fish (Appendix D).

Table 7-29. Surface water TRVs used in the evaluation of fish

COPEC	TRV Type	TRV (µg/L)[a]		TRV Derivation Method	Fish Toxicity Relative to Selected TRV
		Acute	Chronic		
Butyltins					
TBT	estuarine	0.42	0.066	USEPA-calculated saltwater FAV divided by 2 and FCV from USEPA (2003b)	TRV may be overly conservative for fish; toxicity data included in derivation of FCV from 26 invertebrate and 7 fish species indicate fish are less sensitive than some invertebrate species (one reported fish chronic threshold [SMCV] was 0.26 µg/L) (USEPA 2003b).
PCBs					
Total PCBs	estuarine	4.6	0.16	acute TRV is 5th percentile of saltwater SSD based on acute toxicity data, divided by 2; chronic TRV is lowest chronic LOEC (sheepshead minnow reproduction)[f]	Acute TRV may be overly conservative for fish, as the acute toxicity data included only 1 fish and 10 invertebrate species. Chronic TRV is based on toxicity data from the most sensitive fish species (sheepshead minnow) (Appendix D).
	freshwater	1.2	0.27	acute TRV is 5th percentile of acute SSD based on toxicity data from USEPA (1980d) and (USEPA 2016c); chronic TRV derived using an ACR of 8.4	TRV is expected to be protective of fish. Acute toxicity data included 15 fish and 10 invertebrate species. TRVs were based on the lowest SMAV, which was for a fish species (largemouth bass) (Appendix D).
PCDDs/PCDFs					
2,3,7,8-TCDD and TEQs - fish	estuarine	0.025	0.006	acute TRV is lowest sub-chronic LOEC for a saltwater species (*D. rerio*); chronic TRV derived using an ACR of 8.3 from Raimondo et al. (2007)	TRVs are based on sub-chronic toxicity data for the most sensitive fish species (*Z. danio*) and are expected to be protective of fish (Appendix D).
	freshwater	0.0041	9.8×10^{-4}	acute TRV is lowest acute LC50 for a freshwater species (Japanese medaka); chronic TRV derived using an ACR of 8.3 from Raimondo et al. (2007)	TRV is based on toxicity data for the most sensitive fish species (Japanese medaka) and is expected to be protective of fish (Appendix D).

Table 7-29. Surface water TRVs used in the evaluation of fish

COPEC	TRV Type	TRV (µg/L)[a]		TRV Derivation Method	Fish Toxicity Relative to Selected TRV
		Acute	Chronic		
Organochlorine Pesticides					
4,4′-DDE	estuarine	1.25	0.30	acute TRV is lowest acute toxicity value for saltwater invertebrate species (*Nitocra spinipes*) divided by 2; chronic TRV is lowest chronic toxicity value for the same species	TRVs may be overly conservative for fish, as they are based on the lowest acute and chronic toxicity values available in USEPA (2016c), which were both for a copepod (*N. spinipes*) (Appendix D).
	freshwater	2.40	1.40	acute TRV is 5[th] percentile of freshwater SSD based on acute toxicity data, divided by 2; chronic TRV derived using an ACR of 3.6 for DDT-type chemicals	TRV may be overly conservative for fish. Toxicity data included 3 fish and 2 invertebrate species. The chronic TRV is less than the lowest chronic toxicity value identified in USEPA (2016c) (Appendix D).
4,4′-DDT/total DDx	estuarine	0.034	0.019	acute TRV is 5[th] percentile of acute SSD based on saltwater data from USEPA (1980c) and USEPA (2016c), divided by 2; chronic TRV derived using an ACR for DDT-type chemicals	TRV is expected to be protective of fish; TRV represented by SSD that includes toxicity based on 14 fish and 18 invertebrate species (Appendix D).
	freshwater	0.45	0.25	acute TRV is 5[th] percentile of acute SSD based on freshwater data from USEPA (1980c) and USEPA (2016c), divided by 2; chronic TRV derived using an ACR for DDT-type chemicals	TRV is expected to be protective of fish; the acute and chronic TRVs are less than the lowest fish SMAV of 1.4 (largemouth bass); TRV is based on SSD that incorporates toxicity data from 42 fish species (USEPA 1980c) (Appendix D).

Table 7-29. Surface water TRVs used in the evaluation of fish

COPEC	TRV Type	TRV (µg/L)[a] Acute	Chronic	TRV Derivation Method	Fish Toxicity Relative to Selected TRV
Total chlordane	estuarine	0.045	0.0064	USEPA-calculated saltwater CMC and CCC from USEPA (1980b)	TRV may be overly conservative for fish; TRV is based on FCV that incorporates toxicity data from only 4 fish species; however, chronic toxicity data indicate that fish are less sensitive than invertebrates; chronic thresholds [SMCVs] ranged from 0.63 to 11 µg/L (USEPA 1980b).
	freshwater	1.2	0.17	USEPA-calculated freshwater CMC and CCC from USEPA (1980b)	TRV may be overly conservative for fish; TRV is based on FCV that incorporates toxicity data from 9 fish species; however, chronic toxicity data indicate that freshwater fish are less sensitive than invertebrates; the lowest chronic threshold was 1.6 µg/L for bluegill (USEPA 1980b).
Dieldrin	estuarine	0.36	0.084	USEPA-calculated saltwater CMC and CCC from USEPA (1980a)	TRV may be overly conservative for fish; TRV is based on FCV that incorporates toxicity data from 13 fish species; however, chronic toxicity data indicate that freshwater fish are less sensitive than invertebrates; the lowest SMCV reported for fish was 0.22 µg/L for early life stage rainbow trout (USEPA 1980a).
Hexachlorobenzene	saltwater	71	23	lowest acute LC50 for a saltwater species (*Solea solea*) divided by 2; chronic value derived using an ACR	Toxicity data are limited for saltwater species. TRVs are expected to be protective of fish, as both are based on the lowest acute toxicity value for a fish species (Appendix D).
	freshwater	180	57	5^{th} percentile of freshwater SSD based on acute toxicity data; chronic value derived using an ACR	TRV may be overly conservative for fish. TRV is based on invertebrate toxicity; the 5^{th} percentile based on acute toxicity data for fish divided by an ACR based on Raimondo et al. (2007) results in a chronic value of 57 µg/L (Appendix D).

Table 7-29. Surface water TRVs used in the evaluation of fish

COPEC	TRV Type	TRV (µg/L)[a] Acute	TRV (µg/L)[a] Chronic	TRV Derivation Method	Fish Toxicity Relative to Selected TRV
PAHs					
Anthracene	estuarine	34.5	13.5	acute TRV is lowest acute LC50 for a saltwater species (*Mulinia lateralis*) divided by 2; chronic TRV derived using an ACR of 5.09 from DiToro et al. (2000)	Uncertainty exists in using a TRV based on toxicity to an invertebrate species to evaluate risk to fish; no acceptable chronic toxicity data for fish were available (Appendix D).
	freshwater	0.26	0.10	acute TRV is 5th percentile of freshwater SSD based on acute toxicity data divided by 2; chronic value derived using an ACR of 5.09	TRV is expected to be protective of fish. Acute toxicity data included 4 fish and 2 invertebrate species, with fish species being the most sensitive (Appendix D).
Benzo(a)anthracene	estuarine	0.48	0.19	same as freshwater TRVs[f]	same as freshwater TRVs
	freshwater	0.48	0.19	acute TRV is lowest acute LC50 for a freshwater species (*D. magna*); chronic TRV derived using an ACR of 5.09	Uncertainty exists in using a TRV based on toxicity to an invertebrate species to evaluate risk to fish; no acceptable chronic toxicity data for fish were available for comparison (Appendix D).
Benzo(a)pyrene	estuarine	0.51	0.20	acute TRV is lowest acute LC50 for *D. magna*, divided by 2; chronic TRV derived using an ACR of 5.09	Uncertainty exists in using a TRV based on toxicity to an invertebrate species to evaluate risk to fish; no acceptable chronic toxicity data for fish were available for comparison (Appendix D).
	freshwater	2.03	0.80	acute TRV is 5th percentile of freshwater SSD based on acute toxicity data divided by 2; chronic TRV derived using an ACR of 5.09	TRV may be overly conservative for fish. SSD based on acute toxicity data for 2 fish and 3 invertebrate species; data indicate that freshwater fish are less sensitive than invertebrates. The only chronic value reported for fish was 411 µg/L for zebrafish (Appendix D).

Table 7-29. Surface water TRVs used in the evaluation of fish

COPEC	TRV Type	TRV (µg/L)[a] Acute	TRV (µg/L)[a] Chronic	TRV Derivation Method	Fish Toxicity Relative to Selected TRV
Fluoranthene	estuarine	3.02	1.19	acute TRV is 5th percentile of saltwater SSD based on acute toxicity data, divided by 2; chronic TRV derived using an ACR of 5.09	TRVs may be overly conservative for fish. Acute toxicity data included only 3 fish and 16 invertebrate species, with invertebrate species being among the most sensitive (Appendix D).
	freshwater	13.2	5.20	acute TRV is 5th percentile of freshwater SSD based on acute toxicity data; chronic TRV derived using an ACR of 5.09	TRVs are expected to be protective of fish. Acute toxicity data included 4 fish and 10 invertebrate species. TRVs are less than the lowest fish and invertebrate SMAVs (Appendix D).
Pyrene	estuarine	0.46	0.18	acute TRV is lowest acute EC50 for a saltwater species (*M. lateralis*) divided by 2; chronic TRV derived using an ACR of 5.09	Uncertainty in using a TRV based on toxicity to an invertebrate species to evaluate risks to fish; no acceptable chronic toxicity data for fish were available (Appendix D).
	freshwater	2.2	0.84	acute TRV is lowest acute EC50 for a freshwater species (*D. magna*); chronic TRV derived using an ACR of 5.09	Uncertainty in using a TRV based on toxicity to an invertebrate species to evaluate risk to fish; no acceptable chronic toxicity data for fish were available (Appendix D).
SVOCs					
BEHP	estuarine	500	100	acute TRV is lowest LC50 divided by 2; chronic TRV derived using an ACR of 6.9 based on DeFoe et al. (1990)	Uncertainty in using an acute TRV based on toxicity to an invertebrate species to evaluate risk to fish; chronic TRV is expected to be protective of fish because it is based on the lowest available chronic value for a fish species (Appendix D).
	freshwater	24.1	7.0	acute TRV is 5th percentile of freshwater SSD based on acute toxicity data; chronic TRV derived using an ACR of 6.9	TRV is expected to be protective of fish. Acute SSD includes toxicity based on 12 fish and 4 invertebrate species, with multiple fish species being among the most sensitive (Appendix D).

Table 7-29. Surface water TRVs used in the evaluation of fish

COPEC	TRV Type	TRV (µg/L)[a] Acute	TRV (µg/L)[a] Chronic	TRV Derivation Method	Fish Toxicity Relative to Selected TRV
BBP	estuarine	245	71	acute TRV is lowest acute LC50 for a saltwater species (*C. aggregata*); chronic TRV derived using an ACR of 6.9	TRVs are expected to be protective of fish, as both are derived from the lowest acute toxicity value for a fish species (*C. aggregate*) (Appendix D).
	freshwater	107	30.9	acute TRV is 5th percentile of freshwater SSD based on acute toxicity data; chronic TRV derived using an ACR of 6.9	TRVs are expected to be protective of fish. SSD includes toxicity data based on 4 fish and 4 invertebrate species, with fish species being among the most sensitive (Appendix D).
Other					
Cyanide	estuarine	6.1	1.9	acute TRV is 5th percentile of SSD based on acute toxicity data of 13 invertebrate and 3 fish species; chronic TRV derived using an ACR of 8.6 from Gensemer et al. (2006)	TRV may be overly conservative for fish; toxicity data included in SSD indicate fish may be less sensitive than invertebrates (fish acute thresholds range from 59 to 372 µg/L) (Appendix D).
	freshwater	32.3	7.5	Acute TRV is 5th percentile of SSD based on acute toxicity data of 24 invertebrate and 11 fish species; chronic TRV derived using an ACR of 6.5 from Gensemer et al. (2006)	TRV is expected to be protective of fish; low range of values in SSD based on fish toxicity (Appendix D).

a NOAEL TRVs were not developed for surface water; SSD-derived 5th percentile TRVs were based on effects levels from the literature.

b TRVs for metals are based on the dissolved chemical form.

c For COPECs with BLM-based TRVs, the distinction between freshwater and saltwater was based on 3.5 ppth salinity.

d As they are sample specific, the BLM-based TRVs are a range of values (i.e., each individual sample has a corresponding BLM-based TRV).

e Due to lack of chronic copper toxicity data for saltwater species, the sample-specific acute BLM-based TRVs were also used as the chronic TRVs.

f The freshwater TRVs for benzo(a)anthracene were selected as surrogate estuarine TRVs due to lack of saltwater toxicity data.

ACR – acute-to-chronic ratio
AWQC – ambient water quality criteria
BBP – butyl benzyl phthalate
BEHP – bis(2-ethylhexyl) phthalate
BLM – biotic ligand model
FAV – final acute value
FCV – final chronic value
LC50 – concentration that is lethal to 50% of an exposed population
LOEC – lowest-observed-effect concentration
SMAV – species mean acute value
SMCV – species mean chronic value
SSD – species sensitivity distribution
SVOC – semivolatile organic compound
TBT – tributyltin

CCC – criterion continuous concentration
CF – conversion factor
CMC – criterion maximum concentration
COPEC – chemical of potential ecological concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
EC50 – concentration that causes a non-lethal effect in 50% of an exposed population
LPRSA – Lower Passaic River Study Area
NOAEL – no-observed-adverse-effect level
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
PCDD– polychlorinated dibenzo-*p*-dioxin
PCDF –polychlorinated dibenzofuran
ppth – parts per thousand
TCDD – tetrachlorodibenzo-*p*-dioxin
TEQ – toxic equivalent
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT and 4,4′-DDT)
TRV – toxicity reference value
USEPA – US Environmental Protection Agency

7.3.4 Risk characterization

The following section presents the calculated surface water HQs for fish.

7.3.4.1 Surface water HQs

HQs were calculated for the surface water COPECs and are presented in Table 7-30. Appendix G provides EPCs, TRVs, and calculated HQs for the surface water COPECs for fish in a single table (Table G5). HQs were ≤ 1.0 for 26 of the 28 COPECs evaluated.[113] EPCs of surface water samples exceeded the TRVs for two COPECs: copper and cyanide. As the BLM-based TRVs for copper (saltwater and freshwater), lead (freshwater), and zinc (freshwater) were sample specific, HQs were calculated on a sample-specific basis, rather than based on a UCL; therefore, a range of estimated HQs are presented in Table 7-30. The distinction between freshwater and saltwater/estuarine was determined by the salinity of each sample. NJDEP (2011b) defines freshwater as having salinity < 3.5 ppth. Thus, stations with salinities < 3.5 ppth were evaluated as freshwater and stations with salinity > 3.5 ppth were evaluated as estuarine in the metal BLMs. The ranges of sample-specific HQs for copper, lead, and zinc are provided in Table 7-30. All HQs for lead and zinc were < 1.0; for copper, HQs ranged from 0.14 to the maximum HQ of 2.7 within the estuarine reach. Sample-specific copper HQs are shown in Figure 7-8.

Table 7-30. Surface water HQs for fish

COPEC	HQ[a]			
	Estuarine (RM 0–RM 13)		Freshwater (RM 4–RM 17.4)	
	Acute	Chronic	Acute	Chronic
Metals				
Cadmium (dissolved)	0.0013	0.0054	0.0018–0.063[b]	0.0048–0.016[b]
Chromium (dissolved)	0.0008	0.018	0.075	0.11
Copper (dissolved)	0.14–**2.7**[c,d]	0.14–**2.7**[c,d]	0.023–0.65[d]	0.037–**1.0**[d]
Lead (dissolved)	0.012	0.12	< 0.001–0.034[d]	0.0063–0.67[d]
Mercury (dissolved)	0.005	0.0096	0.0093	0.0062
Selenium (dissolved)	0.0019	0.0076	na	0.20
Silver (dissolved)	0.0031	0.0085	0.014	0.038
Zinc (dissolved)	0.1	0.39	0.0024–0.051[c,d]	0.017–0.24[c,d]
Butyltins				
TBT	0.062[e]	0.39[e]	not a COPEC[f]	
SVOCs				
BEHP	0.0034	0.017	0.075	0.26

[113] The total number of COPECs includes the TEQs.

Table 7-30. Surface water HQs for fish

COPEC	HQ[a]			
	Estuarine (RM 0–RM 13)		Freshwater (RM 4–RM 17.4)	
	Acute	Chronic	Acute	Chronic
BBP	0.003	0.01	0.01	0.035
PAHs				
Anthracene	< 0.001	0.0011	0.066	0.17
Benzo(a)anthracene	0.086	0.22	0.12	0.3
Benzo(a)pyrene	0.13	0.33	0.047	0.12
Fluoranthene	0.040	0.10	0.012	0.031
Pyrene	0.27	0.68	0.075	0.2
PCBs				
Total PCBs	0.0055	0.16	0.028	0.13
PCB TEQ - fish	< 0.001	< 0.001	< 0.001	< 0.001
PCDDs/PCDFs				
2,3,7,8-TCDD	0.002	0.009	0.026	0.11
PCDD/PCDF TEQ - fish	0.003	0.012	0.027	0.11
Total TEQ - fish[f]	0.002	0.0093	0.027	0.11
Organochlorine Pesticides				
4,4′-DDE	0.001	0.004	0.001	0.001
4,4,′-DDT	0.012	0.022	0.0015	0.0026
Total DDx	0.096	0.17	0.011	0.019
Total chlordane	0.067	0.47	0.0036	0.025
Dieldrin	0.0031	0.013	not a COPEC	
Hexachlorobenzene	< 0.001	< 0.001	< 0.001	< 0.001
Other				
Cyanide	**1.6**	**5.3**	0.21	0.91

Bold identifies HQ ≥ 1.0.

Shaded cells identify HQs ≥1 based on acute or chronic TRVs.

a HQs were based on UCL EPCs presented in Table 7-28 and TRVs presented in Table 7-29, except where noted.

b HQs based on sample-specific, hardness-based TRVs are presented as a range of values (i.e., each individual sample has a corresponding hardness-based TRV and HQ).

c For BLM applications, freshwater TRV was used to calculate HQ if sample-specific salinity was < 3.5 ppth, and estuarine TRV was used to calculate HQ if sample-specific salinity was ≥ 3.5 ppth. The acute and chronic HQs for copper were the same because the TRVs were the same; acute TRVs were determined to be sufficiently predictive of chronic toxicity.

d HQs based on sample-specific, BLM-based TRVs are presented as a range of values (i.e., each individual sample has a corresponding BLM-based TRV and HQ).

e HQ were based on maximum DL (UCL could not be calculated based on low detection frequency).

f TBT was not detected in any freshwater samples and therefore, no HQs were derived.

g The sum of the PCDD/PCDF TEQ - fish and PCB TEQ - fish does not necessarily equal the total TEQ - fish because EPCs are based on the UCL EPCs of each TEQ group.

BBP – butyl benzyl phthalate
BEHP – bis(2-ethylhexyl) phthalate
BLM – biotic ligand model
COPEC – chemical of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
DL – detection limit
EPC – exposure point concentration
HQ – hazard quotient
na – not applicable
nd – no data
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
PCDD– polychlorinated dibenzo-*p*-dioxin
PCDF –polychlorinated dibenzofuran
ppth – parts per thousand
RM – river mile
SVOC – semivolatile organic compound
TBT – tributyltin
TCDD – tetrachlorodibenzo-*p*-dioxin
TEQ – toxic equivalent
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT and 4,4′-DDT)
TRV – toxicity reference value
UCL – upper confidence limit on the mean

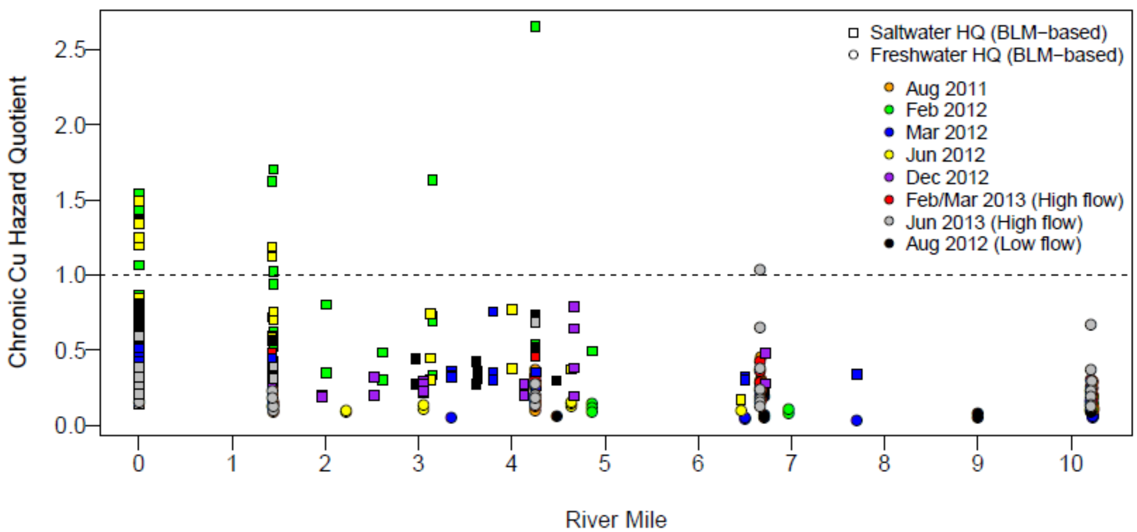


Figure 7-8.Chronic copper BLM-based HQs for individual LPRSA surface water samples

While the risk characterization for cyanide is based upon EPCs for the estuarine and freshwater portions of the LPRSA, Figure 7-9 shows the range of HQs for individual surface water samples throughout the LPRSA, based on the chronic TRVs. The HQs in Figures 7-8 and 7-9 are based on detected concentrations up to RM 10.2, as no surface water samples were collected in that portion of the LPRSA, consistent with the USEPA-approved surface water QAPPs (AECOM 2012c, 2010b, 2012b).

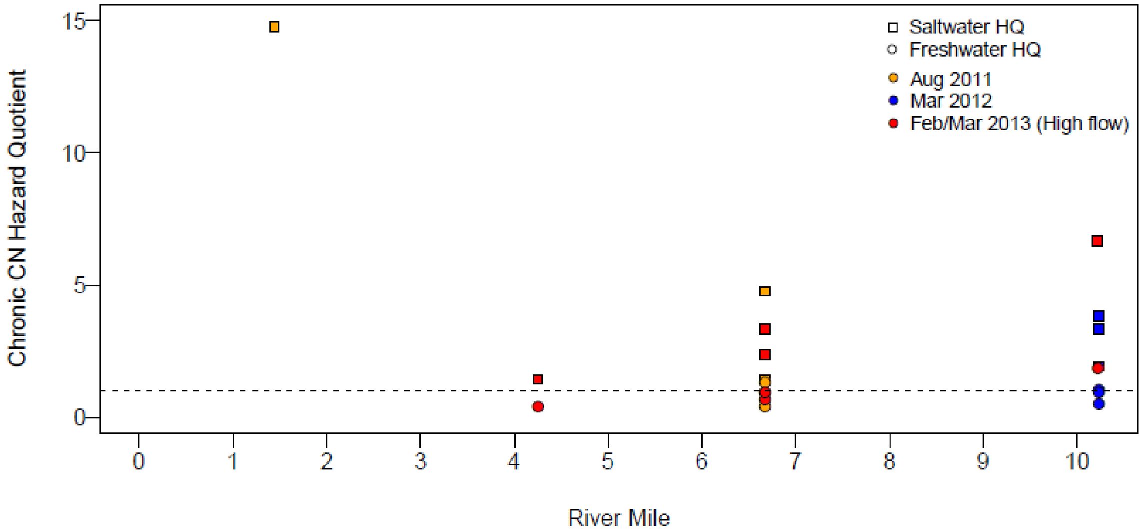


Figure 7-9. Chronic cyanide HQs for individual LPRSA surface water samples

7.3.4.2 Uncertainties in risk characterization

This section discusses uncertainties associated with EPCs that could affect HQ calculations for fish from surface water. General uncertainties associated with the surface water TRVs are discussed in Section 6.2, as well as the limited toxicity data available for TRV derivation for several COPECs and the use of ACRs to derive chronic TRVs. The EPC uncertainties addressed in this section that could be evaluated quantitatively are as follows:

- **Bioavailability in whole-water samples –** Surface water chemistry results for organic chemicals were analyzed in whole-water samples (AECOM 2012c). EPCs for nonionic organic chemicals (e.g., PAHs, PCBs, and organochlorine pesticides) based on total concentrations in whole-water samples may overestimate the fraction of these chemicals that is bioavailable to aquatic organisms. The bioavailability of nonionic organic chemicals is influenced by DOC and particulate organic carbon (POC) present in the water column, concentrations of which determine the fraction of the chemical that is freely dissolved and thus, bioavailable (Burkhard 2000).
- **Representativeness of TRVs –** Some of the selected surface water TRVs may be overly protective of fish, because the TRVs are based on SSDs largely driven by invertebrate species, as described in Table 7-29.
- **Treatment of non-detects for EPCs –** The concentrations of sum components (e.g., PCB congeners) that were not detected were assumed to be zero when calculating totals. The effect on HQs of using one-half the DL or the full DL was evaluated. The treatment of non-detected values in sums (either as zero, one-half the DL, or the full DL) has no effect on the HQ, as shown in Table 7-31.

- **Use of maximum concentrations or DLs as EPCs –** Maximum concentrations were used to represent EPCs for those COPECs (i.e., TBT) that were infrequently detected. This uncertainty was not empirically evaluated because too few detected values were available for the calculation of a UCL; however, due to the low detection frequency of these COPECs, it is unlikely that concentrations pose unacceptable risk to ecological species, regardless of calculated HQs.

Table 7-31. Surface water HQs for fish based on uncertainties in EPCs for total PCBs

Uncertainty	Parameter Values/ Assumptions		Chronic HQs			
			Estuarine		Freshwater	
	Original	Adjusted	Original	Adjusted	Original	Adjusted
Treatment of non-detects	DL = 0 for non-detects	use of one-half the DL or the full DL for non-detects[a]	0.016	0.016	0.028	0.028

Bold identifies HQs ≥ 1.0.

[a] HQs are the same, regardless of treatment of non-detected values as one-half the DL or as the full DL.

DL – detection limit

EPC – exposure point concentration

HQ – hazard quotient

7.3.4.3 Comparison to background

Surface water data from individual samples collected at one background location above Dundee Dam were compared to concentrations in LPRSA surface water from RM 0 to RM 17.4 for the two COPECs with HQs ≥ 1.0. An estuarine background location was not selected for surface water, so LPRSA freshwater locations were compared to the single freshwater location above Dundee Dam. The cumulative frequency of LPRSA freshwater data was ranked relative to the freshwater background data for copper and cyanide, as shown in Figures 7-10 and Figure 7-11, respectively. The freshwater data shown in Figure 7-10 are the LPRSA samples with salinities < 3.5 ppth, which were compared to the freshwater BLM-based TRVs. As shown in Figure 7-10, only one freshwater LPRSA sample had a BLM-based HQ ≥ 1.0; copper concentrations in all other freshwater LPRSA and background samples were below the sample-specific BLM-based TRV. The freshwater data shown in Figure 7-11 are the LPRSA samples collected between RM 4 and RM 17.4, which were compared to the freshwater cyanide TRVs.

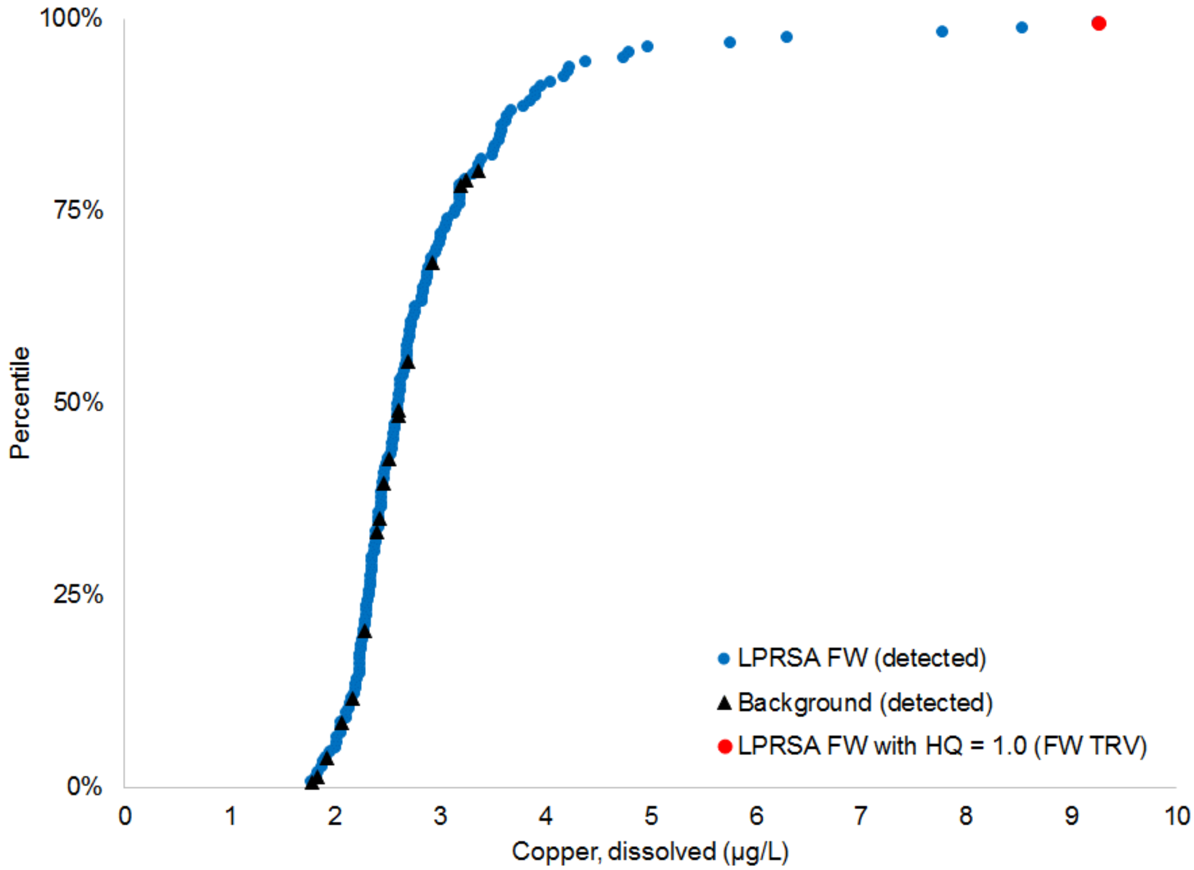


Figure 7-10. Dissolved copper concentrations in freshwater LPRSA and background surface water samples

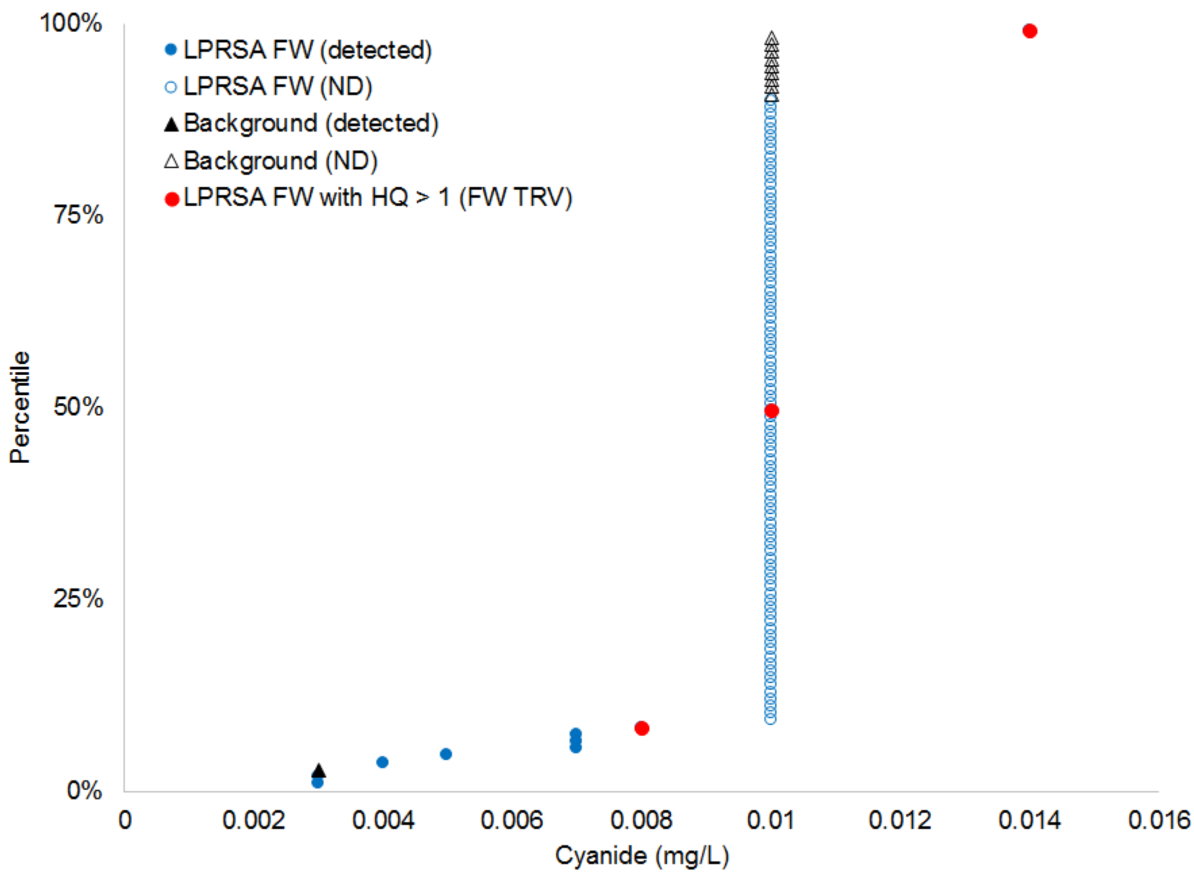


Figure 7-11. Cyanide concentrations in freshwater LPRSA and background surface water samples

Background concentrations of dissolved copper ranged from 1.78 to 3.36 mg/L. Approximately 22% (30 of 139) of all freshwater LPRSA samples (with salinities < 3.5 ppth) were outside the range of background concentrations of dissolved copper (Figure 7-10). Cyanide was detected in one of the background samples at a concentration of 0.003 mg/L. The DL for cyanide was 0.01 mg/L. Only 2 of the 11 LPRSA freshwater samples in which cyanide was detected had concentrations greater than the DL (Figure 7-11).

7.3.5 Summary of uncertainties

The primary uncertainties associated with the surface water risk characterization are the use of EPCs based on whole-water samples rather than the dissolved, bioavailable form for hydrophobic, nonionic organic chemicals; and the use of EPCs based on maximum concentrations where data were infrequently detected. In addition, although the toxicity data are limited (specifically for PAHs), some of the selected surface water TRVs may be overly protective of fish, because the TRVs are largely based on invertebrate toxicity data, and invertebrates generally appear to be more sensitive than fish.

7.3.6 Summary

HQs were < 1.0 for 25 of the 27 COPECs evaluated. Two of the surface water COPECs had HQs ≥ 1.0: copper and cyanide. These COPECs are further evaluated in Section 7.7, where COCs are identified.

Risks from exposure to copper are estimated using the BLM. The copper BLM is a predictive toxicity model that considers the effect of water chemistry characteristics on copper bioavailability. Two versions of the BLM were applied for derivation of copper TRVs, a saltwater BLM and a freshwater BLM. The saltwater BLM was developed to predict copper toxicity to the highly sensitive larval life stage of *M. galloprovincialis* (a bivalve and therefore an overly conservative surrogate for fish). In saltwater, *Mytilus* is the genus most sensitive to copper, and is the basis for the BLM-based, sample-specific TRVs when the salinity of the sample is 3.5 ppth, or greater. The freshwater BLM has been developed for numerous fish and invertebrate species, and is the basis for the freshwater AWQC for copper. Invertebrates are generally the most sensitive organisms to copper; invertebrates represent the 9 most sensitive genera considered in the current freshwater WQC, and 9 of the 10 most sensitive genera considered in the 2003 USEPA saltwater draft WQC update (USEPA 2003a). Therefore, the TRVs are expected to be overly protective of fish. Summer flounder (*Paralichthys dentatus*), with sensitivity similar to that of *M. galloprovincialis*, are the third most sensitive species included in the saltwater acute SSD; however, this species is not closely related to LPRSA species. The LPRSA species winter flounder (*Pseudopleuronectes americanus*) (of the same taxonomic order) are 10-fold less sensitive to copper; therefore, the estuarine TRV is expected to be overly protective of fish in the LPRSA. The freshwater copper BLM was used to derive sample-specific TRVs when the salinity of a sample was < 3.5 ppth. As the freshwater copper TRVs are driven by the sensitivity of invertebrates, potential risks from exposure of fish to copper in both the freshwater and estuarine portions of the river are overestimated.

Cyanide was infrequently detected (i.e., in less than 6% of all LPRSA samples). As the saltwater TRVs for cyanide are based on toxicity data indicating that invertebrates are more sensitive to cyanide than fish, potential risks from exposure of fish to cyanide between RM 0 and RM 13 (i.e., the estuarine portion) could be overestimated.

7.4 Egg Tissue Assessment

Mummichog was the fish species assessed under the modeled egg tissue chemistry evaluation LOE. This LOE is uncertain because it uses modeled, rather than field-collected, egg tissue data. The use of a model to estimate egg tissue concentrations may under- or overestimate concentrations in egg tissue.

7.4.1 COPECs

COPECs for fish egg tissue were identified in the SLERA in cases where the maximum estimated egg concentrations exceeded screening-level TRVs (Section 5). The following fish egg COPECs were identified:

- Methylmercury/mercury [114]
- Total PCBs
- PCDD/PCDF TEQ - fish
- Total TEQ - fish

For these COPECs, exposure-based concentrations (Section 7.4.2) were compared with toxicity-based values (Section 7.4.3) for the derivation of fish egg HQs (Section 7.4.4).

7.4.2 Exposure

Fish egg tissue chemical concentrations for mummichog were estimated using egg-to-adult CFs. The following sections describe CFs for total PCBs, TEQs, and methylmercury/mercury, and present the modeled concentrations based on the CFs.

7.4.2.1 CFs for total PCBs and TEQs

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

$$CF = \frac{C_{egg,lipid}}{C_{adult,lipid}} \qquad \textbf{Equation 7-5}$$

Where:

CF = adult-to-egg conversion factor

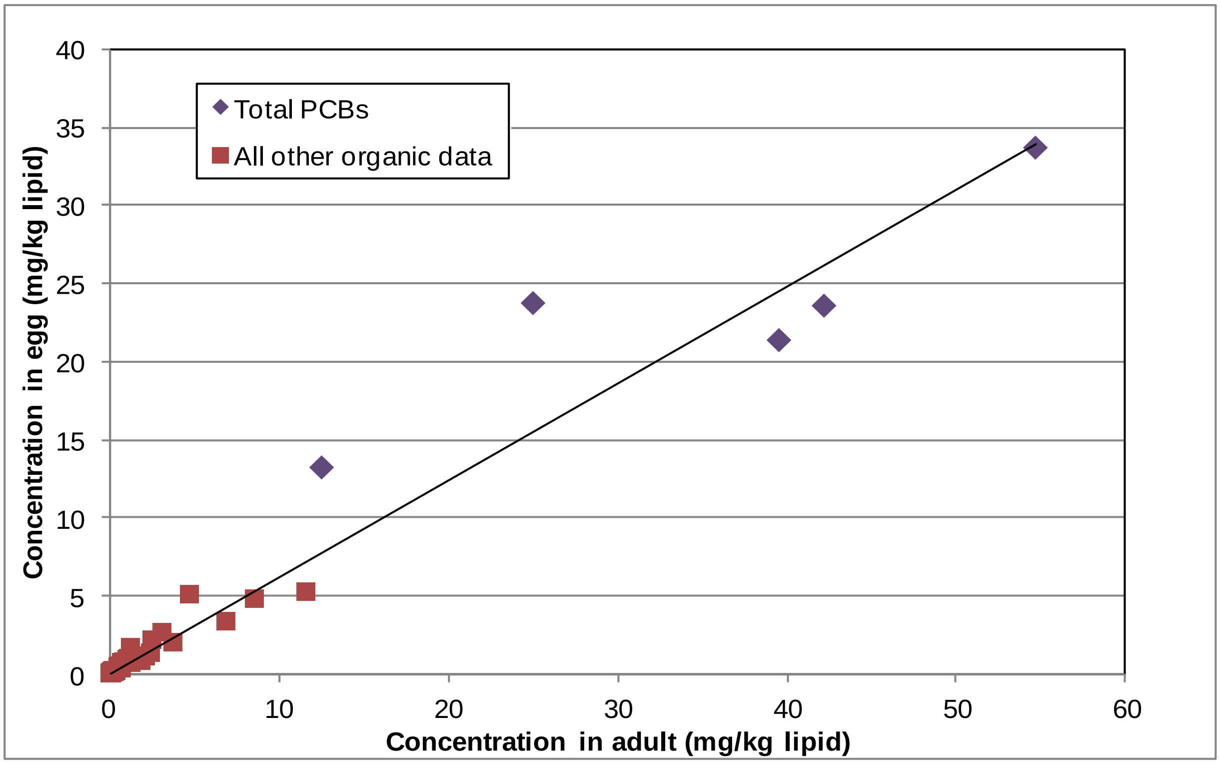
$C_{egg,lipid}$ = lipid-normalized chemical concentration in egg tissue

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

Niimi (1983) conducted a laboratory study examining the relationship between organic chemical (including PCBs and pesticides [alpha-chlordane, DDE, dieldrin, endosulfan, hexachlorobenzene, heptachlor epoxide, lindane, mirex, total chlordane, and DDT]) concentrations in five species of adult female fish captured from Lake Ontario and concentrations in their unfertilized eggs (Niimi 1983). Species evaluated included rainbow trout, white sucker, white bass, smallmouth bass, and yellow perch. Niimi

[114] Total mercury is included as well as methylmercury because some of the TRVs were based on total mercury in tissue. Typically, the majority of total mercury in trophic level fish and invertebrate tissue is in the form of methylmercury. Methylmercury was, on average, 76% of the total mercury in mummichog whole-body tissue collected in 2010.

(1983) found highly significant relationships between organic chemical concentrations in adults and their eggs, with lipids being an important determining factor. Adult and egg concentrations and lipid data from Niimi (1983) were used to derive an adult-to-egg regression relationship. Because lipid content affects the uptake of organic chemical concentrations in biological tissue, the regression was based on mean lipid-normalized adult and egg concentrations ($r^2 = 0.95$, $P < 0.01$; Figure 7-12).



Source: Niimi (1983)

Figure 7-12. Relationship between lipid-normalized organic chemicals in whole bodies of adult fish and their eggs

Based on the data reported in Niimi (1983), the following equation was derived to predict egg tissue concentrations from adult concentrations on a lipid-normalized basis:

$$EPC_{egg,lipid} = \mathbf{0.6213} \times EPC_{adult,lipid}$$ **Equation 7-6**

Where:

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

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

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

Russell et al. (1999) presented data on the relationship between maternal lipid-normalized dorsal muscle tissue concentrations and lipid-normalized egg concentrations for hydrophobic organic chemicals (including 36 individual PCB congeners, 4,4′-DDE, pesticides, and SVOCs) in six fish species (i.e., carp, black crappie [*Pomoxis nigromaculatus*], freshwater drum [*Aplodinotus grunniens*], gizzard shad, quillback [*Carpiodes cyprinus*], and whitefish). The average lipid egg-to-maternal dorsal tissue concentration ratio was 1.22 (95% probability intervals of 0.56 to 2.51) across the six fish species. The authors concluded that the majority of the observed lipid-normalized egg/maternal dorsal tissue concentration ratios for individual chemicals and fish were not significantly different from 1.0. Thus a CF of 1.0 was also used to estimate egg concentrations for total PCBs and TEQ - fish:

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

Where:

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

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

There is uncertainty in assuming muscle tissue concentrations are equivalent to whole-body tissue concentrations.

7.4.2.2 CFs for mercury

For methylmercury/mercury, a robust regression model could not be developed based on the data presented in Niimi (1983). Thus, there is high uncertainty in predicting egg tissue concentrations from whole-body tissue concentrations. Mercury wet weight egg-to-adult CFs from Niimi (1983) ranged from 0.039 to 0.101 (Table 7-32). The maximum CF of 0.101 was used in the following equation to predict egg tissue concentrations from adult concentrations for methylmercury:

$$E_{egg,ww} = \mathbf{0.101} \times C_{adult,ww}$$ **Equation 7-8**

Where:

$C_{egg,ww}$ = chemical concentration in egg tissue (µg/kg ww)

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

Table 7-32. Mercury egg-to-adult fish CFs

Species	Concentration (µg/kg ww)		CF (ww)
	Adult Tissue	Egg Tissue	
Rainbow trout	236	11	0.047
Smallmouth bass	188	8	0.043
White bass	102	4	0.039
White sucker	89	9	0.101
Yellow perch	52	5	0.096
Average			0.065
Maximum			**0.101**

Source: Niimi (1983)
CF – conversion factor
ww – wet weight

In addition, a CF of 1.0 (wherein the egg concentration was assumed to be equal to the whole-body tissue concentration) was evaluated, consistent with 2017 communications between CPG and USEPA.

7.4.2.3 Modeled egg concentrations

The modeled concentrations in mummichog eggs were estimated using adult mummichog whole-body tissue UCL concentrations[115] and are presented in Table 7-33. UCLs for mummichog tissue EPCs were calculated using USEPA's ProUCL® statistical package (Version 5.0.00) (USEPA 2013d) as described in Section 4.3.7.[116]

[115] Fillet and organ-specific samples were not included in UCL calculations.
[116] The UCL recommended by USEPA's ProUCL® software (typically the 95% UCL, but in some cases the 97.5% or even the 99% UCL) was used. The selected UCL statistic is presented in Appendix C1.

Table 7-33. Modeled LPRSA mummichog egg tissue concentrations

COPEC	EPC$_{adult}$		Modeled Egg Tissue Concentration					
			CFs Based on Niimi (1983)			CFs Based on Russell et al. (1999)		
	EPC$_{adult}$[a] (µg/kg-ww)	EPC$_{adult}$[a] (mg/kg-lipid)	CF	EPC$_{egg}$ (mg/kg-lipid)	EPC$_{egg}$ (µg/kg ww)	CF	EPC$_{egg}$ (mg/kg-lipid)	EPC$_{egg}$ (µg/kg ww)
Metals								
Mercury	63	na	0.101 (ww:ww)	na	6.4[b]	1.0 (ww:ww)	na	63[c]
Methylmercury	53	na	0.101 (ww:ww)	na	5.4[b]	1.0 (ww:ww)	na	53[c]
PCBs								
Total PCBs	600	28	0.6213 (lipid:lipid)	17[d]	574[e]	1.0 (lipid:lipid)	28[f]	924[e]
PCCD/PCDF								
PCDD/PCDF TEQ - fish	0.051	0.0022	0.6213 (lipid:lipid)	0.0014[d]	0.045[e]	1.0 (lipid:lipid)	0.0022[f]	0.073[e]
Total TEQ - fish	0.051	0.0022	0.6213 (lipid:lipid)	0.0014[d]	0.045[e]	1.0 (lipid:lipid)	0.0022[f]	0.073[e]

a Based on UCL mummichog whole-body tissue concentrations from LPRSA samples.

b EPC$_{egg}$ was estimated using Equation 7-8.

c EPC$_{egg}$ was assumed equal to EPC$_{adult}$.

d EPC$_{egg}$ was estimated using Equation 7-6

e Wet weight egg concentration was estimated from lipid-normalized value based on the UCL for percent lipids in LPRSA mummichog egg samples (3.3%).

f EPC$_{egg}$ was estimated using Equation 7-7.

COPEC – chemical of potential ecological concern
EPC – exposure point concentration
LPRSA – Lower Passaic River Study Area
na – not applicable
PCB – polychlorinated biphenyl
PCDD– pentachlorodibenzo-*p*-dioxin
PCDF – polychlorinated dibenzofuran
TEQ – toxic equivalent
UCL – upper confidence limit on the mean
ww – wet weight

7.4.3 Effects

This section presents the effects data (i.e., TRVs) selected from the toxicological literature for the COPECs and species that were screened for this BERA based on the SLERA. A range of TRVs was evaluated, including TRVs developed by USEPA Region 2 for the LPRSA and those based on literature. The following subsections describe the general methods used to identify TRVs. Selected TRVs are consistent with the comments, responses, and directives received from USEPA on June 30, 2017 (USEPA 2017b), September 18, 2017 (USEPA 2017e), July 10, 2018 (USEPA 2018), January 2, 2019 (CDM 2019), and March 5, 2019 (USEPA 2019); during face-to-face meetings or conference calls on July 24, September 27, October 3, November 6, 2017, July 24, 2018, and August 16, 2018; and via additional deliverables and communications between the Cooperating Parties Group (CPG) and USEPA from August through December 2017, July through September 2018, and January through June 2019.

7.4.3.1 Methods for selecting TRVs

Two sets of fish egg tissue TRVs were used for the derivation of HQs in this BERA. One set was based on previous documents developed by USEPA Region 2 for the LPRSA:

- USEPA's revised draft of the LPR restoration project FFS (Louis Berger et al. 2014)
- USEPA's first draft of the LPR restoration project FFS (Malcolm Pirnie 2007b)

The second set of TRVs was developed—in the same manner as whole-body tissue TRVs were developed (Section 7.1.3.1)—to evaluate potential effects on early life stages of mummichog.

7.4.3.2 Selected TRVs for fish egg tissue

TRVs are presented in Table 7-34. These TRVs are described in detail in the sections below for each COPEC, and toxicity data used to select TRVs are presented in Appendix E.

Table 7-34. Fish egg tissue TRVs

COPEC	Units (ww)	Range of TRVs[a]								
		TRV-A[b]				TRV-B[c]				
		NOAEL	LOAEL	Endpoint	Source	NOAEL	LOAEL	Endpoint	Source	Document
Metals										
Methylmercury/ mercury	µg/kg	6.0[d]	60	reproduction (catfish)	Birge et al. (1979)	0.006[d]	0.060	reproduction (catfish)	Birge et al. (1979)	draft FFS (Battelle 2007)
PCBs										
Total PCBs	µg/kg	25.8[d]	258	reproduction (common barbels)	Hugla and Thome (1999)	5.04[d]	50.4	reproduction (common barbels)	Hugla and Thome (1999)	USEPA draft BERA comments (USEPA 2015c)
PCDD/PCDFs										
PCDD/PCDF TEQ - fish Total TEQ - fish	ng/kg	7.2	86	growth, survival, reproduction, and behavior (10 species)	Steevens et al. (2005)	7.2	86	growth, survival, reproduction, and behavior (10 species)	Steevens et al. (2005)	revised FFS (Louis Berger et al. 2014)

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

b TRVs were derived from the primary literature based on the process identified in Section 7.4.3.1.

c TRVs were based on USEPA's revised draft of the LPR restoration project FFS (Louis Berger et al. 2014) or USEPA's first draft of the LPR restoration project FFS (Malcolm Pirnie 2007b), as described in Section 7.4.3.1.

d NOAEL extrapolated from LOAEL using an uncertainty factor of 10.

BERA – baseline ecological risk assessment
COPEC – chemical of potential ecological concern
FFS – focused feasibility study
LOAEL – lowest-observed-adverse-effect level
LPR – Lower Passaic River
LPRSA – Lower Passaic River study Area
NJDEP – New Jersey Department of Environmental Protection
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
PCDD– polychlorinated dibenzo-*p*-dioxin
PCDF –polychlorinated dibenzofuran
TEQ – toxic equivalent
TRV – toxicity reference value
USEPA – US Environmental Protection Agency
ww – wet weight

Methylmercury

Three acceptable toxicity studies were identified that evaluated effects of methylmercury on reproduction, behavior, physiology, and survival. Three LOAELs were reported for three fish species (catfish [*Siluriformes*], grayling [*Thymallus thymallus*], and Japanese medaka), ranging from 60 to 29,000 µg/kg ww. Birge et al. (1979) reported the lowest LOAEL of 0.06 mg/kg ww, which was selected as the LOAEL TRV. This LOAEL was determined based on a reported LC50 value from water exposure (0.3 µg/L) to inorganic mercury, which was associated with a catfish egg concentration of 0.060 mg/kg ww (48% survival was observed at hatching and 30% survival at four days post-hatching) (Birge et al. 1979). No NOAEL was reported in this study, so a NOAEL of 0.00060 mg/kg ww was extrapolated from the LOAEL using a factor of 10. There is uncertainty associated with the use of an extrapolation factor to derive the TRV. In addition, there is uncertainty associated with the selected TRVs due to the limited number of studies evaluated (three studies), and because the selected LOAEL is based on a severe effect (a high reduction in survival).

These TRVs were also selected as the NOAEL and LOAEL (Battelle 2007) based on the data reported by Birge et al. (1979) and the use of an extrapolation factor to determine a NOAEL.

Total PCBs

Five acceptable toxicity studies were identified that evaluated the effects of PCB egg tissue concentrations on reproduction and growth. Four LOAELs were reported for four species (Atlantic croaker, brook trout, rainbow trout, and common barbell [*Barbus barbus*]), so data, which ranged from 258 to 77,900 µg/kg/egg, were insufficient for the derivation of an SSD-based TRV (Figure 7-13). Additionally, one NOAEL of 22 µg/kg/egg was reported for Japanese medaka. The lowest LOAEL of 258 µg/kg ww (estimated from the reported concentration of 1,289 mg/kg dw, assuming 80% moisture in tissue) was reported by Hugla and Thome (1999) and was selected as the LOAEL TRV. Reduced hatchability was observed in common barbels fed 12.5 mg/kg PCBs for 75 days (dose associated with LOAEL TRV). The NOAEL TRV (25.8 µg/kg ww) was extrapolated from the LOAEL TRV using an uncertainty factor of 10. There is uncertainty associated with the use of an extrapolation factor to derive the TRV.

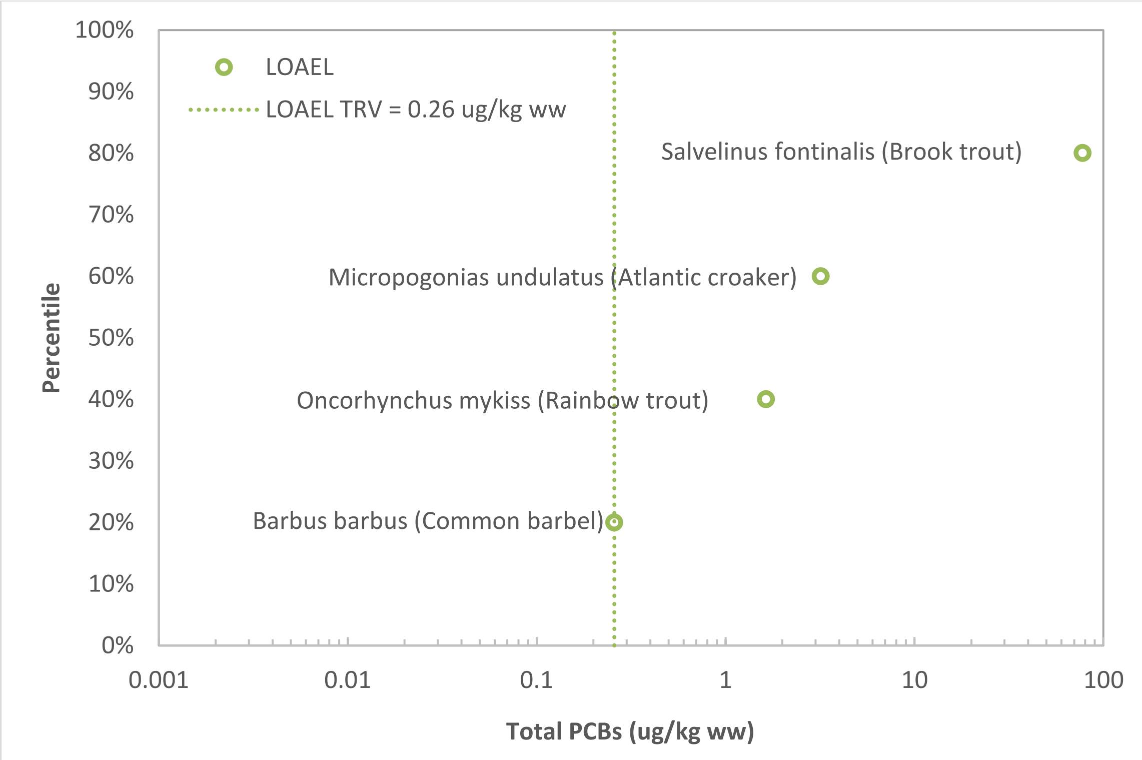


Figure 7-13. Fish egg tissue total PCB toxicity data

A LOAEL of 50.4 µg/kg ww was selected as the LOAEL (USEPA 2015c) based on the same study (Hugla and Thome 1999). Fecundity (i.e., number of eggs per female) was significantly reduced in adult fish fed a lower dose of PCBs (2.5 mg/kg PCBs for 50 days); however, there was no significant effect on egg weight or hatching rate at this dose. Egg tissue concentrations were reported for this dose group (50.4 µg/kg), but appeared to represent concentrations 1 to 2 years after initial exposure to PCBs (and fecundity effect). Thus, the relationship between the selected LOAEL of 50.4 µg/kg and the adverse effects at this dose is uncertain. The USEPA-selected NOAEL of 5.04 µg/kg ww was estimated by dividing the LOAEL by 10. There is uncertainty associated with the use of an extrapolation factor to derive the TRV.

PCDDs/PCDFs - Fish

Nine available toxicity studies were identified that evaluated the effects of PCDD/PCDF (as 2,3,7,8-TCDD) egg tissue concentrations on reproduction. Eighteen LOAELs were reported for 11 fish species (brook trout, channel catfish, fathead minnow, mummichog, lake herring [*Coregonus artedi*], lake trout [*Salvelinus namaycush*], Japanese medaka, northern pike, rainbow trout, white sucker, and zebrafish). These studies reported LOAELs ranging from 0.76 to 2,000 ng/kg ww; LOAELs were based on early life stage survival, growth, and reproduction (i.e., hatchability). Also available from the literature review was Steevens et al. (2005), wherein 5th percentile SSD lower confidence limit (LCL) and UCL values of 7.2 and 86 ng/kg ww, respectively, were calculated based on lipid-normalized reported results (of 1.05 and 0.088 ng/kg-lipids, respectively) and assuming a lipid value of 8.2%. The SSD was based on TRVs from studies that considered the life stage most sensitive to toxicity, and included 10 fish species: brook trout, channel catfish, fathead minnow, Japanese medaka, lake herring, lake trout, northern pike, rainbow trout, white sucker, and zebrafish. The 5th percentile LCL (7.2 ng/kg ww) and UCL (86 ng/kg ww) derived by Steevens et al. (2005) were selected as the NOAEL and LOAEL, respectively. This LOAEL may be overly conservative for mummichog, in that the species-specific LOAEL for mummichog (635 ng/kg ww) (Prince and Cooper 1995) is approximately one order of magnitude greater than the selected LOAEL (86 ng/kg ww).

The LCL and UCL values derived from Steevens et al. (2005) were also selected as the NOAEL and LOAEL (Louis Berger et al. 2014), respectively, for TEQs - fish in fish egg tissue.

7.4.4 Risk characterization

This section presents the HQs calculated for LPRSA fish egg tissue, followed by a comparison of the LPRSA HQs to those calculated for background areas.

7.4.4.1 Egg tissue HQs

HQs based on modeled fish egg concentrations presented in Table 7-33 and fish egg TRVs presented in Table 7-34 were calculated for the fish egg COPECs and are presented in Table 7-35. Appendix G provides EPCs, CFs, TRVs, and calculated HQs for the fish egg tissue COPECs in a single table (Table G6). LOAEL HQs were greater than or equal to 1.0 for total PCBs and mercury (based on a CF of 1.0 only), and NOAEL HQs were greater than or equal to 1.0 for mercury, methylmercury (based on a CF of 1.0 only), total PCBs, PCDD/PCDF TEQ - fish, and total TEQ - fish.

Table 7-35. Fish egg tissue HQs

COPEC	Mummichog Range of HQs[a]			
	CFs based on Niimi (1983)		CFs based on Russell et al. (1999)	
	HQ based on TRV-A[b]	HQ based on TRV-B[c]	HQ based on TRV-A[b]	HQ based on TRV-B[c]
LOAEL HQ				
Metals				
Mercury	0.11	0.11	**1.1**	**1.1**
Methylmercury	0.089	0.089	0.88	0.88
PCBs				
Total PCBs	**2.2**	**11**	**3.6**	**18**
PCDD/PCDF				
PCDD/PCDF TEQ - fish	0.52	0.52	0.84	0.84
Total TEQ - fish	0.52	0.52	0.84	0.84
NOAEL HQ				
Metals				
Mercury	**1.1**	**1.1**	**11**	**11**
Methylmercury	0.89	0.89	**8.8**	**8.8**
PCBs				
Total PCBs	**22**	**114**	**36**	**183**
PCDD/PCDF				
PCDD/PCDF TEQ - fish	**6.3**	**6.3**	**10**	**10**
Total TEQ - fish	**6.3**	**6.3**	**10**	**10**

Bold identifies HQs ≥ 1.0.

Shaded cells identify HQs ≥1 based on LOAEL TRVs.

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

b TRVs were derived from the primary literature based on the process identified in Section 7.4.3.1.

c TRVs were based on USEPA's revised draft of the LPR restoration project FFS (Louis Berger et al. 2014) or first draft of the LPR restoration project FFS (Malcolm Pirnie 2007b).

BERA – baseline ecological risk assessment
CF – conversion factor
COPEC – chemical of potential ecological concern
FFS – focused feasibility study
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
LPR – Lower Passaic River
LPRSA – Lower Passaic River study Area
NJDEP – New Jersey Department of Environmental Protection
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
PCDD– polychlorinated dibenzo-p-dioxin
PCDF –pentachlorodibenzofuran
TEQ – toxic equivalent
TRV – toxicity reference value
USEPA – US Environmental Protection Agency

7.4.4.2 Uncertainties in risk characterization

This section discusses uncertainties associated with exposure assumptions, EPCs, and selected TRVs that could affect HQ calculations for fish eggs. Uncertainties associated with the TEQ methodology are presented in Section 4.1 and general TRV uncertainties are discussed in Section 7.1.3.1. The uncertainty associated with modeled egg concentrations can be evaluated by considering the range of CFs evaluated. The use of CFs based on Niimi (1983) vs. Russell et al. (1999) slightly changes the HQs for PCBs and TEQ - fish COPECs, although it does not change whether or not an HQ is above or below 1.0. The use of a CF of 1.0 for mercury, however, does result in a LOAEL HQ < 1 (1.1 for total mercury but 0.88 for methylmercury), whereas HQs based on CFs from data reported in Niimi (1983) are less than 1.0. The use of a CF of 1.0 for mercury is highly uncertain; based on the data from Niimi (1983), it over-predicts mercury accumulation in eggs by a factor of 10 for the 5 species evaluated (rainbow trout, smallmouth bass, white bass, white sucker, and yellow perch). Based on these data, it may also over-predict potential risks to mummichog.

The uncertainty quantitatively addressed in the remainder of this section is as follows:

- **Treatment of non-detects for EPCs –** The concentrations of congeners that were not detected were assumed to be zero when calculating total PCBs and TEQs. The effect on HQs of using one-half the DL or the full DL was evaluated.

The effect of this uncertainty on LOAEL HQ calculations is presented in Table 7-36. The treatment of non-detected values in sums (either as zero, one-half the DL, or the full DL) has no effect on the HQ.

Table 7-36. Fish mummichog egg HQs based on uncertainties in EPCs and TRVs

Uncertainty	Parameter Values/Assumptions		Total PCBs LOAEL HQ	
	Original	Adjusted	Original HQ Based on TRV-A[a]	Adjusted HQ Based on TRV-A[a]
Treatment of non-detects	DL = 0 for non-detects	DL = one-half the DL or full DL for non-detects[b]	**2.2**	**2.2**

Bold identifies HQs ≥ 1.0.

a TRVs were derived from the primary literature based on the process identified in Section 7.4.3.1.

b LOAEL HQs are the same, regardless of treatment of non-detected values as one-half the DL or as the full DL.

CPG – Cooperating Parties Group
DL – detection limit
EPC – exposure point concentration
HQ – hazard quotient
PCB – polychlorinated biphenyl
LOAEL – lowest-observed-adverse-effect level
TRV – toxicity reference value
USEPA – US Environmental Protection Agency

7.4.4.3 Comparison to background

As described in Section 7.1.4.3, EPCs from the LPRSA were compared to background concentrations for fish species-COPEC pairs with LOAEL HQs ≥ 1.0. Three background datasets were developed for use in this BERA using available data from the following areas: 1) upstream of Dundee Dam, to represent freshwater urban habitat, 2) Jamaica Bay/Lower Harbor, to represent estuarine urban habitat, and 3) Mullica River/Great Bay, to represent estuarine/freshwater rural habitat. These datasets are summarized in Section 4.2, and details on how background values were determined from these datasets are presented in Appendix J. Background values are presented in Table 7-37.

Table 7-37. Comparison of LPRSA fish egg tissue EPCs with background

Species	Units	Modeled Mummichog Egg Concentration					LOAEL TRV-A[d]	LOAEL TRV-B[e]
		LPRSA EPC[a]	Above Dundee Dam[b]		Jamaica Bay/ Lower Harbor[c]			
			UCL	Max. Detect	UCL	Max. Detect		
CFs based on Niimi (1983)								
Methylmercury	µg/kg ww	6.4	na	4.1	6.5	7.7	6	60
Mercury	µg/kg ww	5.4	na	3.5	na	7.2	6	60
Total PCBs	µg/kg ww[f]	574	na	145	1,080	1,820	258	50.4
Total PCBs	mg/kg-lipid	17	na	4.4	33	55	na	na
CFs based on Russell et al. (1999)								
Methylmercury	µg/kg ww	63	na	40	64	77	6	60
Mercury	µg/kg ww	53	na	35	na	71	6	60
Total PCBs	µg/kg ww[g]	924	na	361	1,740	2,930	258	50.4
Total PCBs	mg/kg-lipid	28	na	11	53	89	na	na

Note: The maximum detected concentration for background areas exclude outlier concentrations as described in Appendix J.

a Based on UCL concentration of mummichog (n = 26) collected from the LPRSA.

b Background value derived from one banded killifish sample collected from above Dundee Dam (see Appendix J for details on background datasets).

c Background value derived from mummichog (n = 7) collected from Jamaica Bay/Lower Harbor (see Appendix J for details on background datasets).

d TRVs were derived from the primary literature review.

e TRVs were based on USEPA's revised draft of the LPR restoration project FFS (Louis Berger et al. 2014) or first draft of the LPR restoration project FFS (Malcolm Pirnie 2007b).

f Adult wet weight concentration was converted to adult lipid concentration assuming 3.1% lipids (single lipid value from killifish collected above Dundee Dam). Egg lipid concentration was converted to egg wet weight concentration assuming 3.3% lipids (UCL lipid percent from LPRSA mummichog eggs).

g Adult wet weight concentration was converted to adult lipid concentration assuming 3.6% lipids (UCL lipid percent from 12 mummichog included in Jamaica Bay/Lower Harbor regional estuarine dataset). Egg lipid concentration was converted to egg wet weight concentration assuming 3.3% lipids (UCL lipid percent from LPRSA mummichog eggs).

CF – conversion factor
EPC – exposure point concentration
FFS – focused feasibility study
LOAEL – lowest-observed-adverse-effect level
LPR – Lower Passaic River Study Area
LPRSA – Lower Passaic River Study Area
na – not applicable
PCB – polychlorinated biphenyl
TRV – toxicity reference value
UCL – upper confidence limit on the mean
USEPA – US Environmental Protection agency
ww – wet weight

Modeled mummichog egg concentrations for mercury and total PCBs from above Dundee Dam were less than LPRSA mummichog egg concentrations; however, modeled mummichog egg concentrations for mercury and total PCBs from Jamaica Bay/Lower Harbor (using UCLs and maximum detected concentrations) were greater than LPRSA mummichog egg concentrations (using UCL-based EPCs). Based on these data, concentrations of mercury and total PCBs in small fish appear to be greater in Jamaica Bay/Lower Harbor than in the LPRSA. Differences in average lipid content between sites were small (2.0 and 3.1% for LPRSA and Mullica River/Great Bay, respectively) and so are not the cause of the observed total PCB concentration differences in mummichog. Although the higher mean mummichog lipid content in Jamaica Bay/Lower Harbor could indicate better fish condition, there are other factors that may affect lipid content in fish, such as size, age, sex, reproductive status, genetic background, diet, water temperature, and seasonality (Mraz 2012; Iverson et al. 2002). It is also not known whether this small dataset is representative of concentrations in other small fish present but not collected in these areas.

A field study compared the reproductive capacity of mummichog in Newark Bay, New Jersey (below the LPRSA), to mummichog in a reference location (Great Bay, Tuckerton, New Jersey), using several reproductive metrics, including male and female gonad histology; vitellogenin production; and bile chemistry for specific low-, medium-, and high-molecular-weight PAHs. Examinations of male and female gonad histology show decreased gonadal weight, altered testis morphology in males, and altered gonad development in females. Altered female gonad development was indicated in female Newark Bay mummichog by increased pre-vitellogenic follicles (43% at reference location, 64% at Newark Bay), decreased mid-vitellogenic follicles (22% at reference location, 17% at Newark Bay), and decreased mature stage follicles (25% at reference location, 3% at Newark Bay). Overall, Bugel et al. (2010) concluded that Newark Bay mummichog displayed signs of endocrine disruption and decreased reproductive capacity, despite a lack of significant differences in body size or weight. The specific causes and implications of these histological and biomarker effects on overall reproductive success are unknown.

7.4.5 Summary of uncertainty

The greatest uncertainty in this assessment is the use of an adult-to-egg regression from the literature to estimate mummichog egg concentrations. The use of a bioconcentration factor to estimate mercury concentrations in eggs is uncertain, given that no regression could be derived from the literature; however, the use of a CF of 1.0 is not supported by empirical data and likely over-predicts egg tissue concentrations (and HQs). It is also unknown whether the adult-to-egg regression based on organic tissue concentration data (i.e., PCBs and TEQs - fish) over- or under-predicts LPRSA mummichog egg concentrations (and risks); however, similar risks results occur whether CFs from Niimi (1983) or Russell et al. (1999) are used.

7.4.6 Summary

LOAEL HQs for the fish egg LOE were ≥ 1.0 (ranging from 2.2 to 18) for total PCBs. LOAEL HQs for mercury ranged from 0.11 to 1.1; however, methylmercury HQs were < 1.0 (0.089–0.88). Risks to mummichog using estimated egg tissue concentrations from a literature-based adult-to-egg model may over- or underestimate actual fish egg concentrations. A summary of the fish egg tissue LOAEL HQs is presented in Table 7-38.

Table 7-38. Summary of fish egg tissue LOAEL HQs

COPEC[b]	Range of LOAEL HQs[a]		Key Uncertainties
	HQ Based on TRV-A[c]	HQ Based on TRV-B[d]	
	LOAEL EF Values ≥ 1.0		
Mercury (methylmercury)	0.11–**1.1** (0.089–088)	0.11–**1.1** (0.089–088)	• Range of HQs reflects range of CFs: low-end HQs based on mummichog egg concentration modeled using maximum CF reported in literature reviewed, high-end HQs based on mummichog egg concentrations assumed equal to whole-body concentrations
Total PCBs	**2.2–3.6**	**11–18**	• TRV-A and TRV-B based on same literature source; TRV-A based on observed adverse effect on reproduction (reduced hatchability); TRV-B based on reduced fecundity, but no effect on egg weight or hatchability • Mummichog egg concentration modeled using literature-based CFs and LPRSA mummichog-specific lipid content

Bold identifies HQs ≥ 1.0.

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

b Only COPECs with HQs ≥ 1.0 based on a LOAEL TRV are included in the table.

c TRVs were derived from the primary literature review based on the process identified in Section 7.4.3.1.

d TRVs were based on USEPA's revised draft of the LPR restoration project FFS (Louis Berger et al. 2014) or first draft of the LPR restoration project FFS (Malcolm Pirnie 2007b).

BERA – baseline ecological risk assessment
CF – conversion factor
COPEC – chemical of potential ecological concern
EF – exceedance factor
FFS – focused feasibility study
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
LPR – Lower Passaic River
LPRSA – Lower Passaic River Study Area
NJDEP – New Jersey Department of Environmental Protection
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
TRV – toxicity reference value
USEPA – US Environmental Protection Agency

7.5 MUMMICHOG EGG ASSESSMENT

This section evaluates the egg productivity measurements conducted on five female mummichog collected from the LPRSA in May 2010 (Windward 2011c). Measurements of body-normalized egg counts and egg weights (Table 7-39) were compared to data from the scientific literature to determine if egg production was less than observed in other studies, thus indicating a potential adverse reproductive effect on mummichog. Three studies were found with data on mummichog egg production: two laboratory studies (Bosker et al. 2010; Gutjahr-Gobell 1998), and one observational field study using a northeastern Florida salt marsh (Hsiao et al. 1994).

Table 7-39. Estimated egg counts and mass for LPRSA mummichog

Sample ID (individual fish)	LPRSA RM Segments	Mass (g ww)		Estimated Egg Count	Egg Count Normalized for Body Weight (eggs/g bw)	Egg Weight Normalized for Body Weight (g egg/g bw)
		Adult	Total Egg			
LPR2DD-FH057	RM 2 to RM 4	4	1.5	142	35.5	0.38
LPR2II-FH106		3.5	1.0	154	44	0.29
LPR2II-FH109		3	1.0	139	46.3	0.33
LPR3AA-FH110	RM 4 to RM 6	16	2.5	428	26.8	0.16
LPR4CC-FH093	RM 6 to RM 8	5	1.0	270	54	0.20
Average		**6.3**	**1.4**	**227**	**41**	**0.27**

Source: Hsiao et al. (1994)

bw – body weight
ID – identification
LPRSA – Lower Passaic River Study Area
RM – river mile
ww – wet weight

In a laboratory study that investigated the effects of pulp mill effluent on adult mummichog reproduction, Bosker et al. (2010) estimated the weights of eggs normalized for body weight in 24 individuals in the control group every 3 days during the 21-day study. Average egg weights ranged from 0.2 to 0.5 g egg/g bw over the five sampling periods, with an overall average weight of 0.38 g egg/g bw, whereas the average for LPRSA mummichog was 0.27 g egg/g bw (Table 7-39). There is some uncertainty in comparing egg weights from the LPRSA to egg weights from laboratory control fish, and it is uncertain whether potentially reduced egg weights would affect the mummichog population.

Another laboratory study evaluated the effect of diet on egg production in mummichog (Gutjahr-Gobell 1998). Data on the number of eggs normalized to female body weight were recorded for six different diets. For a diet of brine shrimp nauplii, which is most consistent with the natural diet of mummichog, the average normalized egg count was 8 eggs/g bw; the egg counts for all other diets were less, ranging from 1 to 7 eggs/g bw. The highest and average egg counts from this study of 8 and 4.3 eggs/g bw, respectively, were substantially less than the average of 41 eggs/g bw for LPRSA mummichog (Table 7-39). In the laboratory study, eggs were counted after being collected from egg mats at the bottom of the aquaria, whereas eggs from LPRSA mummichog were stripped from the females, which may account for the greater number of eggs per body weight for LPRSA fish.

The field study in a northeastern Florida salt marsh recorded egg production for mummichog from January through October (Hsiao et al. 1994). The peak number of eggs normalized for body weight in May was 16 eggs/g bw, while the maximum number of eggs observed during the entire study was 19 eggs/g bw. The average number of eggs for the five females from the LPRSA, 41 eggs/g bw (Table 7-39), was more than twice the highest value observed by Hsiao et al. (1994).

In summary, weights of LPRSA mummichog eggs (0.27 g egg/g bw) were similar to but slightly less than those measured in the laboratory (0.38 g egg/g bw). The average egg count for LPRSA mummichog (41 eggs/g bw) was greater than the maximum egg counts from the laboratory study (8 eggs/g bw) (Gutjahr-Gobell 1998) and the field study in a Florida salt marsh (19 eggs/g bw) Hsiao et al. (1994). These data show that egg weights and counts were within the range of other studies and did not indicate adverse reproductive effects that could affect the mummichog population based on this LOE alone. However, data addressing egg fertility would be needed as an additional LOE for a more complete evaluation of overall reproductive success and potential effects on the mummichog population.

7.6 Health Assessment

In accordance with the PFD (Windward and AECOM 2009), fish health observation data collected in the field—from both the LPRSA in 2009 and 2010 (Windward 2010c, 2011c) and the freshwater reference area above Dundee Dam in 2012 (Windward 2019c)—were evaluated. Gross external and internal health observations were made for field-collected fish to help provide general information regarding the overall health of the fish; however, the health assessment data are largely qualitative.

7.6.1 Field health observation results

Fish collected from the LPRSA in 2009 and 2010 (83 fish comprising 23 species in 2009 and 36 fish comprising 15 species in 2010; a total of 119 fish comprising 35 species) and from above Dundee Dam in 2012 (46 fish comprising 10 species) were examined visually for gross external and internal abnormalities, based on the data collection procedures outlined by the USGS Biomonitoring of Environmental Status and Trends (BEST) protocol (Schmitt and Dethloff 2000). The USGS BEST protocol for examining fish provides generic identification parameters for observable conditions, but does not include specific diagnoses of fish health (USGS 2002). A formal diagnosis can be determined only through histopathology and other laboratory expertise. The assessment procedure is largely subjective, and decisions about discoloration, organ normality, and damage resulting from the fishing method are based on best professional judgment.

Table 7-40 presents a summary of the total gross abnormalities observed for all fish assessed within the LPRSA and freshwater reference area above Dundee Dam. In the LPRSA, 24% of fish assessed had external non-fin-related gross abnormalities, and 21%

had internal gross abnormalities. Above Dundee Dam, 15% of fish assessed had external non-fin-related gross abnormalities, and 28% had internal gross abnormalities. Fin-related abnormalities were not included because such abnormalities may have been the result of incidental effects from catch methods and/or fish handling.

Table 7-40. Summary of total gross abnormalities for all fish assessed from LPRSA and above Dundee Dam

Common Name	No. of Fish		External Abnormalities[a]				Internal Abnormalities			
			No. of Fish		% of Fish		No. of Fish		% of Fish	
	LPRSA	Above Dundee Dam	LPRSA	Above Dundee Dam	LPRSA	Above Dundee Dam	LPRSA	Above Dundee Dam	LPRSA	Above Dundee Dam
American eel	5	na	1	na	20%	na	5	na	100%	na
Atlantic croaker	1	na	0	na	0%	na	0	na	0%	na
Atlantic menhaden	5	na	2	na	40%	na	2	na	40%	na
Atlantic silverside	5	na	0	na	0%	na	0	na	0%	na
Banded killifish	2	na	1	na	50%	na	0	na	0%	na
Bay anchovy	3	na	1	na	33%	na	0	na	0%	na
Black crappie	3	na	0	na	0%	na	2	na	67%	na
Bluefish	5	na	0	na	0%	na	0	na	0%	na
Bluegill	5	5	1	1	20%	20%	0	1	0%	20%
Brown bullhead	3	5	1	5	33%	100%	2	0	67%	0%
Channel catfish	2	na	1	na	50%	na	0	na	0%	na
Common carp	7	5	3	0	43%	0%	1	1	14%	20%
Crevalle jack	1	na	0	na	0%	na	0	na	0%	na
Gizzard shad	5	na	0	na	0%	na	0	na	0%	na
Goby (unspecified)	5	na	1	na	20%	na	0	na	0%	na
Hogchoker	1	na	0	na	0%	na	0	na	0%	na
Inland silverside	2	na	0	na	0%	na	0	na	0%	na
Northern pike	na	5	na	0	na	0%	na	1	na	20%
Northern searobin	1	na	0	na	0%	na	0	na	0%	na
Pumpkinseed	5	5	0	0	0%	0%	0	4	0%	80%
Redbreast sunfish	5	5	1	0	20%	0%	1	1	20%	20%
Redfin pickerel	1	na	0	na	0%	na	0	na	0%	na
Rock bass	1	5	0	0	0%	0%	0	2	0%	40%
Satinfin shiner	3	na	0	na	0%	na	0	na	0%	na

Table 7-40. Summary of total gross abnormalities for all fish assessed from LPRSA and above Dundee Dam

Common Name	No. of Fish		External Abnormalities[a]				Internal Abnormalities			
			No. of Fish		% of Fish		No. of Fish		% of Fish	
	LPRSA	Above Dundee Dam	LPRSA	Above Dundee Dam	LPRSA	Above Dundee Dam	LPRSA	Above Dundee Dam	LPRSA	Above Dundee Dam
Silver perch	1	na	0	na	0%	na	0	na	0%	na
Smallmouth bass	5	na	2	na	40%	na	3	na	60%	na
Spottail shiner	5	5	0	1	0%	20%	0	2	0%	40%
Striped bass	5	na	4	na	80%	na	4	na	80%	na
Striped killifish	5	na	0	na	0%	na	1	na	20%	na
Striped mullet	5	na	0	na	0%	na	0	na	0%	na
Tessellated darter	1	1	0	0	0%	0%	0	0	0%	0%
Weakfish	1	na	0	na	0%	na	1	na	100%	na
White catfish	5	na	4	na	80%	na	2	na	40%	na
White perch	2	na	1	na	50%	na	0	na	0%	na
White sucker	5	5	4	0	80%	0%	1	1	20%	20%
Winter flounder	3	na	0	na	0%	na	0	na	0%	na
TOTAL	**119**	**46**	**28**	**7**	**24%**	**15%**	**25**	**13**	**21%**	**28%**

[a] Fin-related abnormalities were not included because such abnormalities may have been the result of incidental effects from catch methods and/or fish handling.

LPRSA – Lower Passaic River Study Area

na – not assessed

For additional comparison, nine of the fish species listed in Table 7-40 were collected from both the LPRSA and above Dundee Dam.[117] For these nine species, the overall incidence of external non-fin-related abnormalities in LPRSA fish (27%) was greater than for fish collected above Dundee Dam (17%). Conversely, the overall incidence of internal abnormalities in LPRSA fish (14%) was less than the incidence of internal abnormalities observed in freshwater reference fish collected from above Dundee Dam (29%).

Table 7-41 presents a summary of the types of external non-fin-related gross abnormalities and internal gross abnormalities recorded in fish from the LPRSA and above Dundee Dam. The greatest incidences of external abnormalities in fish collected from the LPRSA occurred on the anus (6.7%), body surface (5.9%), gills (5.9%), and urogenital opening (5%); the greatest incidences of internal abnormalities occurred on the intestine (7.6%) and liver (5.9%). On fish collected above Dundee Dam, the greatest incidences of external abnormalities occurred on the body surface (8.7%) and gills (6.5%); the greatest incidences of internal abnormalities occurred on the body cavity (11%) and liver (6.5%).

[117] Health observations for some of the 35 fish species collected from the LPRSA may not be directly comparable to those for the 10 fish species collected and assessed from the freshwater background area, because many of the species from the LPRSA were found in a different environment (e.g., only in estuarine water), or were not assessed or found above Dundee Dam. Although northern pike were collected from both areas, health assessments for this species were performed only on individuals collected from above Dundee Dam; the northern pike collected from the LPRSA were retained for potential chemical analysis.

Table 7-41. Pathology evaluation totals and percentages by abnormality type for fish from LPRSA and above Dundee Dam

Location of Abnormality	LPRSA (2009)		LPRSA (2010)		LPRSA (Combined)		Above Dundee Dam (2012)	
	No. of Fish with Abnormality	% of Fish with Abnormality[a]	No. of Fish with Abnormality	% of Fish with Abnormality[a]	No. of Fish with Abnormality	% of Fish with Abnormality[a]	No. of Fish with Abnormality	% of Fish with Abnormality[a]
External[b]								
Anus	8	9.6%	0	0%	8	6.7%	0	0%
Barbels	1	1.2%	1	2.8%	2	1.7%	0	0%
Body form	0	0%	0	0%	0	0%	0	0%
Body surface	3	3.6%	4	11%	7	5.9%	4	8.7%
Bronchial cavity	0	0%	0	0%	0	0%	0	0%
Eyes	2	2.4%	0	0%	2	1.7%	0	0%
Gills	6	7.2%	1	2.8%	7	5.9%	3	6.5%
Isthmus	2	2.4%	1	2.8%	3	2.5%	0	0%
Lips – jaws	1	1.2%	2	5.6%	3	2.5%	0	0%
Opercle	0	0%	0	0%	0	0%	0	0%
Pseudobranch	0	0%	0	0%	0	0%	0	0%
Snout	0	0%	0	0%	0	0%	0	0%
Urogenital opening	6	7.2%	0	0%	6	5.0%	0	0%
Total	**20 [c]**	**24%**	**8 [c]**	**22%**	**28 [c]**	**24%**	**7 [c]**	**15%**
Internal								
Body cavity	3	3.6%	1	2.8%	4	3.4%	5	11%
Gall bladder	2	2.4%	0	0%	2	1.7%	0	0%
Gas bladder	2	2.4%	0	0%	2	1.7%	0	0%
Intestine	8	9.6%	1	2.8%	9	7.6%	1	2.2%
Kidney	1	1.2%	2	5.6%	3	2.5%	1	2.2%
Liver	4	4.8%	3	8.3%	7	5.9%	3	6.5%
Mesenteric fat	0	0%	2	5.6%	2	1.7%	0	0%
Muscle	1	1.2%	0	0%	1	0.8%	0	0%

Table 7-41. Pathology evaluation totals and percentages by abnormality type for fish from LPRSA and above Dundee Dam

Location of Abnormality	LPRSA (2009)		LPRSA (2010)		LPRSA (Combined)		Above Dundee Dam (2012)	
	No. of Fish with Abnormality	% of Fish with Abnormality[a]	No. of Fish with Abnormality	% of Fish with Abnormality[a]	No. of Fish with Abnormality	% of Fish with Abnormality[a]	No. of Fish with Abnormality	% of Fish with Abnormality[a]
Pyloric caeca	0	0%	0	0%	0	0%	1	2.2%
Spleen	2	2.4%	0	0%	2	1.7%	2	4.3%
Stomach	4	4.8%	0	0%	4	3.4%	2	4.3%
Total	**20[c]**	**24%**	**5[c]**	**14%**	**25[c]**	**21%**	**13[c]**	**28%**

a Calculated from the total number of fish assessed (83 fish in 2009, 36 fish in 2010, 119 fish in 2009 and 2010 combined, and 46 fish in 2012).

b Fin-related abnormalities were not included in this summary because such abnormalities may have been the result of incidental effects from catch methods and/or fish handling.

c The total value indicates the number of fish with abnormalities, which is not necessarily equal to the sum of abnormalities, because some fish had more than one abnormality.

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7.6.2 Use of gross abnormalities in determining fish health

The occurrence of gross abnormalities in fish can indicate exposure to chemical and non-chemical stressors. Internal and external abnormalities are absent or occur at very low rates in fish not exposed to stressors, but may occur at greater rates in fish exposed to environmental degradation, chemical pollutants, overcrowding, improper diet, excessive siltation, increased nutrients and organic matter, or other perturbations at more urban sites, resulting in potential physiological stress (Schmitt and Dethloff 2000). Evaluating the prevalence of fish with gross external and internal abnormalities has been used in a number of large-scale monitoring studies, as presented by USEPA's Regional Environmental Monitoring and Assessment Program (REMAP) (1990) and NOAA's National Status and Trends Program (NOAA 2009). These and other studies have generally indicated a greater prevalence of abnormalities in fish from urban sites than in fish from less urban areas. Fish exposed to contaminated sediments through direct contact have been shown to have increased incidence of skin and liver lesions, as well as other deformities and reduced life spans (Johnson et al. 2002; Baumann et al. 1987; Pinkney et al. 2000; Myers et al. 1994).

While certain abnormalities have been identified as incidental effects of fish holding and handling, some occurrences may also be the result of normal conditions as a fish ages (Hinck et al. 2004). In addition, because age cohorts for examined fish were not identified, it may not be appropriate to compare the fish health observations compiled for juvenile fish to those made for older fish. The home range of some fish species adds additional uncertainty to the evaluation of how LPRSA chemicals may affect the occurrence of fish abnormalities, because some species are known to have a large home range, and exposure to chemicals and other factors outside of the LPRSA may affect overall fish health and body condition.

7.6.3 Conclusions

Gross abnormalities have been observed in fish collected from both the LPRSA and above Dundee Dam. For the nine species that were collected from both the LPRSA and above Dundee Dam, the overall incidence of external non-fin-related abnormalities was greater in LPRSA fish, whereas the overall incidence of internal abnormalities was greater in fish from above the Dundee Dam. There are a number of uncertainties in comparing LPRSA data to upstream data, including comparing different species from different aquatic environments (e.g., estuarine vs. freshwater), and comparing species with different home ranges, life histories (i.e., life spans, diets, etc.), and habitat preferences (e.g., benthic vs. pelagic). In addition, the incidence of abnormalities in fish is nearly impossible to attribute to a single factor; rather, it is likely a result of numerous confounding factors, including chemicals, species, age, disease, organic matter, temperature, nutrition, natural parasitic load, season, catch method, and geographic location (Adams et al. 1996). Because of the qualitative nature of the field health observations and the uncertainties associated with their interpretation, conclusive links cannot be established among exposure to chemicals in the LPRSA, effects on LPRSA fish as indicated by field observations, and potential effects on the overall health of fish populations.

7.7 IDENTIFICATION OF PRELIMINARY COCS

The potential for unacceptable risk to fish from COPECs in the LPRSA was evaluated based on the CSM presented in Section 3. Specifically, the risk assessment for fish evaluated Assessment Endpoint No. 5:

- Protection and maintenance (i.e., survival, growth, and reproduction) of omnivorous, invertivorous, and piscivorous fish populations that serve as a forage base for fish and wildlife populations and as a base for sports fisheries

The potential for risk to a number of fish species representing various feeding guilds (benthic omnivores [mummichog, other forage fish, and common carp], invertivores [white perch, channel catfish, brown bullhead, white catfish, and white sucker], and piscivores [American eel, largemouth bass, Northern pike, and smallmouth bass]) was characterized using LPRSA data in the following LOEs:

- **Tissue LOE –** risks to fish characterized using LPRSA fish tissue concentrations
- **Dietary LOE –** risks to fish characterized using LPRSA tissue and sediment data to estimate dietary doses
- **Surface water LOE –** risks to fish characterized using LPRSA surface water concentrations
- **Fish egg tissue LOE –** risks to mummichog characterized using LPRSA fish tissue concentrations to estimate fish egg concentrations

Tissue, dietary doses, surface water, and modeled fish egg concentrations were compared to TRVs based on the literature to derive risk estimates (HQs) in the risk characterization. In addition, several qualitative LOEs were evaluated that involved the evaluation of LPRSA data for mummichog egg counts and gross external and internal health observations.

COPECs with effect-level HQs ≥ 1.0 (based on either a LOAEL TRV for tissue and dietary LOEs or an acute or chronic surface water TRV) in at least one LOE were identified as preliminary COCs (Table 7-42).

Table 7-42. Summary of preliminary COCs for fish

Species by Preliminary COC[b]	Range of LOAEL HQs[a]								
	Fish Tissue LOE		Dietary Dose LOE	Surface Water LOE[c]				Fish Egg Tissue LOE	
				Estuarine		Freshwater			
	HQ/EF Based on TRV-A[d]	HQ/EF Based on TRV-B[e]	HQ	HQ Based on Acute TRV	HQ Based on Chronic TRV	HQ Based on Acute TRV	HQ Based on Chronic TRV	HQ based on TRV-A[d]	HQ based on TRV-B[e]
Mercury									
Mummichog	0.18	0.24	**1.3**	0.005	0.0096	0.0093	0.0062	0.11–**1.1**	0.11–**1.1**
Other forage fish	0.24	0.32						ne[f]	ne[f]
White perch	0.57	0.77	**1.3**					ne[f]	ne[f]
American eel (< 50 cm)	0.74	**1.0**	**1.3**					ne[f]	ne[f]
American eel (≥ 50 cm)			**1.1**					ne[f]	ne[f]
Largemouth bass	**1.9**[g]	**2.6**[g]	0.84					ne[f]	ne[f]
Common carp	0.23	0.31	**1.1**					ne[f]	ne[f]
Smallmouth bass	0.86[g]	**1.2**[g]	0.77					ne[f]	ne[f]
White catfish	0.80	**1.1**	**1.1**					ne[f]	ne[f]
Methylmercury									
Mummichog	0.15	0.20	0.032	not COPEC	not COPEC	not COPEC	not COPEC	0.089–0.88	0.089–0.88
American eel (< 50 cm)	0.80	**1.1**	0.28	not COPEC	not COPEC	not COPEC	not COPEC	ne[f]	ne[f]
American eel (≥ 50 cm)			0.59	not COPEC	not COPEC	not COPEC	not COPEC	ne[f]	ne[f]
Largemouth bass	**1.5**[g]	**2.0**[g]	0.47	not COPEC	not COPEC	not COPEC	not COPEC	ne[f]	ne[f]
Smallmouth bass	0.86[g]	0.85[g]	0.43	not COPEC	not COPEC	not COPEC	not COPEC	ne[f]	ne[f]
White catfish	0.71	0.96	0.26	not COPEC	not COPEC	not COPEC	not COPEC	ne[f]	ne[f]

Table 7-42. Summary of preliminary COCs for fish

Species by Preliminary COC[b]	Range of LOAEL HQs[a]								
	Fish Tissue LOE		Dietary Dose LOE	Surface Water LOE[c]				Fish Egg Tissue LOE	
				Estuarine		Freshwater			
	HQ/EF Based on TRV-A[d]	HQ/EF Based on TRV-B[e]	HQ	HQ Based on Acute TRV	HQ Based on Chronic TRV	HQ Based on Acute TRV	HQ Based on Chronic TRV	HQ based on TRV-A[d]	HQ based on TRV-B[e]
Total PCBs									
Mummichog	0.16	**1.1**	0.35	0.0055	0.16	0.028	0.13	**2.2–3.6**	**11–18**
Other forage fish	0.14	**1.0**						ne[f]	ne[f]
White perch	0.66	**4.7**	0.48					ne[f]	ne[f]
Channel catfish	0.45	**3.2**	0.46					ne[f]	ne[f]
Brown bullhead	0.37	**2.6**						ne[f]	ne[f]
American eel (< 50 cm)	0.53	**3.8**	0.48					ne[f]	ne[f]
American eel (≥ 50 cm)			0.98					ne[f]	ne[f]
Largemouth bass	**2.1**[g]	**15**[g]	0.79					ne[f]	ne[f]
Common carp	**1.4**	**9.8**	0.30					ne[f]	ne[f]
Northern pike	0.53[g]	**3.8**[g]	**1.3**					ne[f]	ne[f]
Smallmouth bass	0.37[g]	**2.6**[g]	0.73					ne[f]	ne[f]
White catfish	0.89	**6.4**	0.42					ne[f]	ne[f]
White sucker	0.76[g]	**5.5**[g]	0.33					ne[f]	ne[f]

Table 7-42. Summary of preliminary COCs for fish

Species by Preliminary COC[b]	Range of LOAEL HQs[a]								
	Fish Tissue LOE		Dietary Dose LOE	Surface Water LOE[c]				Fish Egg Tissue LOE	
				Estuarine		Freshwater			
	HQ/EF Based on TRV-A[d]	HQ/EF Based on TRV-B[e]	HQ	HQ Based on Acute TRV	HQ Based on Chronic TRV	HQ Based on Acute TRV	HQ Based on Chronic TRV	HQ based on TRV-A[d]	HQ based on TRV-B[e]
PCB TEQ - fish									
White perch	0.018 (0.091[h])	**1.2**	**1.0**	0.00000013	0.0000056	0.000012	0.000052	ne[f]	ne[f]
Channel catfish	0.015 (0.078[h])	**1.0**	ne					ne[f]	ne[f]
Brown bullhead	0.011 (0.057[h])	0.72						ne[f]	ne[f]
American eel (< 50 cm)	0.010 (0.052[h])	0.67	0.95					ne[f]	ne[f]
American eel (≥ 50 cm)			**1.8**					ne[f]	ne[f]
Largemouth bass	0.14[g] (0.74[g,h])	**9.4**[g]	**1.6**					ne[f]	ne[f]
Common carp	0.037 (0.19[h])	**2.4**	0.74					ne[f]	ne[f]
Northern pike	0.019[g] (0.010[g,h])	**1.3**[g]	**2.1**					ne[f]	ne[f]
Smallmouth bass	0.012[g] (0.061[g,h])	0.78[g]	**1.5**					ne[f]	ne[f]
White catfish	0.029 (0.15[h])	**1.9**	0.82					ne[f]	ne[f]
White sucker	0.027[g] (0.14[g,h])	**1.8**[g]	0.76					ne[f]	ne[f]

Table 7-42. Summary of preliminary COCs for fish

Species by Preliminary COC[b]	Range of LOAEL HQs[a]								
	Fish Tissue LOE		Dietary Dose LOE	Surface Water LOE[c]				Fish Egg Tissue LOE	
				Estuarine		Freshwater			
	HQ/EF Based on TRV-A[d]	HQ/EF Based on TRV-B[e]	HQ	HQ Based on Acute TRV	HQ Based on Chronic TRV	HQ Based on Acute TRV	HQ Based on Chronic TRV	HQ based on TRV-A[d]	HQ based on TRV-B[e]
2,3,7,8-TCDD									
Mummichog	0.41 (**2.1**[h])	**27**	ne	0.00022	0.009	0.026	0.11	not COPEC	
Other forage fish	0.38 (**2.0**[h])	**26**						ne[f]	ne[f]
White perch	**1.6 (8.3**[h]**)**	**110**	ne					ne[f]	ne[f]
Channel catfish	0.80 (**4.2**[h])	**53**	ne					ne[f]	ne[f]
Brown bullhead	**1.3** (**6.5**[h])	**83**						ne[f]	ne[f]
American eel (< 50 cm)	0.19 (1.0[h])	**13**	ne					ne[f]	ne[f]
American eel (≥ 50 cm)			ne					ne[f]	ne[f]
Largemouth bass	**1.5**[g] **(7.8**[g,h]**)**	**100**[g]	ne					ne[f]	ne[f]
Common carp	**5.1 (27**[h]**)**	**340**	ne					ne[f]	ne[f]
Northern pike	0.79[g] (**4.1**[g,h])	**53**[g]	ne					ne[f]	ne[f]
Smallmouth bass	0.63[g] (**3.3**[g,h])	**42**[g]	ne					ne[f]	ne[f]
White catfish	**1.8 (9.1**[h]**)**	**120**	ne					ne[f]	ne[f]
White sucker	**1.1**[g] (**5.7**[g,h])	**72**[g]	ne					ne[f]	ne[f]

Table 7-42. Summary of preliminary COCs for fish

Species by Preliminary COC[b]	Range of LOAEL HQs[a]								
	Fish Tissue LOE		Dietary Dose LOE	Surface Water LOE[c]				Fish Egg Tissue LOE	
				Estuarine		Freshwater			
	HQ/EF Based on TRV-A[d]	HQ/EF Based on TRV-B[e]	HQ	HQ Based on Acute TRV	HQ Based on Chronic TRV	HQ Based on Acute TRV	HQ Based on Chronic TRV	HQ based on TRV-A[d]	HQ based on TRV-B[e]
PCDD/PCDF TEQ – fish									
Mummichog	0.43 (**2.2**[h])	**28**	**200**	0.00029	0.012	0.027	0.11	0.52–0.84	0.52–0.84
Other forage fish	0.41 (**2.1**[h])	**27**						ne[f]	ne[f]
White perch	**1.7 (8.7**[h]**)**	**110**	**170**					ne[f]	ne[f]
Channel catfish	0.83 (**4.3**[h])	**56**	**190**					ne[f]	ne[f]
Brown bullhead	**1.3 (7.0**[h]**)**	**9**						ne[f]	ne[f]
American eel (< 50 cm)	0.20 (1.0[h])	**13**	**180**					ne[f]	ne[f]
American eel (≥ 50 cm)			**190**					ne[f]	ne[f]
Largemouth bass	**1.5**[g] **(7.8** [g,h]**)**	**100**[g]	**150**					ne[f]	ne[f]
Common carp	**5.2 (27**[h]**)**	**340**	**200**					ne[f]	ne[f]
Northern pike	0.83[g] (**4.3**[g,h])	**56**[g]	**200**					ne[f]	ne[f]
Smallmouth bass	0.63[g] (**3.3** [g,h])	**42**[g]	**140**					ne[f]	ne[f]
White catfish	**1.8 (9.6**[h]**)**	**120**	**160**					ne[f]	ne[f]
White sucker	**1.1**[g] (**5.7**[g,h])	**72**[g]	**190**					ne[f]	ne[f]

Table 7-42. Summary of preliminary COCs for fish

Species by Preliminary COC[b]	Range of LOAEL HQs[a]									
	Fish Tissue LOE		Dietary Dose LOE	Surface Water LOE[c]				Fish Egg Tissue LOE	
				Estuarine		Freshwater			
	HQ/EF Based on TRV-A[d]	HQ/EF Based on TRV-B[e]	HQ	HQ Based on Acute TRV	HQ Based on Chronic TRV	HQ Based on Acute TRV	HQ Based on Chronic TRV	HQ based on TRV-A[d]	HQ based on TRV-B[e]
Total TEQ - fish									
Mummichog	0.43 (**2.2**)	**28**	**210**	0.00022	0.0093	0.027	0.11	0.52–0.84	0.52–0.84
Other forage fish	0.41 (**2.1**[h])	**27**						ne[f]	ne[f]
White perch	**1.7 (8.7[h])**	**110**	**170**					ne[f]	ne[f]
Channel catfish	0.83 (**4.3**[h])	**56**	**190**					ne[f]	ne[f]
Brown bullhead	**1.3 (7.0[h])**	**89**						ne[f]	ne[f]
American eel (< 50 cm)	0.21 (**1.1**[h])	**14**	**190**					ne[f]	ne[f]
American eel (≥ 50 cm)			**200**					ne[f]	ne[f]
Largemouth bass	**1.5[g] (7.8[g,h])**	**100[g]**	**150**					ne[f]	ne[f]
Common carp	**5.2 (27[h])**	**340**	**200**					ne[f]	ne[f]
Northern pike	0.92[g] (**4.8**[g,h])	**61[g]**	**200**					ne[f]	ne[f]
Smallmouth bass	0.68[g] (**3.6**[g,h])	**46[g]**	**140**					ne[f]	ne[f]
White catfish	**1.9 (10[h])**	**130**	**160**					ne[f]	ne[f]
White sucker	**1.1**[g] (**5.7**[g,h])	**72[g]**	**190**					ne[f]	ne[f]
Dieldrin									
Channel catfish	0.24	**1.2**	ne	0.0031	0.013	not COPEC	not COPEC	ne[f]	ne[f]
American eel (< 50 cm)	0.27	**1.4**	ne					ne[f]	ne[f]
American eel (≥ 50 cm)			ne					ne[f]	ne[f]
Largemouth bass	0.20[g]	**1.0**[g]	ne					ne[f]	ne[f]

Table 7-42. Summary of preliminary COCs for fish

Species by Preliminary COC[b]	Range of LOAEL HQs[a]								
	Fish Tissue LOE		Dietary Dose LOE	Surface Water LOE[c]				Fish Egg Tissue LOE	
				Estuarine		Freshwater			
	HQ/EF Based on TRV-A[d]	HQ/EF Based on TRV-B[e]	HQ	HQ Based on Acute TRV	HQ Based on Chronic TRV	HQ Based on Acute TRV	HQ Based on Chronic TRV	HQ based on TRV-A[d]	HQ based on TRV-B[e]
Common carp	0.28	**1.4**	ne					ne[f]	ne[f]
Northern pike	0.22[g]	**1.1**[g]	ne					ne[f]	ne[f]
Total DDx									
Common carp	**1.3**	**1.7**	0.0096	0.096	0.17	0.011	0.019	ne[f]	ne[f]
Cyanide									
LPRSA fish community	ne[i]	ne[i]	ne	**1.6**	**5.3**	0.21	0.91	ne[f]	ne[f]
Cadmium									
Mummichog	0.28	na	**1.3**	0.0013	0.0054	0.0018–0.063	0.0048–0.016	ne[f]	ne[f]
Other forage fish	0.36	na						ne[f]	ne[f]
Common carp	0.20	na	**1.2**					ne[f]	ne[f]
White perch	0.088	na	**1.1**					ne[f]	ne[f]
White sucker	0.081	na	**1.1**					ne[f]	ne[f]
American eel (< 50 cm)	0.55	na	**1.2**					ne[f]	ne[f]

Table 7-42. Summary of preliminary COCs for fish

Species by Preliminary COC[b]	Range of LOAEL HQs[a]								
	Fish Tissue LOE		Dietary Dose LOE	Surface Water LOE[c]				Fish Egg Tissue LOE	
				Estuarine		Freshwater			
	HQ/EF Based on TRV-A[d]	HQ/EF Based on TRV-B[e]	HQ	HQ Based on Acute TRV	HQ Based on Chronic TRV	HQ Based on Acute TRV	HQ Based on Chronic TRV	HQ based on TRV-A[d]	HQ based on TRV-B[e]
Copper									
Mummichog	na	**2.1**	0.28	0.14–**2.7**	0.14–**2.7**	0.023–0.65	0.037–**1.0**	ne[f]	ne[f]
Other forage fish	na	**2.7**						ne[f]	ne[f]
White perch	na	**9.3**	0.36					ne[f]	ne[f]
Channel catfish	na	0.87	0.24					ne[f]	ne[f]
Brown bullhead	na	0.57						ne[f]	ne[f]
American eel (< 50 cm)	na	**1.7**	0.36					ne[f]	ne[f]
American eel (≥ 50 cm)			0.31						
Largemouth bass	na	0.39	0.22					ne[f]	ne[f]

Bold identifies HQs ≥ 1.0.

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

[b] Only COPECs with HQs ≥ 1.0 based on LOAEL TRV are included in table.

[c] Surface water evaluated for the LPRSA fish community. HQs for surface water derived using UCL concentrations compared to the surface water TRV.

[d] TRVs were derived from the primary literature review.

[e] TRVs were based on USEPA's revised draft of the LPR restoration project FFS (Louis Berger et al. 2014) or first draft of the LPR restoration project FFS (Malcolm Pirnie 2007b).

[f] The fish egg LOE was limited to mummichog for total PCBs, PCDD/PCDF TEQ - fish, and total TEQ - fish (Section 7.4.1).

[g] Fewer than six detected concentrations available, so the HQ based on a maximum concentration rather than a UCL concentration.

[h] HQs in parenthesis based on additional alternative SSD-derived LOAEL evaluated (see text in Section 7.1.3 for details).

[i] Cyanide not evaluated using the tissue LOE for fish. Cyanide not analyzed in LPRSA tissue.

BERA – baseline ecological risk assessment
COC – chemical of concern
COI – chemical of interest
COPEC – chemical of potential ecological concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
EF – exceedance factor
FFS – focused feasibility study
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
LOE – line of evidence
LPR – Lower Passaic River
LPRSA – Lower Passaic River Study Area
ne – not evaluated (not a COI and/or COPEC for this LOE)
NJDEP – New Jersey Department of Environmental Protection
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
PCDD– pentachlorodibenzo-*p*-dioxin
PCDF – pentachlorodibenzofuran
SSD – species sensitivity distribution
TCDD – tetrachlorodibenzo-*p*-dioxin
TEQ – toxic equivalent
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT and 4,4′-DDT)
TRV – toxicity reference value
UCL – upper confidence limit on the mean
USEPA – US Environmental Protection Agency

Comparison of the dietary and tissue LOEs is important in determining risk conclusions and recommendations. This is particularly true for regulated metals, given that the evaluation of risks to fish from regulated metals using tissue residues is not recommended per the USEPA framework for metals risk assessment (USEPA 2007e), as it "does not appear to be a robust indicator of toxic dose." Furthermore, USEPA recommends a dietary assessment of inorganic metals only for conservative screening purposes, because the uptake by and toxicity of inorganic metals to fish can vary widely depending upon a number of factors, including (but not limited to) digestive physiology (e.g., gut residence time), food nutritional quality, distribution and chemical form of metals in prey tissue, and environmental conditions under which toxicity is evaluated (e.g., temperature) (USEPA 2007e). The results for those preliminary COCs that were evaluated using both tissue and dietary LOEs for fish are summarized as follows:

- Copper is the only regulated metal with LOAEL HQs ≥ 1.0 based on the tissue residue LOE; however, LOAEL HQs did not exceed 1.0 based on the dietary LOE.
- Cadmium is the only regulated metal with LOAEL HQs ≥ 1.0 based on the dietary dose approach, and LOAEL HQs did not exceed 1.0 based on the tissue residue LOE.
- Mercury, total PCBs, PCB - TEQ, PCDD/PCDF - TEQ, and total TEQ had LOAEL HQs ≥ 1.0 based on both the tissue residue and dietary LOEs, although HQs based on the dietary LOE were greater than the ranges derived from the tissue residue LOE.
- Total DDx had LOAEL HQs ≥ 1.0 based on the tissue residue LOE only (i.e., not based on the dietary LOE).

These preliminary COCs are discussed further in Section 13 in the identification of ecological risk drivers.

The results of this fish risk assessment will be used in the FS as a tool to help risk managers to make potential remedial decisions for the LPRSA. Determining the potential for unacceptable ecological risk at the population level provides information relevant to decisions to be made in the FS or other programmatic environmental management changes. The TRVs used to evaluate risk to fish in this BERA are organism-level effects, including those that affect particular attributes of organisms within a population, such as survival, growth, and reproduction. Survival, growth, and reproduction of individual organisms may, in turn, affect populations of those organisms, depending upon the magnitude and severity of the effect. However, population-level effects—such as size or density of population, population growth, or population survival—are more direct measures of influences on the population as a whole. Since BERAs evaluate populations as assessment endpoints, not individuals, a number of other factors, including the potential magnitude and severity of the effect, should be assessed to determine if a risk driver (defined and identified in Section 13) should be used in developing PRGs and RALs.

8 Bird Assessment

This section presents the risk assessment for the bird species selected for evaluation in the LPRSA BERA: spotted sandpiper, belted kingfisher, and great blue heron. The risk assessment for birds evaluated the following assessment endpoint, according to the PFD (Windward and AECOM 2009):

- **Assessment Endpoint No. 8 –** Protection and maintenance (i.e., survival, growth, and reproduction[118]) of herbivorous, omnivorous, sediment-probing, and piscivorous bird populations[119]

The potential for risks to bird species was characterized using two LOEs for COPECs identified in the SLERA:

- **Dietary LOE –** comparison of estimated COPEC dietary doses to dietary TRVs
- **Egg tissue LOE –** comparison of estimated COPEC concentrations in egg tissue of piscivorous birds to egg tissue TRVs

COPECs with calculated HQs ≥ 1.0 were assessed to determine a list of preliminary COCs for further evaluation in the FS. The bird risk assessment process is outlined in Table 8-1. Sections 8.1 and 8.2 present the dietary and bird egg tissue assessments, respectively. Uncertainties associated with various components of the dietary and bird egg tissue assessments are discussed throughout their respective sections, and key uncertainties are summarized at the ends of Sections 8.1 and 8.2. Section 8.3 identifies bird preliminary COCs, which are further evaluated in Section 13.

Table 8-1. Outline of the bird risk assessment

Section	Section Title	Section Contents
8.1	Dietary Assessment	for each LOE, presents COPECs based on the SLERA, exposure and effects data, HQs, uncertainty discussion, and summary of risk characterization
8.2	Bird Egg Tissue Assessment	
8.3	Identification of preliminary COCs	identifies preliminary COCs

COC – chemical of concern
COPEC – chemical of potential ecological concern
HQ – hazard quotient
LOE – line of evidence
SLERA – screening-level ecological risk assessment

[118] Few aquatic birds currently use the LPRSA for breeding because of habitat constraints. The reproduction assessment endpoint for birds evaluates whether existing chemical concentrations would impact reproduction if suitable habitat were present.

[119] Consistent with the PFD, neither herbivorous nor omnivorous birds were identified (Windward and AECOM 2009) in the CSM as feeding guilds to be quantitatively evaluated. Representative species were not selected because the evaluation of other avian feeding guilds (i.e., sediment-probing and piscivorous birds) are protective of herbivorous and omnivorous birds.

8.1 DIETARY ASSESSMENT

A dietary assessment was conducted for each of the three selected bird species (spotted sandpiper, belted kingfisher, and great blue heron), consistent with the PFD (Windward and AECOM 2009). For each bird species, the assessment was conducted for the COPECs that were identified in the SLERA (see Section 5). This section summarizes the COPECs, describes how exposure and effects concentrations were derived, presents the HQs, and summarizes the uncertainties associated with the dietary assessment.

8.1.1 COPECs

The COPECs for each bird species were identified using a risk-based screening process conducted in the SLERA, wherein doses based on maximum concentrations were compared to dietary screening-level TRVs (Table 8-2; Appendix A); the results are summarized in Section 5. The COPECs identified for the bird dietary LOE included nine metals, total LPAHs, total HPAHs, total PCBs, PCB TEQ - bird, PCDD/PCDF TEQ - bird, total TEQ - bird, and total DDx (Table 8-2).

Table 8-2. Bird dietary COPECs

COPEC		
Metals		
Cadmium	Lead	Selenium
Chromium	Methyl mercury	Vanadium
Copper	Nickel	Zinc
PAHs		
Total LPAHs	Total HPAHs	
PCBs		
Total PCBs	PCB TEQ- bird	
PCDDs/PCDFs		
PCDD/PCDF TEQ - bird	Total TEQ - bird	
Organochlorine Pesticides		
Total DDx		

Note: COPECs are those COIs for which the maximum modeled dietary dose exceeded its TSV. If a TSV was exceeded based on any avian species evaluated in the SLERA, it was retained as a COPEC for all birds.

COI – chemical of interest
COPEC – chemical of potential ecological concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
LPAH – low-molecular-weight polycyclic aromatic hydrocarbon
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
PCDD – polychlorinated dibenzo-*p*-dioxin
PCDF – polychlorinated dibenzofuran
SLERA – screening-level ecological risk assessment
TEQ – toxic equivalent
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT and 4,4′-DDT)
TSV – toxicity screening value

A number of COIs could not be screened as part of the SLERA (Appendix A) because no bird diet TSVs were available. These COIs are presented in Section 5.2.2, along with a discussion of the implications of not being able to evaluate these COIs.

8.1.2 Exposure

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

8.1.2.1 Methods

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

$$Dose = \frac{\left[\left(FIR \times EPC_{prey}\right) + \left(SIR \times EPC_{sed}\right) + \left(WIR \times EPC_{water}\right)\right]}{BW} \times SUF \quad \textbf{Equation 8-1}$$

Where:

Dose	=	daily ingested dose (mg/kg bw/day)
FIR	=	food ingestion rate (kg ww/day)
EPC_{prey}	=	exposure point concentration of chemical in prey tissue (mg/kg ww)
SIR	=	incidental sediment ingestion rate (kg dw/day)
EPC_{sed}	=	exposure point concentration of chemical in sediment (mg/kg dw)
WIR	=	water ingestion rate (L/day)
EPC_{water}	=	exposure point concentration of chemical in water (mg/L)
BW	=	body weight (kg)
SUF	=	site use factor (SUF) (unitless); proportion of time selected species spends foraging in the LPRSA

The body weights, ingestion rates, and SUFs were obtained from the literature for each bird species and are described in Section 8.1.2.2. The EPC for prey for each bird species was calculated from the fractions of different prey types in the bird species' diets and the EPCs for each of those prey types, as follows:

$$EPC_{prey} = (EPC_1 \times F_1) + (EPC_2 \times F_2) \quad \textbf{Equation 8-2}$$

Where:

EPC_{prey}	=	exposure point concentration in prey items (mg COPEC/kg food ww)
$EPC_{1,2}$	=	exposure point concentration in each individual prey type (mg COPEC/kg tissue dw)
$F_{1,2}$	=	fraction ingested of each individual prey type (kg fish/kg food)

The DF of each component in each bird species' diets was based on information obtained from the literature. The DFs assumed for each species and the assumptions used to derive them are described in detail in Section 8.1.2.3. All three of the bird species evaluated were conservatively assumed to use the LPRSA for the entire year (i.e., no adjustment was made for seasonal site use).

8.1.2.2 Body weights, ingestion rates, and SUFs

Average body weights and average ingestion rates for sediment, water, and food for use in the dietary dose calculations were obtained from the literature, as summarized in Table 8-3. The exposure parameters were selected as follows:

- Body weights for birds were based on average male and female body weights reported in USEPA's *Wildlife Exposure Factors Handbook* (USEPA 1993). The effect on HQs of using the maximum and minimum male or female body weights reported by USEPA is evaluated in Section 8.1.4.2.
- FIRs were based on the measured ingestion rate for the species selected for evaluation or similar species, when available, and were expressed on a wet weight basis. Uncertainties associated with the selected FIRs are evaluated in Section 8.1.4.2.
- The percentage of incidentally ingested sediment was based on data from the literature. When species-specific or appropriate surrogate data were unavailable, best professional judgment was used to estimate incidental SIRs. There is some uncertainty associated with the spotted sandpiper SIR because of the wide range of values found in the literature (7.3 to 30% of the diet) for four sandpiper species other than spotted sandpiper. The uncertainty associated with the spotted sandpiper ingestion rate and the estimated SIRs for all of the bird species evaluated is addressed in the Section 8.1.4.2. Incidental SIRs were expressed on a dry weight basis, as a percentage of the dry weight FIR.
- Dry weight FIRs (required for deriving incidental SIRs) were derived from wet weight ingestion rates assuming 80% moisture in prey (based on average moisture contents of 72, 79, and 88% for fish, invertebrates, and worms from the LPRSA, respectively) when dry weight FIRs were not available from the literature.
- Water ingestion rates (WIRs) for birds were based on an allometric equation from Calder and Braun (1983), as cited in USEPA (1993).

Table 8-3. Exposure parameter values for spotted sandpiper, great blue heron, and belted kingfisher

Species	BW (kg)[a]	Food Ingestion		Incidental SI			WIR (L/day)[c]
		FIR (kg ww/day)	Source	SI (%)[b]	SIR (kg dw/day)	Source	
Spotted sandpiper	0.0425	0.027[d]	Nagy (2001)[d]	18[e]	0.0013	Beyer et al. (1994)	0.0071
Great blue heron	2.3	0.40[f]	Kushlan (1978), as cited in USEPA (1993)	1	0.00081[g]	no empirical data available; based on feeding habits and best professional judgment	0.10
Belted kingfisher	0.147	0.074[h]	USEPA (1993)	0.5	0.000074[g]	no empirical data available; based on feeding habits and best professional judgment	0.016

a Average of male and female adult body weights reported by USEPA (1993).

b Based on percentage of the dry diet that is incidentally ingested sediment. Dry weight FIRs for great blue heron and belted kingfisher were estimated from wet weight FIRs assuming 80% moisture in the diet. The dry weight FIR for spotted sandpiper was based on data presented in Nagy (2001).

c WIR is based on Calder and Braun (1983), in which bird WIR = 0.059 x $BW^{0.67}$.

d Based on the body weight-normalized FIR of 0.64 g ww/g bw/day for common sandpiper.

e Based on the average of SIRs measured for four sandpiper species (stilt sandpiper [17%], semipalmated sandpiper [30%], least sandpiper [7.3%], and western sandpiper [18%]).

f FIR (g ww/day) = $10^{0.966\log(BW)-0.64}$ and body weight is in grams.

g Wet weight FIR converted to dry weight FIR assuming 80% moisture in prey (based on average moisture contents of 72, 79, and 88% for fish, invertebrates, and worms, respectively) to determine SIR in kg/day.

h FIR = 0.50 g ww/g bw/day.

BW or bw – body weight
dw – dry weight
FIR – food ingestion rate
SI – sediment ingestion
SIR – sediment ingestion rate
USEPA – US Environmental Protection Agency
WIR – water ingestion rate
ww – wet weight

A SUF of 1 was used for all bird species evaluated, based on the assumption that they obtain 100% of their diet from their preferential foraging (i.e., exposure) areas in the LPRSA. It is possible that some of the bird species evaluated forage outside of the LPRSA, and therefore use the LPRSA as their exposure area less than 100% of the time. The exposure dose, and thus the HQ, is directly proportional to the SUF (Equation 8-1); if a species uses the LPRSA 50% of the time, the HQ will decrease by 50%. The use of an SUF of 1 provides conservative estimates of the potential risks in these cases. The effect on the HQs of varying the SUF is addressed in Section 8.1.4.2.

8.1.2.3 Prey composition and exposure areas

For the dietary dose equation (Equation 8-1), prey ingested by each bird species evaluated included only those prey types for which tissue chemistry data from the LPRSA were available. These tissue data included freshwater and estuarine worms (from the bioaccumulation study), blue crab, and fish. While fish and blue crab data were field collected, worm tissue data were based on the laboratory bioaccumulation study in which worms were exposed to homogenized sediment collected from the 0- to 15-cm depth horizon.

The proportions of worms, blue crab, and fish in the diet of each species are presented in Table 8-4. The rationale for the selection of these prey portions and sizes is presented in more detail for spotted sandpiper, belted kingfisher, and great blue heron later in this section. For great blue heron and belted kingfisher, two dietary scenarios were evaluated. The first scenario assumed that these two species consumed fish from only one size class (≤ 9 cm for belted kingfisher and ≤ 13 cm for great blue heron). The second scenario included a range of size classes for each species (USEPA 2015b, c, 2016g) (Table 8-4).

Table 8-4. Prey composition used to estimate dietary dose for spotted sandpiper, great blue heron, and belted kingfisher

Species	Percentage of Prey Type in Diet								Rationale for Scenario
			Fish Size Range						
	Worm[a]	Blue Crab	0–9 cm	9–13 cm	0–13 cm	13–18 cm	18–30 cm	> 30 cm	
Spotted sandpiper									
Scenario 1	100	0	na	na	na	na	na	na	Evaluate risk using dietary assumption that spotted sandpiper consume only invertebrates.
Belted kingfisher									
Scenario 1	na	15	85	0	na	0	na	na	Evaluate risk assuming belted kingfisher consume only fish 0–9 cm long.
Scenario 2	na	15	31.5	51	na	2.5	na	na	Evaluate risk based on a range of fish size classes for belted kingfisher.
Great blue heron									
Scenario 1	na	0	na	na	100	0	0	0	Evaluate risk assuming great blue heron consume only fish 0–13 cm long.
Scenario 2	na	0	na	na	17	29	40	14	Evaluate risk based on a range of fish size classes for great blue heron; range to include large fish, including common carp.

[a] Includes both freshwater and estuarine worms.
na – not applicable; not a dietary prey item
USEPA – US Environmental Protection Agency

For exposure areas, it was assumed that belted kingfisher would feed only above RM 6, based on site-specific surveys and availability of habitat (as discussed in more detail in the remainder of this section). It was assumed also that mudflats throughout the LPRSA were the preferred foraging areas for spotted sandpiper and great blue heron. Risk to belted kingfisher was also evaluated by assuming that the species forages throughout the LPRSA. In addition, risk to all three bird species was evaluated by assuming that these species forage on a reach-specific basis. For exposure areas from which surface water would be consumed, it was assumed that only freshwater (i.e., water at or upstream of RM 4) would be ingested by any of the bird species evaluated. The exposure areas for bird species for sediment, prey, and surface water are presented in Table 8-5, and the rationale for the selection of these exposure areas is presented in more detail for each bird species evaluated in the remainder of this section.

Table 8-5. LPRSA exposure areas for spotted sandpiper, great blue heron, and belted kingfisher

Species	Exposure Area			Rationale
	Prey	Sediment	Surface Water[a]	
Spotted sandpiper	site wide	site-wide mudflat areas[b]	RM ≥ 4	Evaluate risk to spotted sandpiper assuming they forage throughout the entire LPRSA.
	by reach	mudflats by reach[b,c]	by reach[c,d]	Evaluate risk to spotted sandpiper within 2-mi reaches.
Great blue heron	site wide	site-wide mudflat areas[b]	RM ≥ 4	Evaluate risk to great blue heron assuming they forage throughout the entire LPRSA.
	by reach[c]	mudflats by reach[b,c]	by reach[c,d]	Evaluate risk to great blue heron within 2-mi reaches.
Belted kingfisher	≥ RM 6 for fish 0–9 cm long and site wide for blue crab	RM ≥ 6	RM ≥ 6	Evaluate risk to belted kingfisher assuming they breed and forage only above RM 6 based on site-specific information.
	site wide[c]	site wide[c]	RM ≥ 4[d]	Evaluate risk to belted kingfisher on a site-wide basis.
	by reach[c]	by reach[c]	by reach[c, d]	Evaluate risk to belted kingfisher within 2-mi reaches.

[a] Surface water included in the drinking water exposure pathway was limited to freshwater.

[b] Mudflats are defined as areas within -2 ft MLLW and < 6º slope and include all grain sizes.

[c] Derived based on process identified by USEPA (2015b), USEPA (2015c), USEPA (2016g).

[d] For those reaches without freshwater (i.e., Reaches 1 and 2), surface water from the next closest reach with freshwater (i.e., Reach 3) was used.

LPRSA – Lower Passaic River Study Area

MLLW – mean lower low water

RM – river mile

USEPA – US Environmental Protection Agency

Spotted Sandpiper

Prey Composition

The diet of the spotted sandpiper consists primarily of aquatic invertebrates and insects, including flying insects, worms, fish, crustaceans, mollusks, and carrion (Oring et al. 1983). Along the shoreline, spotted sandpiper wade into the water's edge and probe or peck at prey in or near the sediment (Oring et al. 1997).

Spotted sandpipers primarily inhabit mudflats across the LPRSA. Avian surveys (Windward 2011a, 2019e) indicate that spotted sandpiper have been observed in all portions of the LPRSA during spring, summer, and fall. As such, it was considered relevant to include both estuarine and freshwater worms in the sandpiper diet.

The diet of the spotted sandpiper was represented by estuarine and freshwater worm tissue data from the laboratory bioaccumulation study, because field-collected data for benthic invertebrates were not available. The worm chemistry dataset from the bioaccumulation study consisted of 19 worm tissue samples, 5 of which were estuarine and 14 of which were freshwater. Each of the 19 worm tissue samples was weighted equally in the sandpiper dietary dose calculation.

Exposure Area

Spotted sandpiper require exposed areas (e.g., mudflats or sandbars) with firm, fine-grained (e.g., silt or sand) sediment for feeding (Oring et al. 1997). These habitats were found throughout the LPRSA during the avian surveys (Windward 2011a, 2019e; BBL 2002). Although they have a lower preference for areas near human activity, their habitat is generally not limited by land use or shoreline features (Windward 2011a, 2019e). This information suggests that the exposure area of spotted sandpiper includes only mudflat areas throughout the LPRSA.

A home range of about 22 km^2 has been reported for spotted sandpipers, with dispersal distances ranging from 22.5 to 30.5 km (Reed and Oring 1993). However, during their reproductive season, their territories are much smaller, ranging from 812 to 20,000 m^2 (0.0008 to 0.02 km^2) with beach lengths ranging from 20 to 400 m (Oring et al. 1997). Data for other sandpiper species suggests that their home ranges vary from approximately 0.80 to 8.05 km on stopovers during migration or during breeding:

- Incubating buff-breasted sandpiper females were observed within 1 km of the nest site, and females and their broods were seen within a 1- to 3-km^2 area (Lanctot and Laredo 1994).
- The mean distances from nest of upland sandpiper females and males were 869 m and 241 m, respectively (Ailes and Toepfer 1977 as cited in Houston and Bowen 2001).
- Stilt sandpipers foraged from territory up to 8 km away from the nest (Klima and Jehl 1998; Jehl 1973).

- Purple sandpipers were reported to fly a mean distance of 0.65 to 1.6 km from the nest during incubation (Pierce 1993).
- Western sandpipers (*Calidris mauri*) were reported to journey 4 to 6 km while foraging on beaches during migratory stopovers in the Fraser River estuary in British Columbia (Butler et al. 2002).

There is potential for breeding spotted sandpiper to be present in the vicinity of the LPRSA based on evidence (not specified) observed in a survey block containing Kearny Marsh (near the southern end of the Hackensack Meadowlands, approximately 1 mi from the LPRSA; Map 1-1) (Walsh et al. 1999; as cited in Ludwig et al. 2010). Therefore, it is possible that spotted sandpipers could feed within subareas of the LPRSA during their breeding season, so in addition to calculating HQs using site-wide data, HQs were calculated using data from 2-mi increments of the LPRSA (the approximate average home range of other sandpiper species on stopovers during migration or breeding). A 100% SUF, which could be an overestimation of risk, was used for these smaller areas, based on the assumption that breeding pairs would forage in a smaller area.

Great Blue Heron

Prey Composition

The diet of the great blue heron consists predominately (at least 75 but up to 100%) of fish (Collazo 1985; Kirkpatrick 1940; Alexander 1976; Quinney 1982). However, great blue herons are also opportunistic foragers that feed on anything they can capture and can fit into their mouths, such as crustaceans, amphibians, small mammals, reptiles, and insects (Howell 1932; Cottam and Uhler 1945; HeronConservation 2016; Peifer 1979; Vennesland and Butler 2011). Proportions of different taxa eaten vary considerably by feeding location, season, and life stage of the heron (Alexander 1976; Butler 1991).

In an analysis of the stomach contents of great blue heron from along a Michigan lake and river, fish (primarily trout) made up 94 to 98% of the species' diet; crustaceans and amphibians made up < 2 to 5%, and birds and mammals made up 0 to 1% (percentages calculated on a wet weight basis) (Alexander 1977; as cited in USEPA 1993). Because empirical data were not available to model dietary concentrations for amphibians, birds, or mammals, and the dietary proportion of crustaceans was quite small (1%), the LPRSA diet for heron was modeled assuming 100% fish ingestion.

Based on an average beak length of 13.5 cm (Poole 2011), Krebs (1974) determined that more than 92% of great blue heron fish prey are small or medium sized (up to about 6.8 cm in length), and the remaining fish prey are greater than or equal to the length of the beak (13.5 cm). Because most fish consumed by great blue heron are less than 13 cm long and there is no indication that heron favor particular fish species, fish in the diet of great blue heron were limited to all fish ≤ 13 cm in length in Scenario 1 of the risk calculations (Table 8-4). Although some studies (e.g., Hoffman 1978) have stated that it is possible that great blue heron may consume some larger fish (up to about 25 cm long), it is likely that fish of this size represent a very small fraction of the total number of fish typically consumed.

Larger fish (including common carp > 30 cm) were also considered as potential prey for great blue heron (USEPA 2015b, c, 2016g). Four general size classes of fish were determined as possible prey for great blue heron, as follows: 0 to 13 cm, 13 to 18 cm, 18 to 30 cm, and > 30 cm. Quantitative data from the literature were evaluated to determine appropriate prey portions of fish within these general size classes. Data from the literature were reported as frequency of occurrence within size classes, and were converted into weight for those size classes using the species-specific length-to-weight calculator from FishBase (www.fishbase.org)[120] (Table 8-6). On average, the percentages of fish, by weight, in the great blue heron diet were 17% for fish 0 to 13 cm, 29% for fish 13 to 18 cm, 40% for fish 18 to 30 cm, and 14% for fish > 30 cm. These percentages were evaluated for the great blue heron's diet in Scenario 2 of the risk calculations (Table 8-4).

Table 8-6. Summary of great blue heron prey items reported in the literature

Location	Sample Type/Size	Fish Size Class Occurrence (based on item frequency)				Fish Size Class Occurrence (based on weight)[a]				Citation
		0–13 cm	13–18 cm	18–30 cm	> 30 cm	0–13 cm	13–18 cm	18–30 cm	> 30 cm	
Michigan rivers, lakes, and streams	stomach contents	57%	19%	24%	< 1%	9%	16%	72%	3%	Alexander (1976)
British Columbia	observations of prey; size estimated on bill size	93%	7%	0%	0%	35%	65%	0%	0%	Krebs (1974)
Coastal New Jersey marsh	observations of prey sizes by month	60%	12%	18%	11%	6%	7%	49%	38%	Willard (1977)
Average		-	-	-	-	**17%**	**29%**	**40%**	**14%**	

[a] Fish size class portions based on wet weight were estimated using the species-specific length-to-weight calculator from FishBase (www.fishbase.org).

Exposure Area

Great blue herons have a higher preference for emergent vegetation/natural shoreline features, and a lower preference for developed habitat areas with limited vegetation (e.g., riprap) (Short and Cooper 1985; Granholm 2008); however, great blue heron were found throughout the LPRSA during the avian surveys (Windward 2011a, 2019e; BBL 2002), and they showed no preference among fresh, brackish, or saltwater habitats

[120] Where lengths were provided for species, the closest available taxa or body type from FishBase was applied. Otherwise, an average was estimated using the minnow family length-weight calculation as an intermediate body type.

(Kushlan 1978; Willard 1977; Chapman and Howard 1984). Great blue heron use both exposed sediment and shoreline and tend to hunt for prey only in shallow water; as a result, the sediment exposure area for the dietary assessment of great blue heron was limited to mudflat areas (≤ 2 ft mean lower low water [MLLW]). Several studies on great blue heron home ranges (i.e., distance from breeding grounds to foraging areas) in various locations throughout North America have reported that foraging grounds are generally within 0 to 8 km of breeding colonies, and that 15 to 20 km is generally the farthest that great blue herons will travel from the colony to foraging areas (USEPA 1993). Based on several studies in which distance from colonies to foraging areas ranged from 0.55 to 34.1 km, Henning et al. (1999) observed the median distance traveled by great blue herons to foraging sites to be 12 km. Because of the great blue heron's large foraging range, the exposure area included the entire LPRSA. Risks to great blue heron were also evaluated assuming that the species forages from only specific river reaches (USEPA 2015b, c, 2016g). Because there are other foraging areas in the vicinity of the LPRSA (e.g., Kearny Marsh), it is likely that great blue heron do not feed exclusively from the LPRSA. Therefore, risks are likely overestimated by using an SUF of 1 in Equation 8-1; this is reflected in the uncertainty evaluation that calculates HQs using an SUF of 0.5.

Belted Kingfisher

Prey Composition

The belted kingfisher diet consists primarily of fish, but the species has also been known to eat crustaceans, mollusks, insects, amphibians, reptiles, young birds, small mammals, and berries (Prose 1985; Kelly et al. 2009). Empirical data from the literature based on the dietary composition of belted kingfisher are summarized in Table 8-7. No amphibians or crayfish were found in LPRSA sampling, therefore, the diet of belted kingfisher in the LPRSA for the risk calculations was assumed to consist of 85% fish and 15% crab based on the general estimates reported in the literature.

Table 8-7. Summary of belted kingfisher prey items reported in the literature

Location	Sample Type/Size	Reported Prey	Citation
Housatonic River, Connecticut	prey items in nests	86% fish (minnows, sunfish, and perch); 14% crayfish	ARCADIS (2002)
Near Ithaca, New York	stomach contents	74% fish; 13.4% crayfish; 5.6% reptiles/amphibians; 7% insects (based on frequency of occurrence)	Gould (1934), as cited in Salyer and Lagler (1949)
United States	stomach contents	75% fish; 16% crayfish; 5% amphibians (frogs); 4% insects (based on frequency of occurrence)	Howell (1932)
Maritime Nova Scotia along 23 waterways	regurgitated pellets	94% fish; 5% crayfish; 1% insects (mean values based on frequency of occurrence)	White (1953)

Location	Sample Type/Size	Reported Prey	Citation
Ohio stream habitats	prey presented to nestlings	87% fish; 13% crayfish	Davis (1982)
Michigan rivers, lakes, and streams	stomach contents	46–95% fish; 5–55% invertebrates; 0–27% amphibians; 0–1% mammals (based on weight)	Alexander (1976); Salyer and Lagler (1949)
Lake Itasca, Minnesota	prey dropped during nesting season	100% fish	Cornwell (1963)

A belted kingfisher's prey ranges from 2.5 to 17.8 cm in length but is generally less than 10 cm (Bent 1940; Salyer and Lagler 1949). Fish as long as 9 cm occurred in more than 90% of the stomach contents of belted kingfishers from Michigan rivers and lakes (Alexander 1976). For modeling the belted kingfisher diet in Scenario 1 of the risk calculations (Table 8-4), the fish portion of the species' prey was assumed to be comprised of fish as long as 9 cm, since the majority of their prey is small fish.

Larger fish were also considered as potential prey for belted kingfisher (USEPA 2015b, c, 2016g). Three general size classes of fish were determined based on the reported prey sizes, which ranged from 2.5 to 18 cm: 0 to 9 cm; 9 to 13 cm; and 13 to 18 cm. Quantitative data from the literature were evaluated to determine appropriate prey portions of fish within these general size classes. Data from the literature were reported as frequency of occurrence within size classes and were converted to weight for those size classes using the species-specific length-to-weight calculator from FishBase (www.fishbase.org)[121] (Table 8-8). On average, the percentages of fish, by weight, in the belted kingfisher's diet were 37% fish 0 to 9 cm, 60% fish 9 to 13 cm, and 3% fish 13 to 18 cm. For Scenario 2 of the risk calculations, these percentages were applied to the overall portion of fish in the belted kingfisher diet (85%) (Table 8-4).

[121] Where lengths were provided for species, the closest available taxa or body type from FishBase was applied. Otherwise, an average was estimated using the minnow family length-weight calculation as an intermediate body type.

Table 8-8. Summary of belted kingfisher prey items reported in the literature

Location	Sample Type/Size	Fish Size Class Portion (based on frequency of occurrence)			Fish Size Class Portion (based on wet weight)[a]			Citation
		0–9 cm	9–13 cm	13–18 cm	0–9 cm	9–13 cm	13–18 cm	
Michigan rivers, lakes, and streams	stomach contents	91%	9%	0%	57%	43%	0%	Alexander (1976)
Lake Itasca, Minnesota	prey dropped during nesting season	54%	46%	0%	38%	62%	0%	Cornwell (1963)
Ohio stream habitats	prey presented to nestlings	40%	57%	3%	17%	74%	9%	Davis (1982)
	Average	-	-	-	**37%**	**60%**	**3%**	

a Fish size class portions based on wet weight were estimated using the species-specific length-to-weight calculator from FishBase (www.fishbase.org).

Exposure Area

Belted kingfisher were found throughout the LPRSA during the avian surveys (Windward 2011a, 2019e; BBL 2002; Ludwig et al. 2010); however, belted kingfisher prefer shoreline areas with trees for perching and have a low preference for areas with scrub-shrub or limited vegetation (Kelly et al. 2009; Cornwell 1963; Windward 2011a, 2019e). The literature suggests that because belted kingfisher are sensitive to disturbances, they usually do not nest in areas that have significant human activity (Cornwell 1963). They are generally restricted to areas with clear water and prefer steep, silted banks for nesting (Kelly et al. 2009). In a survey conducted by USACE, NJDOT, and NOAA in 2006 to identify belted kingfisher burrows in the lower 16 mi of the Passaic River, a total of nine belted kingfisher burrows were found along the LPRSA: two near RM 4, one at RM 7.5, one at RM 8.5, four between RM 11.1 and RM 11.4, and one at RM 13.1 (Baron 2011). However, none of the burrows were active and most showed evidence of mammal use (Baron 2011). In general, belted kingfisher breeding habitat was found to be limited in the lower 6 mi of the Passaic River (Baron 2011); areas above the lower 6 mi have suitable habitat and may support breeding pairs (Ludwig et al. 2010). Therefore, an exposure area for belted kingfisher limited to areas upstream of RM 6 was evaluated. Risks to belted kingfisher were also evaluated assuming that the species forages from the entire LPRSA, as well as assuming that it forages from only specific river reaches (Table 8-5) (USEPA 2015b, c, 2016g).

Belted kingfishers are highly territorial; they have a home range of approximately 0.93 (Cornwell 1963) to 2.19 km (Bent 1940; as cited in Kelly et al. 2009) during the breeding season, and a generally smaller home range during the rest of the year (TAMS 1999). Brooks and Davis (1987) observed similar belted kingfisher territory sizes in Ohio and Pennsylvania, ranging between 1.0 and 2.2 km. Therefore, the use of an SUF of 1 in Equation 8-1 is reasonable for belted kingfisher.

8.1.2.4 Exposure point concentrations

To calculate dietary doses, EPCs were calculated for each of the media types ingested (i.e., prey, surface water, and sediment) by each bird species evaluated for use in Equation 8-1. For prey concentrations, EPCs were calculated separately for each of the prey types (worms, blue crab, and fish in specified size categories) for the bird species for the evaluation of the different dietary scenarios. In addition to grouping by prey type and size, data were grouped by exposure area, as indicated in Table 8-9. For each of the data groups in Table 8-9, EPCs were calculated as the UCL using USEPA's ProUCL® statistical package (Version 5.0.00) (USEPA 2013d), as described in Section 4.3.7.[122] If a dataset had fewer than six samples, a UCL was not calculated; instead, the maximum concentration was used as the EPC. There is greater uncertainty in the estimated exposure concentration when sample sizes are small; it is possible that using the maximum concentration could overestimate exposure. The summary statistics, UCLs, and selected EPCs for each of the data groups in Table 8-9 are included in Appendix C. Uncertainties associated with the use of non-detects in calculations of total PAHs, total PCBs, and TEQs - bird are discussed in Section 8.1.4.2.

Table 8-9. Data groups for calculation of EPCs in prey, surface water, and sediment

Species and Exposure Area	Prey Type and Exposure Area			Surface Water Exposure Area	Sediment Exposure Area
	Prey Type[a,b]	% in Diet	Exposure Area		
Spotted sandpiper: site wide					
Scenario 1	worms	100	site-wide mudflats	RM ≥ 4[c]	site-wide mudflats
Spotted sandpiper: reach specific					
Scenario 1	worms	100	by reach	by reach[d]	mudflats by reach
Great blue heron: site wide					
Scenario 1	fish 0–13 cm	100	site-wide mudflats	RM ≥ 4[c]	site-wide mudflats
Scenario 2	fish 0–13 cm	17	site-wide mudflats	RM ≥ 4[c]	site-wide mudflats
	fish 13–18 cm	29			
	fish 18–30 cm	40			
	fish > 30 cm	14			
Great blue heron: reach specific					
Scenario 1	fish 0–13 cm	100	mudflats by reach	by reach[d]	mudflats by reach

122 The UCL recommended by USEPA's ProUCL® software (typically the 95% UCL, but in some cases the 97.5 or even the 99% UCL) was used. ProUCL® creates interpolated values for non-detects based on the distribution of the detected concentrations. The selected UCL statistic is presented in Appendix C1.

Table 8-9. Data groups for calculation of EPCs in prey, surface water, and sediment

Species and Exposure Area	Prey Type and Exposure Area			Surface Water Exposure Area	Sediment Exposure Area
	Prey Type[a,b]	% in Diet	Exposure Area		
Scenario 2	fish 0–13 cm	17	mudflats by reach	by reach[d]	mudflats by reach
	fish 13–18 cm	29			
	fish 18–30 cm[e]	40			
	fish > 30 cm	14			
Belted kingfisher: RM ≥ 6					
Scenario 1	blue crab[f]	15	RM ≥ 6	RM ≥ 6	RM ≥ 6
	fish 0–9 cm	85			
Scenario 2	blue crab[f]	15	RM ≥ 6	RM ≥ 6	RM ≥ 6
	fish 0–9 cm	31.5			
	fish 9–13 cm	51			
	fish 13–18 cm	2.5			
Belted kingfisher: site wide					
Scenario 1	blue crab[f]	15	site wide	RM ≥ 4[c]	site wide
	fish 0–9 cm	85			
Scenario 2	blue crab	15	site wide	RM ≥ 4[c]	site wide
	fish 0–9 cm	31.5			
	fish 9–13 cm	51			
	fish 13–18 cm	2.5			
Belted kingfisher: by reach					
Scenario 1	blue crab[g]	15	by reach	by reach[d]	by reach
	fish 0–9 cm[h]	85			
Scenario 2	blue crab[g]	15	by reach	by reach[d]	by reach
	fish 0–9 cm[g]	31.5			
	fish 9–13 cm	51			
	fish 13–18 cm[i]	2.5			

Note: If fewer than six samples were available for calculating a UCL, the maximum concentration was used.

a As represented by whole-body tissue concentrations.

b For composite fish samples, length is based on the maximum length of any fish in the sample.

c Includes only freshwater.

d Surface water data were available for only Reaches 3 through 6. Therefore, in Reaches 1 and 2, surface water EPCs for Reach 3 were used, and in Reaches 7 and 8, surface water EPCs for Reach 6 were used.

e Data for fish in the 13–18 cm category were not available for Reaches 1 and 2, so EPCs for fish in the 18–30 cm category were substituted in those reaches.

f Data for blue crab were not available for Reaches 6 through 8, so data from Reaches 1 through 5 were used to calculate the EPCs for the ≥ RM 6 and site- wide exposure areas.

g Data for blue crab were not available for Reaches 6 through 8, so data from Reach 5 were used to calculate the EPCs for those reaches.

h Data for fish in the 0–9 cm category were not available in Reaches 1, 7, and 8, so EPCs for fish in the 9–13 cm category were substituted in those reaches.

i Data for fish in the 13–18 cm category were not available in Reaches 1 and 2, so EPCs for fish in the 9–13 cm category were substituted those reaches.

EPC – exposure point concentration
RM – river mile

8.1.2.5 Estimated doses

Dietary doses were calculated for site-wide exposures based on Equation 8-1 using the prey, sediment, and surface WIRs and body weights from Table 8-3, the prey composition from Table 8-4, and the EPCs for prey, sediment, and surface water (based on UCLs or the maximum concentrations if there were fewer than six detected values) from Appendix C. These doses are presented in Table 8-10.

Table 8-10. Dietary doses calculated for spotted sandpiper, great blue heron, and belted kingfisher

COPEC	Units	Exposure Area	Dietary Dose				
			Spotted Sandpiper Diet Scenario	Great Blue Heron Diet Scenario		Belted Kingfisher Diet Scenario	
			1	1	2	1	2
Cadmium	mg/kg bw/day	site wide	0.23	0.0086	0.0082	0.031	0.029
		RM ≥ 6	na	na	na	0.032	0.037
		by reach	0.10–0.44	0.0045–0.021	0.0023–0.016	0.014–0.044	0.018–0.050
Chromium	mg/kg bw/day	site wide	19	2.0	3.0	5.3	5.8
		RM ≥ 6	na	na	na	6.2	8.4
		by reach	5.8–41	0.77–5.5	0.30–6.5	0.55–14	2.1–13
Copper	mg/kg bw/day	site wide	9.5	0.62	1.8	3.4	3.4
		RM ≥6	na	na	na	3.4	3.7
		by reach	5.6–16.7	0.60–1.1	0.65–6.3	2.9–4.3	2.9–4.6
Lead	mg/kg bw/day	site wide	14	0.51	0.29	1.3	1.2
		RM ≥ 6	na	na	na	1.6	1.4
		by reach	5.7–19	0.19–1.0	0.11–0.43	0.47–2.1	0.47–1.9
Methylmercury	µg/kg bw/day	site wide	2.0	9.6	24	24	33
		RM ≥ 6	na	na	na	21	22
		by reach	0.69–2.5	3.0–12	23–41	12.0–41	14–41

Table 8-10. Dietary doses calculated for spotted sandpiper, great blue heron, and belted kingfisher

COPEC	Units	Exposure Area	Dietary Dose				
			Spotted Sandpiper Diet Scenario	Great Blue Heron Diet Scenario		Belted Kingfisher Diet Scenario	
			1	1	2	1	2
Nickel	mg/kg bw/day	site wide	9.3	1.2	2.1	3.4	3.9
		RM ≥ 6	na	na	na	3.9	5.7
		by reach	2.7–24	0.55–3.9	0.20–4.7	0.5–9.5	1.4–9.3
Selenium	mg/kg bw/day	site wide	0.38	0.12	0.17	0.36	0.36
		RM ≥ 6	na	na	na	0.36	0.34
		by reach	0.21–0.4	0.05–0.16	0.12–0.26	0.17–0.39	0.18–0.42
Vanadium	mg/kg bw/day	site wide	1.2	0.14	0.07	0.41	0.35
		RM ≥ 6	na	na	na	0.45	0.47
		by reach	1.0–2.1	0.07–0.24	0.04–0.11	0.11–0.58	0.19–0.58
Zinc	mg/kg bw/day	site wide	40	7.6	5.6	22	21
		RM ≥ 6	na	na	na	21	20
		by reach	25–59	4.5–8.5	5.3–6.8	14–25	14–24
Total HPAHs	µg/kg bw/day	site wide	2341	89	50	173	241
		RM ≥ 6	na	na	na	194	291
		by reach	917–4795	21–122	21–81	55–384	66–378
Total LPAHs	µg/kg bw/day	site wide	591	22	31	51	69
		RM ≥ 6	na	na	na	50	89
		by reach	124–964	8.0–67	25–43	20–165	25–161
Benzo(a)pyrene	µg/kg bw/day	site wide	402	7.0	4.9	17	22
		RM ≥ 6	na	na	na	19	26
		by reach	112–478	2.5–19.3	1.6–7.7	5.6–29	7.4–38
Total PAHs	µg/kg bw/day	site wide	2840	97	76	221	311
		RM ≥ 6	na	na	na	244	367
		by reach	1080–5981	29–207	43–121	74–494	93–482

Table 8-10. Dietary doses calculated for spotted sandpiper, great blue heron, and belted kingfisher

COPEC	Units	Exposure Area	Dietary Dose: Spotted Sandpiper Diet Scenario 1	Dietary Dose: Great Blue Heron Diet Scenario 1	Dietary Dose: Great Blue Heron Diet Scenario 2	Dietary Dose: Belted Kingfisher Diet Scenario 1	Dietary Dose: Belted Kingfisher Diet Scenario 2
Total PCBs	µg/kg bw/day	site wide	241	98	330	353	293
		RM ≥ 6	na	na	na	357	312
		by reach	65–575	44–165	155–569	130–445	161–340
PCB TEQ - bird	ng/kg bw/day	site wide	43	9.3	25	34	33
		RM ≥ 6	na	na	na	35	39
		by reach	10–109	3.5–15	9.0–45	14–40	17–43
PCDD/PCDF TEQ - bird	ng/kg bw/day	site wide	127	9.4	27	13	13
		RM ≥ 6	na	na	na	40	33
		by reach	2.0–584	2.7–20	7.2–52	15–53	14–40
Total TEQ - bird	ng/kg bw/day	site wide	152	9.4	51	70	60
		RM ≥ 6	na	na	na	75	70
		by reach	12–703	6.3–34	22–99	25–88	18–79
Total DDx	ng/kg bw/day	site wide	17	31	35	40	34
		RM ≥ 6	na	na	na	39	36
		by reach	4.3–38	5.1–15	18–65	17–49	19–41

bw – body weight
COPEC – chemical of potential ecological concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
LPAH – low-molecular-weight polycyclic aromatic hydrocarbon
na – not applicable (not a COPEC for this species)
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
PCDD – polychlorinated dibenzo-*p*-dioxin
PCDF – polychlorinated dibenzofuran
TEQ – toxic equivalent
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT and 4,4′-DDT)

8.1.3 Effects

This section presents the effects data (i.e., TRVs) selected from the toxicological literature for the COPECs that were screened into this BERA based on the SLERA. A range of TRVs was evaluated. The selection was based on a comprehensive review of the primary literature and an assessment of acceptability. Selected TRVs were also based on previous documents developed by USEPA Region 2 for the LPRSA. Selected TRVs are consistent with the comments, responses, and directives received from

USEPA on June 30, 2017 (USEPA 2017b), September 18, 2017 (USEPA 2017e), July 10, 2018 (USEPA 2018), January 2, 2019 (CDM 2019), and March 5, 2019 (USEPA 2019); during face-to-face meetings or conference calls on July 24, September 27, October 3, November 6, 2017, July 24, 2018, and August 16, 2018; and via additional deliverables and communications between the Cooperating Parties Group (CPG) and USEPA from August through December 2017, July through September 2018, and January through June 2019.

8.1.3.1 Methods for selecting TRVs

Two sets of bird dietary TRVs were used for the derivation of HQs in this BERA. One set of TRVs was based on previous documents developed by USEPA Region 2 for the LPRSA:

- USEPA's revised draft of the LPR restoration project FFS (Louis Berger et al. 2014)
- USEPA's first draft of the LPRSA FFS (Malcolm Pirnie 2007b)
- USEPA's LPR restoration project PAR (Battelle 2005)

The second set of TRVs was based on a comprehensive review of the primary literature and an assessment of acceptability. TRVs were selected by first conducting a literature search for relevant toxicological studies. These studies were then evaluated for acceptability of use. For those studies considered acceptable, NOAEL and LOAEL daily doses were derived as described in Appendix E. TRVs were then selected for each COPEC based on an evaluation of all the acceptable NOAEL and LOAEL TRVs. Details regarding the literature search and study acceptability are presented in Appendix E. The TRV derivation and selection processes, along with general uncertainties regarding the use of TRVs to estimate risk, are described in more detail below.

TRV Derivation

Dietary TRVs for birds were expressed as a daily dose in mg/kg bw/day. However, many studies reported toxicity results as the chemical concentration in food associated with adverse effects, rather than as a daily dose. If the daily exposure dose was not presented in a study, it was derived using the reported concentration in food, the animal's body weight (kg), either the ingestion rate (kg/day) reported in the study or in the scientific literature, and the following equation:

$$TRV = \frac{C_{diet} \times IR}{BW} \qquad \textbf{Equation 8-3}$$

Where:

TRV	=	toxicity reference value (mg/kg bw/day)
C_{diet}	=	chemical concentration in diet (mg/kg)
IR	=	ingestion rate (kg/day)
BW	=	body weight (kg)

Detailed information regarding the conversion of dietary concentrations reported in the literature to body weight-normalized TRVs is presented in Appendix E.

TRV Selection Process

The first step in selecting TRVs was to determine if sufficient toxicity data were available to derive an SSD (i.e., n ≥ 5 species). An SSD is a statistical model that can be used to calculate a chemical concentration protective of a predetermined percentage of a group of species. SSDs are intended to provide an indication of both the total range and distribution of species sensitivities in natural communities, even when the actual range of sensitivities is unknown (Stephan 2002). In practice, SSDs are most commonly presented as a CDF of the toxicity of a chemical to a group of laboratory test species. The 5th percentile of the distribution was selected as the LOAEL TRV (estimated to protect 95% of the bird species present in the LPRSA), and the NOAEL TRV was extrapolated from the LOAEL TRV using an uncertainty factor of 10. Additional details on the SSD approach are presented in Section 7.1.3.1. If toxicity data were not available for at least five species to derive an SSD, the lowest appropriate LOAEL was selected.

TRV Uncertainty

Some of the general uncertainties associated with using laboratory toxicity studies to estimate potential effects on wildlife species that should be considered in the interpretation of risk characterization results include the following:

- The reported adverse effects do not necessarily occur in all species (i.e., there could be species-specific responses).
- The concentrations that elicit adverse effects can vary greatly among species.
- The concentrations and exposure conditions that elicit adverse effects in the laboratory may not be representative of concentrations and conditions to which wildlife are exposed *in situ.*
- Some of the endpoints evaluated and reported in laboratory studies may not have meaningful relevance to an organism's likelihood of surviving or successfully reproducing.
- Even if individual organisms experience ecologically relevant effects, these effects may not be predictive of population-level responses in a complete ecological context in which time-varying exposures among individuals and population dynamics are also functions of the effects of other co-occurring natural and anthropogenic stressors. This is particularly important since this BERA, in accordance with USEPA guidance, is an assessment of population-level risks.
- There is uncertainty associated with the use of an extrapolation factor to derive a NOAEL.

COPEC-specific uncertainties associated with bird diet TRVs are discussed in the following section (Section 8.1.3.2).

8.1.3.2 Selected TRVs for birds

The bird dietary TRVs are presented in Table 8-11, followed by a discussion of the derivation of the range of TRVs.

Table 8-11. Bird dietary TRVs

COPEC	Units	Range of TRVs[a]								
		TRV-A[b]				TRV-B[c]				
		NOAEL	LOAEL	Endpoint	Source	NOAEL	LOAEL	Endpoint	Source	Document
Metals										
Cadmium	mg/kg bw/day	0.4[d]	4.0	growth (quail)	Richardson et al. (1974)	0.080	10.4	growth (quail)	Richardson et al. (1974), as cited in USEPA (2002f)	2005 PAR (Battelle 2005)
Chromium	mg/kg bw/day	10.5[d]	105	survival and growth (chicken)	Chung et al. (1985)	nd	nd	nd	nd	nd
Copper	mg/kg bw/day	1.9[d]	19	growth (chicken)	Jensen and Maurice (1978)	2.3	4.7	growth (turkey)	Kashani et al. (1986), as cited in USEPA (2007c)	revised FFS (Louis Berger et al. 2014)
Lead	mg/kg bw/day	5.5	28	growth (quail)	Morgan et al. (1975)	0.19	1.9	reproduction (quail)	Edens and Garlich (1983), as cited in USEPA (2005b)	revised FFS (Louis Berger et al. 2014)
Methylmercury	µg/kg bw/day	9.6[d]	96	survival, growth, and reproduction (6 species)	SSD-derived 5th percentile value	13	26	reproduction (duck)	Heinz 1974, 1975, (1979), as cited in USEPA (1995a)	revised FFS (Louis Berger et al. 2014)
Nickel	mg/kg bw/day	15	33	growth (chicken)	Weber and Reid (1968)	1.38	56.3	growth (duck)	Cain & Pafford (1981), as cited in USEPA (2002f)	2005 PAR (Battelle 2005)
Selenium	mg/kg bw/day	0.42	0.82	reproduction (duck)	Heinz et al. (1989)	0.23	0.93	reproduction (duck)	Heinz et al. (1989), as cited in USEPA (2002f)	2005 PAR (Battelle 2005)
Vanadium	mg/kg bw/day	1.2	2.3	growth (chicken)	Ousterhout and Berg (1980)	nd	nd	nd	nd	nd
Zinc	mg/kg bw/day	82	124	growth (chicken)	Roberson and Schaible (1960)	17.2	172	growth/ reproduction (duck)	Gasaway & Buss (1972), as cited in USEPA (2002f)	2005 PAR (Battelle 2005)

Table 8-11. Bird dietary TRVs

COPEC	Units	Range of TRVs[a]								
		TRV-A[b]				TRV-B[c]				
		NOAEL	LOAEL	Endpoint	Source	NOAEL	LOAEL	Endpoint	Source	Document
PAHs										
Total LPAHs	µg/kg bw/day	nd	nd	nd	nd	670	6,700	survival (blackbird)	Schafer et al. (1983)	revised FFS (Louis Berger et al. 2014)
Total HPAHs	µg/kg bw/day	nd	nd	nd	nd	48	480	reproduction (pigeon)	Hough et al. (1993)	revised FFS (Louis Berger et al. 2014)
Benzo(a)pyrene (HPAH)	µg/kg bw/day	140[d]	1,400	reproduction (pigeon)	Hough el al. (1993)	nd	nd	nd	nd	nd
Total PAHs	µg/kg bw/day	40,000	na	growth (duck)	Patton and Dieter (1980)	nd	nd	nd	nd	nd
PCDDs/PCDFs										
TEQ - bird	ng/kg bw/day	14	140	mortality/ growth/ reproduction (pheasant)	Nosek et al. (1992)	2.8	28	reproduction (pheasant)	Nosek et al. (1992), as cited in USEPA (1995a)	revised FFS (Louis Berger et al. 2014)
PCBs										
Total PCBs	µg/kg bw/day	140[d]	1,400	reproduction (dove)	Peakall et al. (1972); Peakall and Peakall (1973)	400	500	reproduction (chicken)	Chapman (2003)	revised FFS (Louis Berger et al. 2014)
Organochlorine Pesticides										
Total DDx	µg/kg bw/day	25[d]	250	survival, growth, and reproduction (10 species)	SSD-derived 5th percentile value	0.9	27	reproduction (pelican)	Anderson et al. (1975), as cited in USEPA (1995a)	revised FFS (Louis Berger et al. 2014)

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

[b] TRVs were derived from the primary literature review based on process identified in Section 8.1.3.1.

[c] TRVs were based on USEPA's revised draft of the LPR restoration project FFS (Louis Berger et al. 2014), first draft of the LPR restoration project FFS (Malcolm Pirnie 2007b), or LPR restoration project PAR (Battelle 2005).

[d] NOAEL extrapolated from LOAEL using an uncertainty factor of 10.

BERA – baseline ecological risk assessment
bw – body weight
COPEC – chemical of potential ecological concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
FFS – focused feasibility study
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
LPAH – low-molecular-weight polycyclic aromatic hydrocarbon
LOAEL – lowest-observed-adverse-effect level
LPR – Lower Passaic River
na – not available
nd – not derived
NJDEP – New Jersey Department of Environmental Protection
NOAEL – no-observed-adverse-effect level
PAH – polycyclic aromatic hydrocarbon
PAR – pathways analysis report
PCB – polychlorinated biphenyl
PCDD – polychlorinated dibenzo-*p*-dioxin
PCDF – polychlorinated dibenzofuran
SSD – species sensitivity distribution
TEQ – toxic equivalent
TRV – toxicity reference value
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT and 4,4′-DDT)
USEPA – US Environmental Protection Agency

Cadmium

Six acceptable studies were available that evaluated the toxicity of cadmium to birds. These studies were conducted with only three species (chickens, mallards, and Japanese quail), and data were therefore not sufficient for calculating an SSD. The lowest LOAEL of 4.0 mg/kg bw/day resulted in Japanese quail body weights that were 15% less than the controls after a six-week exposure to cadmium in the diet (Richardson et al. 1974). This LOAEL was selected as the LOAEL TRV. It is unclear whether a 15% reduction in growth would result in adverse effects on the population; therefore, this LOAEL represents a conservative estimate of possible adverse effect. Because a NOAEL was not available from this study, the NOAEL TRV was estimated as 0.4 mg/kg bw/day using an uncertainty factor of 10. There is uncertainty associated with the use of an extrapolation factor to derive a NOAEL.

A LOAEL TRV of 10.4 mg/kg ww and a NOAEL TRV of 0.08 mg/kg ww were also selected for cadmium (Battelle 2005) based on Richardson et al. (1974) and USEPA (2002f). The LOAEL of 10.4 mg/kg bw/day was based on a mid-range adverse effect level using a cellular endpoint (testicular development) based on data from Richardson et al. (1974).

Chromium

Four acceptable toxicity studies were available that evaluated the effects of dietary chromium ingestion on birds; all of these studies were conducted with chickens. One additional study conducted with mallards (Haseltine et al. unpublished) was cited in Sample et al. (1996), but the data are unpublished and unavailable for review. The unpublished study (Haseltine et al. unpublished) reportedly found reproductive effects on American black duck (*Anas rubripes*) at a dietary dose of 5 mg/kg bw/day. This effect level was the lowest LOAEL among the five chromium studies. However, because the original data could not be reviewed, this value was not selected as the LOAEL TRV. The lowest published TRV of 105 mg/kg bw/day, based on adverse effects on growth and survival in chickens, was selected as the LOAEL TRV (Chung et al. 1985). There were no published studies available for review that found adverse effects on bird reproduction. There was no NOAEL from the study with the selected LOAEL, so a NOAEL TRV of 10.5 mg/kg bw/day was derived from the LOAEL TRV using a factor of 10. There is uncertainty associated with the use of an extrapolation factor to derive a NOAEL.

No TRVs for chromium were available from the draft FFS (Malcolm Pirnie 2007b) or revised FFS (Louis Berger et al. 2014).

Copper

TRVs for copper were derived from the review of eight acceptable studies on the toxicity of copper to birds. All of these studies were conducted with chickens, so an SSD could not be derived. The LOAEL of 19 mg/kg bw/day was selected because it was the lowest LOAEL among the eight studies. This LOAEL represents the dose for a growth endpoint at which the body weight of chicks was reduced by 10% compared to the control after exposure to copper in the diet for four weeks (Jensen and Maurice 1978). It is unknown whether a 10% reduction in growth would result in adverse effects on the population; therefore, this LOAEL represents a conservative estimate of a possible adverse effect. The NOAEL TRV of 1.9 mg/kg bw/day was derived from the LOAEL TRV using a factor of 10 because there was no dose below the LOAEL in the study. There is uncertainty associated with the use of an extrapolation factor to derive a NOAEL.

A LOAEL TRV of 4.7 mg/kg ww and a NOAEL TRV of 2.3 mg/kg ww were also selected for copper from previous documents developed by USEPA Region 2 for the LPRSA (Louis Berger et al. 2014) based on USEPA's ecological soil screening level (Eco-SSL) document for copper USEPA (2007c). These TRVs, as cited, were based on turkey growth using data from Kashani et al. (1986). The LOAEL TRV was based on a 4% decrease in body weight in turkeys compared to the control at 8 weeks, with a recovery in body weight at 12 weeks, and no effect on body weight at 12, 16, 20, or 24 weeks (Kashani et al. 1986). Similar to the LOAEL based on chick growth (Jensen and Maurice 1978), it is unknown whether this small reduction in growth (4%) would result in adverse effects on the population (e.g., translate to exposed individuals being less competitive or more prone to predation throughout this critical life stage); therefore, this LOAEL represents a highly conservative estimate of a possible adverse effect.

Lead

TRVs for lead were derived from a review of seven studies to evaluate the toxicity of lead to birds. Additional studies were available; however, they were not considered acceptable because they exposed birds through oral intubation or intra-peritoneal injection. The four dietary studies used mallards, American kestrels, ringed turtle doves, and Japanese quail. Data were insufficient for the development of an SSD. Among the four acceptable studies, the lowest LOAEL was based on a five-week test that evaluated growth in Japanese quail exposed to lead in their diet (Morgan et al. 1975). The LOAEL of 28 mg/kg bw/day resulted in an approximate 10% decrease in body weight compared to the control. The other three dietary studies (Finley et al. 1976; Pattee 1984; Kendall and Scanlon 1981) did not observe any adverse effects on growth, survival, or reproduction at the highest doses tested, which ranged from 2.5 to 10.9 mg/kg bw/day. The LOAEL TRV from Morgan et al. (1975) was selected along with the NOAEL TRV of 5.5 mg/kg bw/day from the same study (Table 8-11). It is unknown whether a 10% reduction in growth would result in adverse effects on the population; therefore, this LOAEL represents a conservative estimate of a possible adverse effect.

NOAEL and LOAEL TRVs of 0.19 and 1.9 mg/kg bw/day, respectively, were also selected for lead from previous documents developed by USEPA Region 2 for the LPRSA (Louis Berger et al. 2014) based on USEPA's Eco-SSL document for lead USEPA

(2005b). These TRVs were based on Japanese quail egg production using data from Edens and Garlich (1983). TRVs based on domestic reproductive endpoints are uncertain because domesticated species (e.g., chickens and quails) are bred to have very high egg-laying rates, and it is not evident that effects noted in Japanese quail egg production rates would reflect adverse effects on reproduction in wild birds as appropriate for the LPRSA.

Methylmercury

A TRV for methylmercury was derived based on a review of 11 acceptable studies from the literature that evaluated the effects of exposure to methylmercury in the diet on birds. LOAELs were available for effects on growth, reproduction, and mortality for six species (American kestrel, great egret, Japanese quail, mallard, northern bobwhite, and zebra finch) and range from 64 to 8,780 µg/kg bw/day. An SSD was developed to derive a TRV (Figure 8-1). The 5th percentile determined from the SSD was 96 µg/kg bw/day; this value was used as a TRV for methylmercury. This SSD-derived LOAEL is within the range of measured LOAELs reported from the literature. The NOAEL TRV (9.6 µg/kg ww) was extrapolated from the LOAEL TRV using an uncertainty factor of 10. There is uncertainty associated with the use of an extrapolation factor to derive a NOAEL.

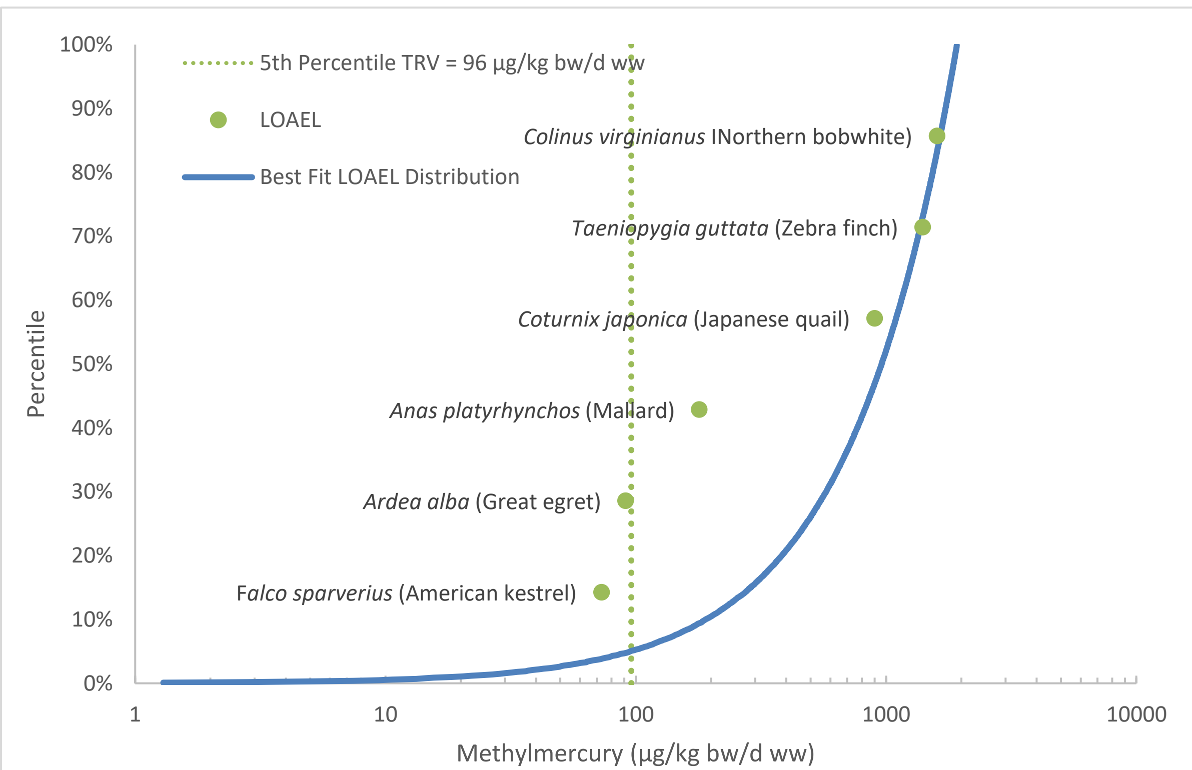


Figure 8-1.SSD derived from bird dietary toxicity data for methylmercury

A NOAEL and LOAEL of 13 and 26 µg/kg bw/day, respectively, were selected for methylmercury from previous documents developed by USEPA Region 2 for the

LPRSA (Louis Berger et al. 2014) based on USEPA (1995a). These TRVs were based on mallard reproduction using data from Heinz (1979, 1974, 1975). Uncertainty factors were used to derive the values (Louis Berger et al. 2014). The LOAEL TRV was derived from the reported LOAEL based on an interspecies extrapolation factor of three in Louis Berger et al. (2014), which assumes that mallards are three times less sensitive than the selected avian species evaluated. Additional uncertainty is associated with this TRV, because it is based on the use of methylmercury dicyandiamide, a fungicide that is not a form of mercury expected to be associated with the LPRSA.

Nickel

TRVs for nickel were based on a review of three acceptable toxicity studies conducted with chickens and mallard ducks. Among the three acceptable studies, the lowest LOAEL was based on reduced growth in chickens after a four-week exposure to nickel in the diet (Weber and Reid 1968). The LOAEL of 33 mg/kg bw/day resulted in a 31% decrease in body weight compared to the control. The NOAEL from this same study was 15 mg/kg bw/day. The NOAEL and LOAEL from this study were selected as the TRVs for nickel.

A NOAEL and LOAEL of 1.38 and 56.3 mg/kg bw/day, respectively, were also selected for nickel (Louis Berger et al. 2014). These values were from the USEPA Region 9 Biological Technical Assistance Group (BTAG) TRVs (USEPA 2002f) and were based on mallard growth using data from Cain and Pafford (1981).

Selenium

TRVs for selenium were derived based on a review of six toxicity studies that were conducted with chickens, mallard ducks, and screech owls. Sufficient data were not available to develop an SSD. The lowest dose at which an effect was observed in these six studies was 0.82 mg/kg bw/day, from a study that exposed mallard ducks to selenium in their diet for 100 days (Heinz et al. 1989). At 0.82 mg/kg bw/day, a significant effect on offspring growth and survival was observed compared to the control; no effects were observed at the next lowest dose of 0.42 mg/kg bw/day. Therefore, the NOAEL and LOAEL TRVs for selenium are 0.42 and 0.82 mg/kg bw/day, respectively. The TRVs derived from this study are the same as those reported by Sample et al. (1996).

A NOAEL and LOAEL of 0.23 and 0.93 mg/kg bw/day, respectively, were also selected for selenium (Battelle 2005). These values were USEPA Region 9 BTAG TRVs (USEPA 2002f) and were based on Heinz et al. (1989).

Vanadium

Three acceptable studies were found that evaluated the effects of dietary vanadium on birds; two of these studies were performed with chickens (Ousterhout and Berg 1980; Davis et al. 2002) and one with mallards (White and Dieter 1978). An SSD could not be developed with the available data. Only one of the three studies reported adverse effects for the endpoints monitored; in Ousterhout and Berg (1980), the body weight of chickens decreased by approximately 10%[123] in birds fed 2.3 mg/kg bw/day, but there was no significant decrease at a dose of 1.2 mg/kg bw/day. These values were selected for use as the NOAEL and LOAEL TRVs (Table 8-11). It is unknown whether a 10% reduction in growth would result in adverse effects on the population; therefore, this LOAEL represents a conservative estimate of a possible adverse effect. There is also uncertainty with the selected TRVs due to the limited toxicity data available.

TRVs for vanadium were not available from the draft FFS (Malcolm Pirnie 2007b) or revised FFS (Louis Berger et al. 2014), so a second set of TRVs for vanadium was not selected.

Zinc

TRVs for zinc were derived based on a review of six toxicity studies conducted with chickens and mallard ducks. Data were insufficient for the development of an SSD. A decrease in chick body weight by as much as 21% (compared to the control) was observed after exposure to zinc at a dietary dose of 124 mg/kg bw/day for five weeks (Roberson and Schaible 1960). The next lowest dose of 82 mg/kg bw/day did not have a significant effect on growth. Therefore, the NOAEL and LOAEL TRVs for zinc were 82 and 124 mg/kg bw/day, respectively.

A NOAEL and LOAEL of 17.2 and 172 mg/kg bw/day, respectively, were selected for zinc (Battelle 2005). These values were USEPA Region 9 BTAG TRVs (USEPA 2002f) and were based on mallard gonadal weight using data from Gasaway and Buss (1972).

PAHs

For PAHs, TRVs were based on LPAH and HPAH sums (Louis Berger et al. 2014). TRVs were also based on total PAHs and individual PAH compounds as available from the toxicological literature (i.e., benzo(a)pyrene as a surrogate for HPAHs).

Total PAHs

The NOAEL TRV for total PAHs was derived from a study by Patton and Dieter (1980), the only available acceptable study that exposed birds to a PAH mixture in the diet. In this study, mallard ducks were exposed to a PAH mixture for seven months; no effect on body weight or survival was observed after seven months' exposure at the highest dose of 40,000 µg/kg bw/day. At this dose, a reduction in growth was observed over the first two months, but this temporary effect was recovered over the duration of the exposure, so it was not considered an adverse effect. No LOAEL dose could be derived from this study because no effects were observed at the highest dose tested.

[123] Body weight had decreased by 142 g at the end of the study, although the initial weight was not provided. The 10% decrease was estimated using an initial body weight of 1,475 g, the average for a mature white leghorn hen (NRC 1994).

There is some uncertainty associated with the assumption that a temporary reduction in body weight would not adversely affect wild populations. There is also uncertainty associated with this NOAEL due to the small number of toxicity studies available from the literature review (only one study). Furthermore, the NOAEL is an unbounded NOAEL based on exposure to an aromatic mixture that contained some percentage of PAHs, but did not include benzo(a)pyrene, so it is uncertain how the mixture of PAHs in that study compares to the composition of potential PAH exposure in the LPRSA.

Total LPAHs

No studies that met TRV acceptability criteria for literature review were found that exposed birds to LPAH mixtures in the diet. NOAEL and LOAEL TRVs of 670 and 6,700 µg/kg bw/day, respectively, were selected from previous documents developed by USEPA Region 2 for the LPRSA for LPAH (Louis Berger et al. 2014). These values were derived from Schafer et al. (1983), wherein birds were exposed to individual LPAH compounds (acenaphthene, fluorene, phenanthrene, and anthracene) for 48 hrs to determine the acute LD50 (dose that is lethal to 50% of an exposed population) values. While details of the text procedures were not clearly specified, Schafer et al. (1983) noted that exposure was based on food consumption data over an 18-hr period. There is uncertainty regarding whether exposure over a short duration reflects contaminant absorption in a natural setting. There is also uncertainty associated with the extrapolation of a toxicity value for one PAH compound to a mixture of LPAHs, because this approach assumes that the potency of all individual compounds in the mixture are equivalent. In addition, the effect concentrations in the study were adjusted using an interspecies extrapolation factor of three and an ACR factor of five to derive the LOAEL, based on the assumption that red-winged blackbirds (*Agelaius phoeniceus*) were three times more sensitive than selected avian species evaluated (Louis Berger et al. 2014).

Benzo(a)pyrene and Total HPAHs

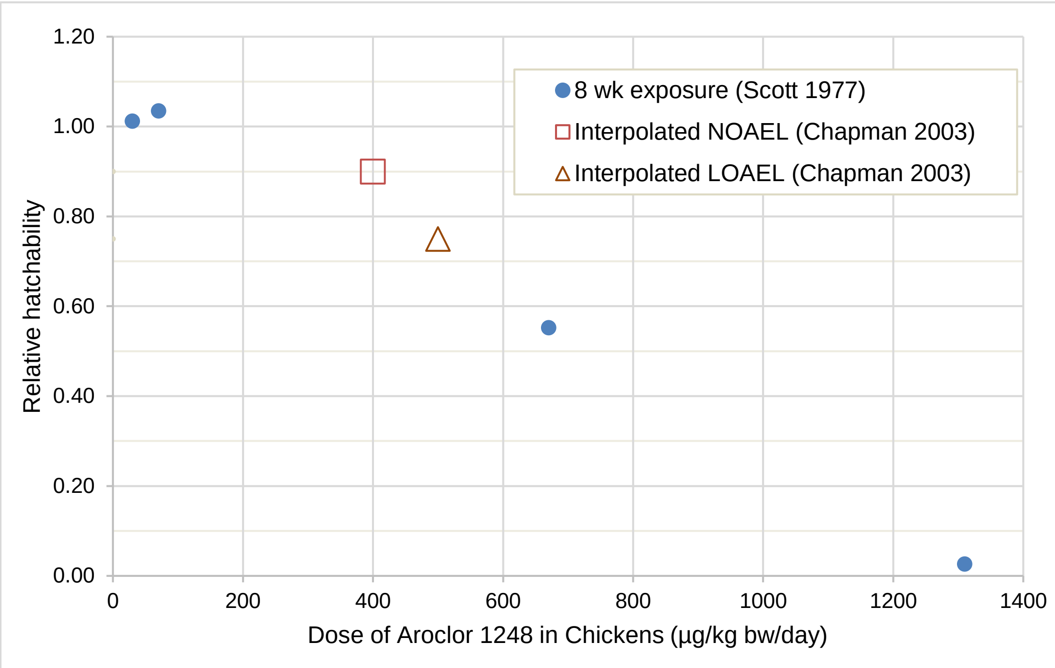
No acceptable studies were found that exposed birds to HPAH mixtures in the diet; instead, the lowest LOAEL for a single HPAH was selected and used as a surrogate for HPAHs. Two studies were reviewed that evaluated the effects of exposure to benzo(a)pyrene on birds. Only one of these studies used a dietary exposure pathway (Rigdon and Neal 1963), and no effects were observed in the study. Therefore, the study by Hough et al. (1993), which resulted in an observed effect from exposure to benzo(a)pyrene, was used to derive the LOAEL TRV. A LOAEL TRV of 1,400 µg/kg bw/day was selected based on observed cessation of egg laying in pigeons exposed for five months to this daily dose (Hough et al. 1993). This value was derived by dividing the weekly dose (10,000 µg/kg bw/day) by 7 days to determine the daily dose exposure. The NOAEL TRV was estimated from the LOAEL TRV using an uncertainty factor of 10. There is uncertainty associated with these TRVs because they are based on a weekly injection exposure; it is unknown how representative this route of exposure is of bioavailability and absorption via dietary exposure in wild populations. In addition, there were limited data available on avian toxicity, and the NOAEL was derived using an extrapolation factor of 10. There is uncertainty associated with the derivation of a NOAEL based on an extrapolation factor.

This same study (Hough et al. 1993) was used to derive a TRV for HPAHs. In keeping with the revised draft of the LPRSA FFS (Louis Berger et al. 2014), TRVs based on benzo(a)pyrene were recommended for comparison to the sum of all HPAHs. NOAEL and LOAEL TRVs of 48 and 480 µg/kg bw/day, respectively, were selected for HPAH (Louis Berger et al. 2014) by dividing an estimated daily dose by an interspecies extrapolation factor of 3. This interspecies extrapolation factor was based on the assumption that pigeons were three times less sensitive than selected avian species evaluated (Louis Berger et al. 2014). The NOAEL was derived from the LOAEL based on a factor of 10. There is uncertainty associated with the derivation of a NOAEL based on an extrapolation factor. There is uncertainty associated with these TRVs, as discussed above and due to the application of an interspecies extrapolation factor. In addition, other HPAHs are not known to be as toxic as benzo(a)pyrene, so the comparison of a dose of total HPAHs to a benchmark dose based on benzo(a)pyrene is highly conservative.

Total PCBs

TRVs for total PCBs were derived from a review of 11 acceptable studies (Appendix E) on the effects of dietary PCBs (i.e., individual Aroclors or Aroclor mixtures) on birds, excluding chickens. These acceptable studies were conducted on five species (American kestrel, mallard, ring-necked pheasant, ringed turtle dove, and screech owl) and all studies evaluated reproductive effects. Data were insufficient for development of an SSD because effects were only reported for four bird species. The lowest LOAEL from these 11 studies was 1,400 µg/kg bw/day, based on reduced hatching success in ringed turtle dove (Peakall et al. 1972; Peakall and Peakall 1973). No NOAEL was reported in the study with ringed turtle doves, so an uncertainty factor of 10 was applied to the LOAEL for a NOAEL TRV of 140 µg/kg bw/day. There is uncertainty associated with the use of a NOAEL derived using an uncertainty factor.

A NOAEL and LOAEL of 400 and 500 µg/kg bw/day, respectively, were also selected from previous documents developed by USEPA Region 2 for the LPRSA for total PCBs (Louis Berger et al. 2014). These TRVs were based on the interpolated no-effect value (a 10% decrease relative to control) and the interpolated low-effect value (a 25% decrease relative to control) using chicken hatchability data as described in Chapman (2003). The interpolated NOAEL and LOAEL were 400 and 500 µg/kg bw/day, respectively. The interpolated value from Chapman (2003) was based on the data reported by Scott (1977) and Lillie et al. (1975), wherein an empirical NOAEL and LOAEL of 70 and 670 µg/kg bw/day, respectively, were determined in the studies following 8 weeks of exposure to Aroclor 1248; at the LOAEL, chicken hatchability was significantly reduced (55% of control) (Figure 8-2). The Chapman (2003)-interpolated LOAEL (500 µg/kg bw/day; based on an effect threshold of 25% relative to control) was slightly less than the empirical LOAEL (670 µg/kg bw/day; based on hatchability reduced from control by 55%). TRVs based on domestic reproductive endpoints are uncertain, because domesticated species are bred to have high egg-laying rates compared to wild bird species. It is not known whether an effect threshold of 25% reduction in chicken egg hatchability is predictive of potential population-level effects in wild birds.



Source: Chapman (2003)

Figure 8-2. Interpolated bird dose total PCB data

PCDDs/PCDFs - Bird

Toxicity data were limited for the effects of PCDDs/PCDFs and dioxin-like PCBs on birds. Only two acceptable toxicity studies were available, which exposed ring-necked pheasants via intra-peritoneal injection (Nosek et al. 1992) and chickens via oral intubation (Schwetz et al. 1973) to 2,3,7,8-TCDD. Data were insufficient for the development of an SSD. The lowest LOAEL of 140 ng/kg bw/day (Nosek et al. 1992) was based on a 57% increase in mortality observed 16 weeks after the initiation of weekly injections for 10 weeks. No mortality was observed at the NOAEL of 14 ng/kg bw/day. The study by Schwetz et al. (1973) resulted in a much higher LOAEL of 1,000 ng/kg bw day based on an 80% increase in mortality in chickens compared to the control after a 21-day exposure. Both USEPA and CPG selected the lowest LOAEL of 140 ng/kg bw/day and the associated NOAEL of 14 ng/kg bw/day from these two studies as the basis for the TRVs to be used in this BERA; however, USEPA applied a species sensitivity factor of 5 to derive recommended TRVs. The resulting NOAEL and LOAEL were 2.8 and 28 ng/kg bw/day, respectively (Louis Berger et al. 2014), based on USEPA criteria (USEPA 1995a). The interspecies extrapolation factor of five was used to account for data indicating that pheasants were not among the most sensitive species.

This extrapolation factor was based on the observation that chickens, which were in the high-sensitivity group, had a LOAEL that was approximately one order of magnitude greater than the LOAEL for ring-necked pheasants for the same endpoint. There is uncertainty associated with the use of extrapolation factors to derive TRVs.

Based on the TEFs, the four most-potent dioxin-like PCBs in birds are PCBs 77, 81, 126, and 169, but the uncertainty regarding bird TEFs is high. For PCB 77, five studies produced TEFs ranging over three orders of magnitude (< 0.0003 to 0.15) for the various bird species tested (Van den Berg et al. 1998). For PCB 81, two studies tested several species and identified TEFs ranged from 0.001 to 0.5. For PCBs 126 and 169, data were available from only one study, so the associated uncertainty has not yet been quantified.

The high variability in TEFs may be due, in part, to differences in species sensitivity to dioxin-like compounds. Bird species can be grouped into three general classes of sensitivity to dioxin-like compounds, based on documented species differences in the amino acid sequences at the Ah receptor. These species differences affect not only overall sensitivity, but also the relative potency (i.e., TEFs) of individual dioxin-like compounds (Farmahin et al. 2013). Another issue with bird TEFs is their relevance for assessing dietary exposure risks. TEFs are estimated based on ethoxyresorufin-O-deethylase induction or *in ovo* studies. Such studies are most accurate for assessing effects based on tissue congener concentrations in whole embryos (USEPA 2008), not dietary exposure studies.

Ring-necked pheasants and chickens (the two species used in the dietary TEQ toxicity studies) are in the moderate- and high-sensitivity groups, respectively, as classified by Farmahin et al. (2013). It is interesting to note that in the toxicity studies described, chickens were less sensitive than ring-necked pheasants, even though chickens were predicted to be in the high-sensitivity group.

Total DDx

A review of 20 acceptable toxicity studies that evaluated the effects of DDD, DDE, DDT, or mixtures of DDT compounds in the diet of birds was conducted. LOAELs based on eggshell thickness, adult survival, offspring survival, or hatchability were reported for 10 bird species (i.e., American kestrel, bald eagle, barn owl, American black duck, double-crested cormorant, Japanese quail, mallard, ring dove, ring-necked pheasant, and white Pekin duck) and ranged from 150 to 71,100 µg/kg bw/day. An SSD was developed to derive a TRV (Figure 8-3). The 5th percentile determined from the SSD was 250 µg/kg bw/day; this TRV was selected. This SSD-derived LOAEL is within the range of measured LOAELs reported from the literature. The NOAEL TRV (25 µg/kg ww) was extrapolated from the LOAEL TRV using an uncertainty factor of 10. There is uncertainty associated with the use of an extrapolation factor to derive a NOAEL.

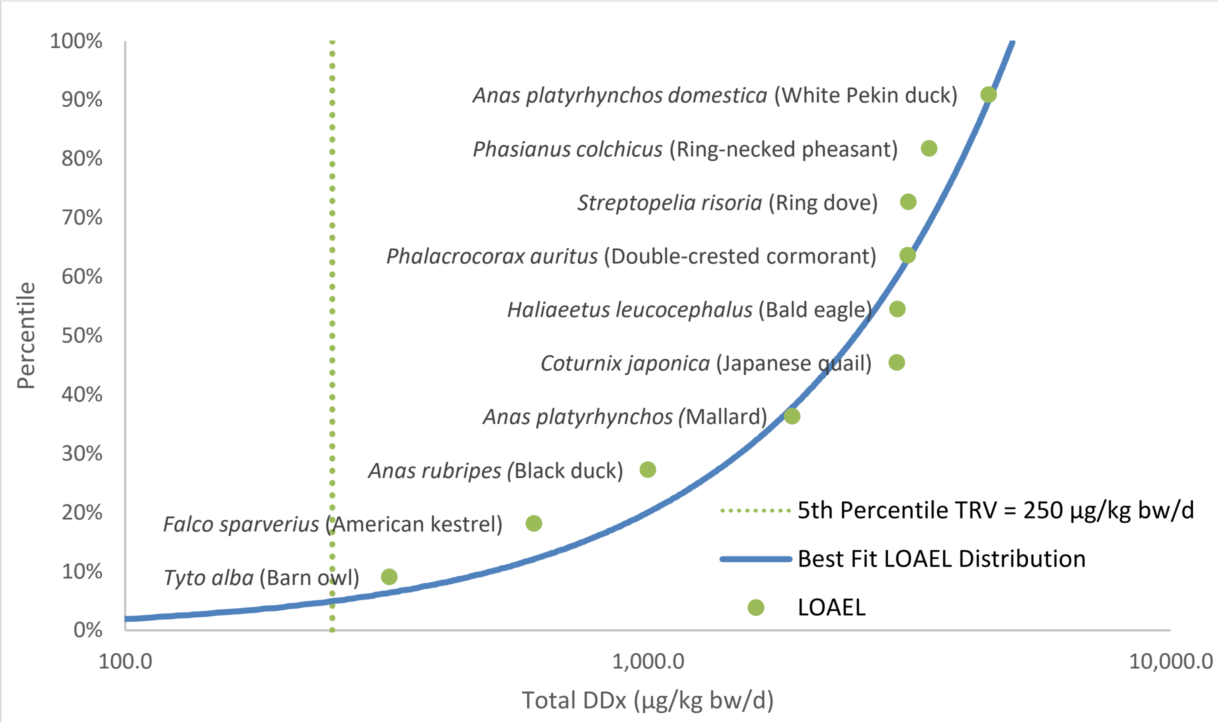


Figure 8-3.SSD derived from bird dietary toxicity data for total DDx

A NOAEL and LOAEL of 0.9 and 27 µg /kg bw/day, respectively, were selected for total DDx (Louis Berger et al. 2014) based on USEPA (1995a). These TRVs were cited based on pelican reproduction data from Anderson et al. (1975). The LOAEL TRV of 27 µg/kg bw/day was based on results from a field study that compared observations about productivity and eggshell thinning to standards known to support a stable population (Anderson et al. 1975). USEPA (2017d) noted that 27 µg /kg bw/day was the geomean concentration in anchovies that was associated with substantial improvements in productivity per pelican breeding pair and in eggshell thickness. However, while Anderson et al. (1975) suggested that the pelican populations off the coast of California were recovering in 1974, reference to previous studies of eastern brown pelicans indicated that the reproductive sustainability of the populations had not yet stabilized, as average egg shell thickness (weighted average of crushed and intact eggs) was still 21% less than the pre-1943 mean.

There is uncertainty associated with the TRVs derived from Anderson et al. (1975), as no analysis was performed to determine the significance of changes (in eggshell thinning and productivity), and a critical threshold level indicative of an adverse effect could not be determined—only an associated concentration at which productivity had improved but not recovered relative to historical levels. In addition, no consideration was given to the impacts that may have resulted from exposure to multiple chemicals, although DDE was the only contaminant detected in anchovies that is directly linked to eggshell thinning in birds (Anderson et al. 1975), and the mode of action of DDE on calcium regulation in birds, which contributes to eggshell thickness, is well documented

(Lundholm 1987). Extrapolation factors were used to derive the NOAEL TRV from the LOAEL, thereby creating uncertainty.

8.1.4 Risk characterization

This section presents the HQs for birds (Section 8.1.4.1), as well as uncertainties associated with the HQ calculations (Section 8.1.4.2). In addition to the dietary HQ calculations presented in Section 8.1.4.1, alternate HQs are calculated in Section 8.1.4.2 based on the identified uncertainties. These alternate HQs are calculated to determine if any of the uncertainties could result in risk conclusions that are different from those determined by the original HQs. Appendix G details the dietary doses, TRVs, and calculated HQs for the bird dietary COPECs (Tables G7 through G15).

8.1.4.1 Dietary HQs

Dietary HQs were calculated for the COPECs that were identified for further evaluation in this BERA based on the results from the SLERA for birds using the EPCs described in Table 8-6 (based on UCLs or maximum concentrations if < 6 detected values) to calculate dietary doses. The dietary doses were compared to the NOAEL and LOAEL TRVs (Table 8-11) to calculate HQs. The following sections discuss results by bird species.

Spotted Sandpiper

Dietary HQs were calculated for spotted sandpiper on both a site-wide and a reach-specific basis using a range of TRVs (Table 8-12). Only one dietary scenario (100% worms in the diet) was used. The following COPECs had LOAEL HQs ≥ 1.0: copper, lead, total HPAHs, total PCBs, PCB TEQ - bird, PCDD/PCDF TEQ - bird, total TEQ - bird, and total DDx. The following sections discuss each of the COPECs with LOAEL HQs ≥ 1.0. Complete results, including EPCs used in Equation 8-1 and reach-specific results, are presented in Appendix G.

Table 8-12. Spotted sandpiper dietary HQs

COPEC by Area	Dose	Units	Range of TRVs[a]				Range of HQs[a]			
			TRV-A[b]		TRV-B[c]		HQ Based on TRV-A[b]		HQ Based on TRV-B[c]	
			NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL
Cadmium										
Site wide	0.23	mg/kg bw/day	0.4	4.0	0.080	10.4	0.57	0.057	**2.9**	0.022
By reach	0.10–0.44						0.25–**1.1**	0.029–0.11	**1.3–5.5**	0.010–0.042
Chromium										
Site wide	19.2	mg/kg bw/day	10.5	105	na	na	**1.8**	0.18	na	na
By reach	5.8–41						0.56–**3.9**	0.056–0.39	na	na
Copper										
Site wide	9.5	mg/kg bw/day	1.9	19	2.3	4.7	**5.0**	0.50	**4.1**	**2.0**
By reach	5.6–16.7						**3.0–8.8**	0.30–0.88	**2.4–7.3**	**1.2–3.6**
Lead										
Site wide	14	mg/kg bw/day	5.5	28	0.19	1.9	**2.5**	0.49	**73**	**7.3**
By reach	5.7–19						**1.0–3.4**	0.20–0.59	**30–100**	**3.0–10**
Methyl mercury										
Site wide	2.0	µg/kg bw/day	9.6	96	13	26	0.21	0.021	0.15	0.077
By reach	0.69–2.5						0.072–0.27	0.0072–0.027	0.05–0.20	0.027–0.10
Nickel										
Site wide	9.3	mg/kg bw/day	15	33	1.38	56.3	0.62	0.28	**6.7**	0.17
By reach	2.7–24						0.18–**1.6**	0.082–0.71	**2.0–17**	0.048–0.42

Table 8-12. Spotted sandpiper dietary HQs

COPEC by Area	Dose	Units	Range of TRVs[a]				Range of HQs[a]			
			TRV-A[b]		TRV-B[c]		HQ Based on TRV-A[b]		HQ Based on TRV-B[c]	
			NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL
Selenium										
Site wide	0.38	mg/kg bw/day	0.42	0.82	0.23	0.93	0.90	0.46	**1.7**	0.41
By reach	0.21–0.4						0.54–**1.2**	0.26–0.63	0.93–**2.2**	0.24–0.56
Vanadium										
Site wide	1.2	mg/kg bw/day	1.2	2.3	na	na	**1.0**	0.54	na	na
By reach	1.0–2.1						0.81–**1.8**	0.42–0.92	na	na
Zinc										
Site wide	40	mg/kg bw/day	82	124	17.2	172	0.48	0.32	**2.3**	0.23
By reach	25–59						0.30–0.72	0.20–0.48	**1.5–2.8**	0.15–0.34
Total HPAHs										
Site wide	2341	µg/kg bw/day	na	na	48	480	na	na	na	**4.9**
By reach	917–4795						na	na	na	**1.9–10**
Total LPAH										
Site wide	591	µg/kg bw/day	na	na	670	6,700	na	na	0.88	0.088
By reach	124–964						na	na	0.18–**1.4**	0.018–0.14
Benzo(a)pyrene										
Site wide	402	µg/kg bw/day	140	1400	na	na	**2.9**	0.29	na	na
By reach	112–478						0.80–**3.4**	0.080–0.34	na	na

Table 8-12. Spotted sandpiper dietary HQs

COPEC by Area	Dose	Units	Range of TRVs[a]				Range of HQs[a]			
			TRV-A[b]		TRV-B[c]		HQ Based on TRV-A[b]		HQ Based on TRV-B[c]	
			NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL
Total PAHS										
Site wide	2840	µg/kg bw/day	40,000	na	na	na	0.071	na	na	na
By reach	1080–5981						0.028–0.17	na	na	na
Total PCBs										
Site wide	241	µg/kg bw/day	140	1,400	400	500	**1.7**	0.17	0.60	0.48
By reach	65–575						0.47–**4.1**	0.047–0.41	0.16–**1.4**	0.13–**1.2**
PCB TEQ - bird										
Site wide	43	ng/kg bw/day	14	140	2.8	28	**3.1**	0.31	**15**	**1.5**
By reach	10–109						0.073–**7.8**	0.073–0.78	**3.7–39**	0.37–**3.9**
PCDD/PCDF TEQ - bird										
Site wide	127	ng/kg bw/day	14	140	2.8	28	**9.1**	0.91	**45**	**4.5**
By reach	2.0–584						0.14–**42**	0.014–**4.2**	0.71–**208**	0.071–**21**
Total TEQ - bird										
Site wide	152	ng/kg bw/day	14	140	2.8	28	**11**	**1.1**	**54**	**5.4**
By reach	12–703						0.89–**50**	0.089–**5.0**	**4.4–251**	0.44–**25**
Total DDx										
Site wide	17	µg/kg bw/day	25	250	0.9	27	0.69	0.069	**19**	0.64
By reach	4.3–38						0.17–**1.5**	0.018–0.15	**4.7–43**	0.16–**1.4**

Bold identifies HQs ≥ 1.0.
Shaded cells identify HQs ≥ 1.0 based on a LOAEL TRV.

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

[b] TRVs were derived from the primary literature review based on process identified in Section 8.1.3.1.

[c] TRVs were based on USEPA's revised draft of the LPR restoration project FFS (Louis Berger et al. 2014), first draft of the LPR restoration project FFS (Malcolm Pirnie 2007b), or LPR restoration project PAR (Battelle 2005).

BERA – baseline ecological risk assessment
bw – body weight
COPEC – chemical of potential ecological concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
FFS – focused feasibility study
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
LPAH – low-molecular-weight polycyclic aromatic hydrocarbon
LPR – Lower Passaic River
LPRSA – Lower Passaic River study Area
LOAEL – lowest-observed-adverse-effect level
na – not applicable (no TRV available)
NJDEP – New Jersey Department of Environmental Protection
NOAEL – no-observed-adverse-effect level
PAH – polycyclic aromatic hydrocarbon
PAR – pathways analysis report
PCB – polychlorinated biphenyl
PCDD – polychlorinated dibenzo-*p*-dioxin
PCDF – polychlorinated dibenzofuran
TEQ – toxic equivalent
TRV – toxicity reference value
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT and 4,4′-DDT)
USEPA – US Environmental Protection Agency

Copper

LOAEL HQs for copper ranged from 0.5 to 2.0 on a site-wide basis, and ranged from 0.30 to 3.6 by reach. The highest LOAEL HQ was 3.6 in Reach 4 (RM 6 to RM 8).

Lead

LOAEL HQs for lead ranged from 0.49 to 7.3 on a site-wide basis, and ranged from 0.20 to 10 by reach. The highest HQ of 10 was in Reach 4 (RM 6 to RM 8).

Total HPAHs

For total HPAHs, LOAEL HQs were 4.9 on a site-wide basis and ranged from 1.9 to 10 by reach. The highest HQ was in Reach 4 (RM 6 to RM 8). HQs using the TRV for benzo(a)pyrene were < 1.0.

Total PCBs

LOAEL HQs for total PCBs ranged from 0.17 to 0.48 on a site-wide basis, and ranged from 0.047 to 1.2 by reach. The highest HQ of 1.2 was in Reach 4 (RM 6 to RM 8). None of the other LOAEL HQs, on either a site-wide or reach-specific basis, were ≥ 1.0.

PCB TEQ - bird, PCDD/PCDF TEQ - bird, and total TEQ - bird

LOAEL HQs for PCB TEQ - bird ranged from 0.31 to 1.5 on a site-wide basis, and ranged from by 0.073 to 3.9 by reach. LOAEL HQs for PCDD/PCDF TEQ ranged from 0.91 to 4.5 on a site-wide basis, and ranged from 0.014 to 21 by reach. LOAEL HQs for total TEQ ranged from 1.1 to 5.4 on a site-basis, and ranged from 0.089 to 25 by reach. On a reach-specific basis, the highest HQs were in Reach 4 (RM 6 to RM 8), which had HQs of 3.9, 21, and 25 for PCB TEQ, PCDD/PCDF TEQ, and total TEQ, respectively.

Total DDx

LOAEL HQs for total DDx ranged from 0.15 to 0.64 on a site-wide basis, and ranged from 0.085 to 1.4 by reach. The only LOAEL HQs ≥ 1.0 were those calculated on a reach-specific basis for Reaches 4 (RM 6 to RM 8) and 5 (RM 8 to RM 10), which had HQs of 1.0 and 1.4, respectively.

Great Blue Heron

Dietary HQs were calculated for great blue heron on both a site-wide and reach-specific basis using a range of TRVs (Table 8-13). Two dietary scenarios were used: Scenario 1, which used consumption of fish 0 to 13 cm long only; and Scenario 2, in which fish consumption was divided into four size classes (0 to 13 cm, 13 to 18 cm, 18 to 30 cm, and > 30 cm, with dietary percentages of 17, 29, 40, and 14, respectively) (Table 8-4). The following COPECs had LOAEL HQs ≥ 1.0: copper, methylmercury, total PCBs, PCB TEQ - bird, PCDD/PCDF TEQ - bird, total TEQ - bird, and total DDx. The following sections discuss each of the COPECs with LOAEL HQs ≥ 1.0. Complete results, including EPCs used in Equation 8-1 and reach-specific results, are presented in Appendix G.

Table 8-13. Great blue heron dietary HQs

Diet Scenario	Area	Dose	Units	Range of TRVs[a]				Range of HQs[a]			
				TRV-A[b]		TRV-B[c]		HQs Based on TRV-A[b]		HQs Based on TRV-B[c]	
				NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL
Cadmium											
1	site wide	0.0086	mg/kg bw/day	0.4	4.0	0.080	10.4	0.022	0.0022	0.11	0.0083
	by reach	0.0045–0.021						0.011–0.053	0.0011–0.053	0.056–0.26	0.00043–0.0020
2	site wide	0.01						0.021	0.0021	0.10	0.00079
	by reach	0.00–0.02						0.006–0.041	0.00058-0.0041	0.029–0.21	0.00022–0.0016
Chromium											
1	site wide	2.0	mg/kg bw/day	10.5	105	na	na	0.19	0.019	na	na
	by reach	0.77–5.5						0.073–0.39	0.0073–0.053	na	na
2	site wide	3.0						0.28	0.028	na	na
	by reach	0.30–6.5						0.028–0.62	0.0033–0.062	na	na
Copper											
1	site wide	0.62	mg/kg bw/day	1.9	19	2.3	4.7	0.33	0.033	0.27	0.13
	by reach	0.60–1.1						0.29–0.55	0.029–0.055	0.24–0.45	0.12–0.22
2	site wide	1.8						0.94	0.094	0.78	0.38
	by reach	0.65–6.3						0.034–**3.3**	0.03–0.33	0.28–**2.7**	0.14–**1.3**
Lead											
1	site wide	0.51	mg/kg bw/day	5.5	28	0.19	1.9	0.093	0.018	2.7	0.27
	by reach	0.19–1.0						0.036–0.18	0.069–0.036	**1.0–5.2**	0.10–0.52
2	site wide	0.29						0.053	0.010	**1.5**	0.15
	by reach	0.11–0.43						0.021–0.078	0.0041–0.015	0.60–**2.3**	0.06–0.23

Table 8-13. Great blue heron dietary HQs

Diet Scenario	Area	Dose	Units	Range of TRVs[a]				Range of HQs[a]			
				TRV-A[b]		TRV-B[c]		HQs Based on TRV-A[b]		HQs Based on TRV-B[c]	
				NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL
Methylmercury											
1	site wide	9.6	µg/kg bw/day	9.6	96	13	26	1.0	0.10	0.74	0.37
	by reach	3.0–12						0.31–**1.3**	0.031–0.13	0.23–0.92	0.11–0.46
2	site wide	24						**2.5**	0.25	**1.9**	0.94
	by reach	23–41						**2.4–4.2**	0.24-0.42	0.8–**3.1**	0.9–**1.6**
Nickel											
1	site wide	1.2	mg/kg bw/day	15	33	1.38	56.3	0.083	0.038	0.90	0.022
	by reach	0.55–3.9						0.035–0.26	0.016–0.12	0.38–**2.8**	0.094–0.068
2	site wide	2.1						0.14	0.062	**1.5**	0.036
	by reach	0.20–4.7						0.014–0.31	0.0062–0.14	0.15–**3.4**	0.0036–0.083
Selenium											
1	site wide	0.12	mg/kg bw/day	0.42	0.82	0.23	0.93	0.28	0.14	0.51	0.13
	by reach	0.052–0.16						0.12–0.37	0.064–0.19	0.23–0.68	0.056–0.17
2	site wide	0.17						0.41	0.21	0.76	0.19
	by reach	0.12–0.26						0.28–0.62	0.14–0.32	0.50–**1.2**	0.12–0.29
Vanadium											
1	site wide	0.14	mg/kg bw/day	1.2	2.3	na	na	0.11	0.059	na	na
	by reach	0.074–0.24						0.061–0.20	0.032–0.11	na	na
2	site wide	0.071						0.059	0.031	na	na
	by reach	0.040–0.11						0.033–0.089	0.017–0.046	na	na

Table 8-13. Great blue heron dietary HQs

Diet Scenario	Area	Dose	Units	Range of TRVs[a]				Range of HQs[a]			
				TRV-A[b]		TRV-B[c]		HQs Based on TRV-A[b]		HQs Based on TRV-B[c]	
				NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL
Zinc											
1	site wide	7.6	mg/kg bw/day	82	124	17.2	172	0.092	0.061	0.44	0.044
	by reach	4.5–8.5						0.054–0.10	0.036–0.069	0.26–0.50	0.026–0.050
2	site wide	5.6						0.068	0.045	0.32	0.032
	by reach	5.3–6.8						0.065–0.083	0.043–0.055	0.31–0.040	0.031–0.040
Total HPAHs											
1	site wide	89	µg/kg bw/day	na	na	48	480	na	na	**1.9**	0.19
	by reach	21–122						na	na	0.44–**4.0**	0.044–0.34
2	site wide	50						na	na	**1.0**	0.10
	by reach	21–81						na	na	0.43–**1.7**	0.04–0.17
Total LPAHs											
1	site wide	22	µg/kg bw/day	na	na	670	6,700	na	na	0.033	0.0033
	by reach	8.0–67						na	na	0.011–0.10	0.0011–0.010
2	site wide	31						na	na	0.046	0.0046
	by reach	25–43						na	na	0.033–0.065	0.0033–0.059

Table 8-13. Great blue heron dietary HQs

Diet Scenario	Area	Dose	Units	Range of TRVs[a]				Range of HQs[a]			
				TRV-A[b]		TRV-B[c]		HQs Based on TRV-A[b]		HQs Based on TRV-B[c]	
				NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL
Benzo(a)pyrene											
1	site wide	7.0	µg/kg bw/day	140	1,400	na	na	0.050	0.0050	na	na
	by reach	2.5–19.3						0.018–0.14	0.0018–0.014	na	na
2	site wide	4.9						0.035	0.0035	na	na
	by reach	1.6–7.7						0.012–0.055	0.0012–0.0055	na	na
Total PAHs											
1	site wide	97	µg/kg bw/day	40,000	na	na	na	0.0024	na	na	na
	by reach	29–207						0.00071–0.059	na	na	na
2	site wide	76						0.0019	na	na	na
	by reach	43–121						0.0011-0.0029	na	na	na
Total PCBs											
1	site wide	98	µg/kg bw/day	140	1,400	400	500	0.70	0.70	0.25	0.20
	by reach	44–165						0.31–**1.2**	0.03–0.12	0.11–0.41	0.09–0.33
2	site wide	330						**2.4**	0.24	0.83	0.66
	by reach	155–569						**1.5–4.1**	0.11–0.41	0.39–**1.4**	0.31–**1.1**
PCB TEQ - bird											
1	site wide	9.3	ng/kg bw/day	14	140	2.8	28	0.67	0.067	**3.3**	0.33
	by reach	3.5–15						0.25–**1.1**	0.030–0.11	**1.3–5.5**	0.13–0.48
2	site wide	25						**1.8**	0.18	**9.0**	0.90
	by reach	9.0–45						0.64–**3.3**	0.064–0.33	**3.2–16**	0.32–**1.6**
PCDD/PCDF TEQ - bird											
1	site wide	9.37	ng/kg bw/day	14	140	2.8	28	0.67	0.067	**3.3**	0.33
	by reach	2.7–20						0.24–1.45	0.024–0.14	0.98–**7.2**	0.10–0.72

Table 8-13. Great blue heron dietary HQs

Diet Scenario	Area	Dose	Units	Range of TRVs[a]				Range of HQs[a]			
				TRV-A[b]		TRV-B[c]		HQs Based on TRV-A[b]		HQs Based on TRV-B[c]	
				NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL
2	site wide	27						**1.9**	0.19	**9.6**	**1.0**
	by reach	7.2–52						0.52–**3.7**	0.052–0.37	2.9–**19**	0.26–**1.9**
Total TEQ - bird											
1	site wide	9.4	ng/kg bw/day	14	140	2.8	28	0.67	0.067	**3.3**	0.33
	by reach	6.3–34						0.45–**2.4**	0.045–0.24	2.2–**12**	0.22–**1.2**
2	site wide	51						**3.7**	0.37	**18**	**1.8**
	by reach	22–99						**1.1–7.1**	0.11–0.71	5.5–**35**	0.55–**3.5**
Total DDx											
1	site wide	31	µg/kg bw/day	25	250	0.9	27	0.43	0.043	**12**	0.40
	by reach	5.1–15						0.20–0.70	0.020–0.070	**5.6–19**	0.19–0.65
2	site wide	35						**1.4**	0.14	**39**	**1.3**
	by reach	18–65						0.73–2.6	0.073–0.26	**20–61**	0.67–**2.4**

Bold identifies HQs ≥ 1.0.

Shaded cells identify HQs ≥ 1.0 based on a LOAEL TRV.

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

[b] TRVs were derived from the primary literature based on process identified in Section 8.1.3.1.

[c] TRVs were based on USEPA's revised draft of the LPR restoration project FFS (Louis Berger et al. 2014), first draft of the LPR restoration project FFS (Malcolm Pirnie 2007b), orLPR restoration project PAR (Battelle 2005).

BERA – baseline ecological risk assessment
bw – body weight
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
FFS – focused feasibility study
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
HQ – hazard quotient
LPAH – low-molecular-weight polycyclic aromatic hydrocarbon
LOAEL – lowest-observed-adverse-effect level
LPR – Lower Passaic River
LPRSA – Lower Passaic River study Area
na – not applicable (no TRV available)
NJDEP – New Jersey Department of Environmental Protection
NOAEL – no-observed-adverse-effect level
PAH – polycyclic aromatic hydrocarbon
PAR – pathways analysis report
PCB – polychlorinated biphenyl
PCDD – polychlorinated dibenzo-p-dioxin
PCDF – polychlorinated dibenzofuran
TEQ – toxic equivalent
TRV – toxicity reference value
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT and 4,4′-DDT)
USEPA – US Environmental Protection Agency

Copper

The LOAEL HQs for copper ranged from 0.033 to 0.38 on a site-wide basis, and ranged from 0.029 to 1.3 by reach. Only Reach 2 (RM 2 to RM 4) had a LOAEL HQ that was ≥ 1.0 (1.3), and only for copper under diet Scenario 2 (higher percentage of prey items greater than 13 cm in length).

Methylmercury

The LOAEL HQs for methylmercury ranged from 0.10 to 0.94 on a site-wide basis, and ranged from 0.031 to 1.6 by reach. The HQ exceedances under diet Scenario 2 ranged from 1.0 to 1.6 in all reaches except Reach 4 (RM 6 to RM 8), where the HQ was < 1.0.

Total PCBs

LOAEL HQs for total PCBs ranged from 0.70 to 0.33 on a site-wide basis, and ranged from 0.03 to 1.1 by reach. There were two LOAEL HQs ≥ 1.0 for total PCB congeners, with HQs of 1.0 in Reach 4 (RM 6 to RM8) and 1.1 in Reach 5 (RM 8 to 10) using diet Scenario 2.

PCB TEQ - bird

LOAEL HQs for PCB TEQ - bird ranged from 0.067 to 0.90 on a site-wide basis, and ranged from 0.030 to 1.6 by reach. There were three LOAEL HQs ≥ 1.0 for PCB TEQ: in Reaches 3 (RM 4 to RM 6), 4 (RM 6 to RM 8), and 5 (RM 8 to RM 10), with HQs ranging from 1.3 to 1.6 using diet Scenario 2.

PCDD/PCDF TEQ - bird

LOAEL HQs for PCDD/PCDF TEQ - bird ranged from 0.067 to 1.0 on a site-wide basis, and ranged from 0.024 to 1.9 by reach. On a reach-specific basis, there were three LOAEL HQs ≥ 1.0 for PCDD/PCDF TEQ: in Reaches 3 (RM 4 to RM 6), 4 (RM 6 to RM 8), and 5 (RM 8 to RM 10), with HQs ranging from 1.1 to 1.9 using diet Scenario 2.

Total TEQ - bird

LOAEL HQs ranged from 0.067 to 1.8 on a site-wide basis, and ranged from 0.045 to 3.5 by reach. On a reach-specific basis for diet Scenario 1, there were two HQ exceedances: an HQ of 1.2 in Reach 4 (RM 6 to RM 8) and an HQ of 1.1 in Reach 5 (RM 8 to 10). For diet Scenario 2, HQ exceedances existed in Reaches 1 (RM 0 to RM 2), 3 through 5 (RM 4 through RM 10), and 7 (RM 12 to RM 14), with HQs ranging from 1.4 to 3.5.

Total DDx

LOAEL HQs ranged from 0.043 to 1.3 on a site-wide basis, and ranged from 0.020 to 2.3 by reach. LOAEL HQs ≥ 1.0 for total DDx were based on using diet Scenario 2. The site-wide LOAEL HQ was 1.3, and the reach-specific LOAEL HQs ranged from 1.1 to 2.4 in six reaches.

Belted Kingfisher

Dietary HQs were calculated for belted kingfisher using three exposure area assumptions (at and above RM 6, site wide, and reach specific), and using a range of TRVs (Table 8-14). Two dietary scenarios were used: Scenario 1, which included consumption of fish only 0 to 9 cm long; and Scenario 2, which divided fish consumption into three size classes of 0 to 9 cm, 9 to 13 cm, and 13 to 18 cm with dietary percentages of 31.5, 51, and 2.5, respectively (Table 8-4). The following COPECs for belted kingfisher had LOAEL HQs ≥ 1.0: lead, methylmercury, PCB TEQ - bird, PCDD/PCDF TEQ - bird, total TEQ - bird, and total DDx. The following sections discuss each of the COPECs with LOAEL HQs ≥ 1.0. Complete results, including EPCs used in Equation 8-1 and reach-specific results, are presented in Appendix G.

Table 8-14. Belted kingfisher dietary HQs

Diet Scenario	Area	Dose	Units	Range of TRVs[a]				Range of HQs[a]			
				TRV-A[b]		TRV-B[c]		HQ Based on TRV-A[b]		HQ Based on TRV-B[c]	
				NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL
Cadmium											
1	site wide	0.031	mg/kg bw/day	0.4	4.0	0.080	10.4	0.077	0.0077	0.39	0.0030
	RM ≥ 6	0.032						0.081	0.0081	0.41	0.0031
	by reach	0.014–0.044						0.034–0.11	0.0034–0.011	0.17–0.55	0.0013–0.0042
2	site wide	0.029						0.071	0.0071	0.36	0.0027
	RM ≥ 6	0.037						0.091	0.0091	0.46	0.0035
	by reach	0.018–0.050						0.047–0.13	0.0044–0.013	0.022–0.63	0.0017–0.0048
Chromium											
1	site wide	5.3	mg/kg bw/day	10.5	105	na	na	0.51	0.051	na	na
	RM ≥ 6	6.2						0.59	0.059	na	na
	by reach	0.55–14						0.052–**1.3**	0.0052–0.13	na	na
2	site wide	5.8						0.55	0.055	na	na
	RM ≥ 6	8.4						0.80	0.080	na	na
	by reach	2.1–13						0.20–**1.3**	0.020–0.13	na	na

Table 8-14. Belted kingfisher dietary HQs

Diet Scenario	Area	Dose	Units	Range of TRVs[a]				Range of HQs[a]			
				TRV-A[b]		TRV-B[c]		HQ Based on TRV-A[b]		HQ Based on TRV-B[c]	
				NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL
Copper											
1	site wide	3.4	mg/kg bw/day	1.9	19	2.3	4.7	**1.8**	0.18	**1.5**	0.72
	RM ≥ 6	3.4						**1.8**	0.18	**1.5**	0.72
	by reach	2.9–4.3						**1.5–2.2**	0.15–0.22	**1.3–1.9**	0.61–0.91
2	site wide	3.4						**1.8**	0.18	**1.5**	0.73
	RM ≥ 6	3.7						**2.0**	0.20	**1.6**	0.80
	by reach	2.9–4.6						**1.5–2.4**	0.15–0.24	**1.3–2.0**	0.61–0.97
Lead											
1	site wide	1.3	mg/kg bw/day	5.5	28	0.19	1.9	0.22	0.044	**6.5**	0.65
	RM ≥ 6	1.6						0.25	0.049	**7.2**	0.72
	by reach	0.47–2.1						0.074–0.34	0.015–0.066	**2.2–9.8**	0.22–0.98
2	site wide	1.2						0.24	0.048	**7.1**	0.71
	RM ≥ 6	1.4						0.29	0.056	**8.3**	0.83
	by reach	0.47–1.9						0.086–0.38	0.017–0.075	**2.5–11**	0.25–**1.1**

Table 8-14. Belted kingfisher dietary HQs

Diet Scenario	Area	Dose	Units	Range of TRVs[a]				Range of HQs[a]			
				TRV-A[b]		TRV-B[c]		HQ Based on TRV-A[b]		HQ Based on TRV-B[c]	
				NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL
Methylmercury											
1	site wide	24	µg/kg bw/day	9.6	96	13	26	**2.5**	0.25	**1.8**	0.92
	RM ≥ 6	21						**2.2**	0.22	**1.6**	0.81
	by reach	12.0–41						**1.3–4.3**	0.13–0.43	0.95–**3.1**	0.48–**1.6**
2	site wide	33						**3.5**	0.35	**2.6**	**1.3**
	RM ≥ 6	22						**2.3**	0.23	**1.7**	0.81
	by reach	14–41						**1.5–4.3**	0.15–0.43	**1.1–3.1**	0.55–**1.6**
Nickel											
1	site wide	3.4	mg/kg bw/day	15	33	1.38	56.3	0.23	0.10	**2.5**	0.061
	RM ≥ 6	3.9						0.26	0.12	**2.9**	0.070
	by reach	0.46–9.5						0.031–0.46	0.014–0.21	0.33–**6.9**	0.0082–0.17
2	site wide	3.9						0.26	0.12	**2.8**	0.069
	RM ≥ 6	5.7						0.38	0.17	**4.1**	0.10
	by reach	1.4–9.3						0.092–0.62	0.042–0.28	**1.0–6.7**	0.024–0.17

Table 8-14. Belted kingfisher dietary HQs

Diet Scenario	Area	Dose	Units	Range of TRVs[a]				Range of HQs[a]			
				TRV-A[b]		TRV-B[c]		HQ Based on TRV-A[b]		HQ Based on TRV-B[c]	
				NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL
Selenium											
1	site wide	0.36	mg/kg bw/day	0.42	0.82	0.23	0.93	0.85	0.43	**1.5**	0.38
1	RM ≥ 6	0.36	mg/kg bw/day	0.42	0.82	0.23	0.93	0.85	0.43	**1.5**	0.38
1	by reach	0.17–0.39	mg/kg bw/day	0.42	0.82	0.23	0.93	0.88–0.40	0.20–0.45	0.73–**1.7**	0.18–0.42
2	site wide	0.36	mg/kg bw/day	0.42	0.82	0.23	0.93	0.85	0.44	**1.6**	0.39
2	RM ≥ 6	0.34	mg/kg bw/day	0.42	0.82	0.23	0.93	0.82	0.42	**1.5**	0.37
2	by reach	0.18–0.42	mg/kg bw/day	0.42	0.82	0.23	0.93	0.42–0.99	0.22–0.51	0.77–**1.8**	0.19–0.45
Vanadium											
1	site wide	0.41	mg/kg bw/day	1.2	2.3	na	na	0.34	0.18	na	na
1	RM ≥ 6	0.45	mg/kg bw/day	1.2	2.3	na	na	0.37	0.19	na	na
1	by reach	0.11–0.58	mg/kg bw/day	1.2	2.3	na	na	0.09–0.49	0.05–0.25	na	na
2	site wide	0.35	mg/kg bw/day	1.2	2.3	na	na	0.29	0.15	na	na
2	RM ≥ 6	0.47	mg/kg bw/day	1.2	2.3	na	na	0.39	0.20	na	na
2	by reach	0.19–0.58	mg/kg bw/day	1.2	2.3	na	na	0.17–0.48	0.08–0.25	na	na

Table 8-14. Belted kingfisher dietary HQs

Diet Scenario	Area	Dose	Units	Range of TRVs[a]				Range of HQs[a]			
				TRV-A[b]		TRV-B[c]		HQ Based on TRV-A[b]		HQ Based on TRV-B[c]	
				NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL
Zinc											
1	site wide	22	mg/kg bw/day	82	124	17.2	172	0.27	0.18	**1.3**	0.13
1	RM ≥ 6	21	mg/kg bw/day	82	124	17.2	172	0.26	0.17	**1.2**	0.12
1	by reach	14–25	mg/kg bw/day	82	124	17.2	172	0.17–0.31	0.11–0.20	0.79–**1.5**	0.079–0.15
2	site wide	21	mg/kg bw/day	82	124	17.2	172	0.26	0.17	**1.2**	0.12
2	RM ≥ 6	20	mg/kg bw/day	82	124	17.2	172	0.25	0.16	**1.2**	0.12
2	by reach	14–24	mg/kg bw/day	82	124	17.2	172	0.17–0.29	0.11–0.19	0.80–**1.4**	0.080–0.14
Total HPAHs											
1	site wide	173	µg/kg bw/day	na	na	48	480	na	na	**3.6**	0.36
1	RM ≥ 6	194	µg/kg bw/day	na	na	48	480	na	na	**4.0**	0.40
1	by reach	55–384	µg/kg bw/day	na	na	48	480	na	na	**1.1–8.0**	0.11–0.80
2	site wide	241	µg/kg bw/day	na	na	48	480	na	na	**5.0**	0.50
2	RM ≥ 6	291	µg/kg bw/day	na	na	48	480	na	na	**6.1**	0.61
2	by reach	66–378	µg/kg bw/day	na	na	48	480	na	na	**1.4–7.8**	0.14–0.79

Table 8-14. Belted kingfisher dietary HQs

Diet Scenario	Area	Dose	Units	Range of TRVs[a]				Range of HQs[a]			
				TRV-A[b]		TRV-B[c]		HQ Based on TRV-A[b]		HQ Based on TRV-B[c]	
				NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL
Total LPAHs											
1	site wide	51	µg/kg bw/day	na	na	670	6,700	na	na	0.077	0.008
	RM ≥ 6	50						na	na	0.074	0.007
	by reach	20–165						na	na	0.030–0.25	0.0030–0.025
2	site wide	69						na	na	0.10	0.010
	RM ≥ 6	89						na	na	0.13	0.013
	by reach	25–161						na	na	0.037–0.24	0.0037–0.024
Benzo(a)pyrene											
1	site wide	17	µg/kg bw/day	140	1,400	na	na	0.12	0.012	na	na
	RM ≥ 6	19						0.14	0.014	na	na
	by reach	5.6–29						0.040–0.21	0.0040–0.021	na	na
2	site wide	22						0.16	0.016	na	na
	RM ≥ 6	26						0.19	0.019	na	na
	by reach	7.4–38						0.053–0.27	0.0053–0.027	na	na

Table 8-14. Belted kingfisher dietary HQs

Diet Scenario	Area	Dose	Units	Range of TRVs[a] TRV-A[b] NOAEL	Range of TRVs[a] TRV-A[b] LOAEL	Range of TRVs[a] TRV-B[c] NOAEL	Range of TRVs[a] TRV-B[c] LOAEL	Range of HQs[a] HQ Based on TRV-A[b] NOAEL	Range of HQs[a] HQ Based on TRV-A[b] LOAEL	Range of HQs[a] HQ Based on TRV-B[c] NOAEL	Range of HQs[a] HQ Based on TRV-B[c] LOAEL
Total PAHs											
1	site wide	221	µg/kg bw/day	40,000	na	na	na	0.0055	na	na	na
1	RM ≥ 6	244	µg/kg bw/day	40,000	na	na	na	0.0061	na	na	na
1	by reach	74–494	µg/kg bw/day	40,000	na	na	na	0.0018–0.012	na	na	na
2	site wide	311	µg/kg bw/day	40,000	na	na	na	0.0078	na	na	na
2	RM ≥ 6	367	µg/kg bw/day	40,000	na	na	na	0.0092	na	na	na
2	by reach	93–482	µg/kg bw/day	40,000	na	na	na	0.0023–0.012	na	na	na
Total PCBs											
1	site wide	353	µg/kg bw/day	140	1,400	400	500	**2.5**	0.25	0.88	0.71
1	RM ≥ 6	357	µg/kg bw/day	140	1,400	400	500	**2.6**	0.26	0.89	0.71
1	by reach	130–445	µg/kg bw/day	140	1,400	400	500	0.93–**3.2**	0.093–0.32	0.33–**1.1**	0.26–0.89
2	site wide	293	µg/kg bw/day	140	1,400	400	500	**2.1**	0.21	0.73	0.59
2	RM ≥ 6	312	µg/kg bw/day	140	1,400	400	500	**2.2**	0.22	0.78	0.62
2	by reach	161–340	µg/kg bw/day	140	1,400	400	500	**1.2–2.6**	0.11–0.26	0.42–0.89	0.32–0.71

Table 8-14. Belted kingfisher dietary HQs

Diet Scenario	Area	Dose	Units	Range of TRVs[a]				Range of HQs[a]			
				TRV-A[b]		TRV-B[c]		HQ Based on TRV-A[b]		HQ Based on TRV-B[c]	
				NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL
PCB TEQ - bird											
1	site wide	34	ng/kg bw/day	14	140	2.8	28	**2.5**	0.25	**12**	**1.2**
	RM ≥ 6	35						**2.5**	0.25	**13**	**1.3**
	by reach	14–40						**1.0–3.0**	0.10–0.29	**4.9–15**	0.49–**1.5**
2	site wide	33						**2.3**	0.23	**12**	**1.2**
	RM ≥ 6	39						**2.8**	0.28	**14**	**1.4**
	by reach	17–43						**1.2–3.1**	0.12–0.31	**6.0–15**	0.60–**1.5**
PCDD/PCDF TEQ - bird											
1	site wide	13	ng/kg bw/day	14	140	2.8	28	**2.7**	0.27	**13**	**1.3**
	RM ≥ 6	40						**2.9**	0.29	**14**	**1.4**
	by reach	15–53						0.90–**3.8**	0.090–0.38	**4.5–19**	0.45–**1.9**
2	site wide	13						**2.1**	0.21	**10**	**1.0**
	RM ≥ 6	33						**2.4**	0.24	**12**	**1.2**
	by reach	14–40						0.90–**3.1**	0.090–0.31	**4.5–16**	0.45–**1.6**

Table 8-14. Belted kingfisher dietary HQs

Diet Scenario	Area	Dose	Units	Range of TRVs[a] TRV-A[b] NOAEL	Range of TRVs[a] TRV-A[b] LOAEL	Range of TRVs[a] TRV-B[c] NOAEL	Range of TRVs[a] TRV-B[c] LOAEL	Range of HQs[a] HQ Based on TRV-A[b] NOAEL	Range of HQs[a] HQ Based on TRV-A[b] LOAEL	Range of HQs[a] HQ Based on TRV-B[c] NOAEL	Range of HQs[a] HQ Based on TRV-B[c] LOAEL
Total TEQ - bird											
1	site wide	70	ng/kg bw/day	14	140	2.8	28	**5.0**	0.50	**25**	**2.5**
1	RM ≥ 6	75	ng/kg bw/day	14	140	2.8	28	**5.3**	0.53	**27**	**2.7**
1	by reach	25–88	ng/kg bw/day	14	140	2.8	28	**1.8–6.3**	0.18–0.63	**9.1–31**	0.91–**3.1**
2	site wide	60	ng/kg bw/day	14	140	2.8	28	**4.3**	0.43	21	**2.1**
2	RM ≥ 6	70	ng/kg bw/day	14	140	2.8	28	**5.0**	0.50	**25**	**2.5**
2	by reach	18–79	ng/kg bw/day	14	140	2.8	28	**2.1–5.7**	0.11–0.56	**11–28**	**1.0–2.8**
Total DDx											
1	site wide	40	µg/kg bw/day	25	250	0.9	27	**1.6**	0.16	**44**	**1.5**
1	RM ≥ 6	39	µg/kg bw/day	25	250	0.9	27	**1.6**	0.16	**43**	**1.4**
1	by reach	17–49	µg/kg bw/day	25	250	0.9	27	0.66-**2.0**	0.066–0.20	**18–54**	0.61–**1.8**
2	site wide	34	µg/kg bw/day	25	250	0.9	27	**1.4**	0.14	**38**	**1.3**
2	RM ≥ 6	36	µg/kg bw/day	25	250	0.9	27	**1.4**	0.14	**40**	**1.3**
2	by reach	19–41	µg/kg bw/day	25	250	0.9	27	0.75–**1.7**	0.075–0.17	**21–46**	0.69–**1.5**

Bold identifies HQs ≥ 1.0.
Shaded cells identify HQs ≥ 1.0 based on a LOAEL TRV.

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

[b] TRVs were derived from the primary literature review based on process identified in Section 8.1.3.1.

[c] TRVs were based on USEPA's revised draft of the LPR restoration project FFS (Louis Berger et al. 2014) or first draft of the LPR restoration project FFS (Malcolm Pirnie 2007b).

BERA – baseline ecological risk assessment
bw – body weight
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
FFS – focused feasibility study
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
LPAH – low-molecular-weight polycyclic aromatic hydrocarbon
LPR – Lower Passaic River
LPRSA – Lower Passaic River Study Area
na – not applicable (no TRV available)
NJDEP – New Jersey Department of Environmental Protection
NOAEL – no-observed-adverse-effect level
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
PCDD – polychlorinated dibenzo-*p*-dioxin
PCDF – polychlorinated dibenzofuran
RM – river mile
TEQ – toxic equivalent
TRV – toxicity reference value
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT and 4,4′-DDT)
USEPA – US Environmental Protection Agency

Lead

LOAEL HQs for lead ranged from 0.044 to 0.71 on a site-wide basis, from 0.049 to 0.83 for RM ≥ 6, and from 0.015 to 1.1 by reach. The LOAEL HQ was ≥ 1.0 (1.1) for lead only in Reach 4 (RM 6 to RM 8) using diet Scenario 2 (higher percentage of prey items greater than 13 cm in length).

Methylmercury

LOAEL HQs for methylmercury ranged from 0.22 to 1.3 on a site-wide basis, from 0.23 to 0.81 for RM ≥ 6, and from 0.13 to 1.6 by reach. Under diet Scenario 1, LOAEL HQs were ≥ 1.0 for methylmercury only in Reaches 1 (RM 0 to RM 2) and 2 (RM 2 to RM 4), with HQs of 1.6 and 1.4, respectively. Under diet Scenario 2, LOAEL HQs were ≥ 1.0 on a site-wide basis (1.3), and ranged from 1.3 to 1.6 on a reach-specific basis in Reaches 1 through 3 (RM 0 to RM 6).

PCB TEQ - bird, PCDD/PCDF TEQ - bird, and total TEQ - bird

LOAEL HQs for PCB TEQ ranged from 0.23 to 1.2 on a site-wide basis, from 0.25 to 1.4 for RM ≥ 6, and from 0.10 to 1.5 by reach. LOAEL HQs for PCDD/PCDF TEQ ranged from 0.21 to 1.3 on a site-wide basis, from 0.24 to 1.4 for RM ≥ 6, and from 0.090 to 1.9 by reach. LOAEL HQs for total TEQ ranged from 0.43 to 2.5 on a site-wide basis, from 0.50 to 2.7 for RM ≥ 6, and from 0.11 to 3.1 by reach. LOAEL HQs ≥ 1.0 for all TEQs existed under both diet scenarios and all three exposure area assumptions. HQs ≥ 1.0 ranged from 1.1 to 1.5 for PCB TEQ, from 1.0 to 1.9 for PCDD/PCDF TEQ, and from 1.0 to 3.1 for total TEQ.

Total DDx

LOAEL HQs ranged from 0.14 to 1.5 on a site-wide basis, from 0.14 to 1.4 for RM ≥ 6, and from 0.066 to 1.8 by reach. LOAEL HQs were ≥ 1.0 under both diet scenarios and all three exposure area assumptions, ranging from 1.0 to 1.8.

8.1.4.2 Uncertainties in risk characterization

This section discusses uncertainties that affect HQ calculations for birds through their diet. This evaluation was conducted for COPECs with LOAEL HQs ≥1.0, as identified in Section 8.1.4.1. This section discusses and presents an analysis of uncertainties in the diet composition and exposure area assumptions, EPC calculations, and TRVs. In addition, a discussion of uncertainties associated with TEFs used to calculate TEQs is presented in this section.

Dietary Composition Uncertainties

For great blue heron and belted kingfisher, two dietary exposure scenarios were evaluated in the HQ calculations: Scenario 1, in which only small fish were consumed (0 to 13 cm for great blue heron and 0 to 9 cm for belted kingfisher); and Scenario 2, in which percentages of different size classes of fish varied and included larger fish (see Table 8-4). In general, the use of the Scenario 2 dietary assumptions resulted in slightly higher HQs for great blue heron and belted kingfisher. Studies used for quantifying prey composition with larger size classes of fish show considerable variability with regard to prey sizes greater than 13 cm. Given these uncertainties, prey fractions under Scenario 2 could overestimate exposure for great blue heron and belted kingfisher.

Exposure Area Uncertainties

Risk to each of the bird species was evaluated on a reach-specific basis (each reach is approximately 2 mi in length). Risk to belted kingfisher was evaluated on a site-wide basis, in addition to the exposure area that included only the LPRSA above RM 6.

During their reproductive season, spotted sandpiper may have relatively smaller territories than during other life stages in order to defend their nests and feed their young (Section 8.1.2.3). In addition, spotted sandpiper have preferential habitats for foraging and may forage in smaller areas if adequate food is available. Therefore, it is reasonable to assume that a spotted sandpiper exposure area may be limited to a 2-mi stretch of river beach (i.e., the approximate size of the area for a breeding pair). However, there is greater uncertainty in assuming that great blue heron would be limited to a 2-mi stretch because of their large foraging range (see great blue heron exposure area discussion in Section 8.1.2.3). There is also uncertainty in using the entire site for belted kingfisher because of site-specific information indicating that their breeding habitat is limited within the lower 6 mi of the Passaic River (Baron 2011).

An additional uncertainty associated with evaluating risks on a reach-specific basis (rather than site wide) is that the available data for estimating EPCs is reduced, thereby increasing the likelihood that a maximum detected concentration will be used to represent the EPC. It is possible that using the maximum concentration could overestimate exposure.

Exposure Assumptions and EPC Uncertainties

A quantitative evaluation was conducted by varying certain exposure parameter assumptions and EPC calculations to determine the effect on HQs. The exposure assumptions and EPC uncertainties that were evaluated are as follows:

- **Body weight –** The average of the male and female body weights was used in the HQ calculations. The effect on HQs of using the maximum and minimum male or female body weights reported in USEPA's *Wildlife Exposure Factors Handbook* (USEPA 1993) was evaluated.
- **Sediment ingestion rate –** The SIR for spotted sandpiper (18% of the FIR) was based on the average SIR for four different sandpiper species other than spotted sandpiper, for which data were not available. The highest rates were for semipalmated sandpiper (30%), western sandpiper (18%), and stilt sandpiper (17%), and the lowest rate was for least sandpiper (7.3%). The effect on HQs of using the highest (30%) and lowest (7.3%) SIRs was evaluated. The SIRs for

great blue heron (1% of the FIR) and belted kingfisher (0.5% of the FIR) were based on species feeding habits and best professional judgment, and were adjusted (0 and 2% of the FIR) to determine effects on HQs.

- **Food ingestion rate –** FIRs used for spotted sandpiper, great blue heron, and belted kingfisher were approximately 64, 18, and 50% of their body weights, respectively (using body weights and FIRs presented in Table 8-3). The effect on HQs of using alternative FIRs (±2%) was evaluated.
- **Site use factor –** An SUF of 1 was used for spotted sandpiper, belted kingfisher, and great blue heron exposure. This SUF assumes there are no habitat constraints that could limit the use of the LPRSA by birds for breeding or foraging. The effect on HQs of using an alternative SUF of 0.5 was evaluated for great blue heron because of the relatively large foraging range of the species.
- **Prey size –** The original HQ calculations for great blue heron and belted kingfisher assumed that these species consumed only fish less than a certain size limit in Scenario 1. This updated evaluation considered the possibility that great blue heron and belted kingfisher consume fish of any size and without any preference for a particular size class. This could be an overestimate of risk.
- **Crab consumption by great blue heron –** The original HQ calculations assumed that great blue heron did not consume crab as part of their diet. This updated analysis evaluated the possibility that crab could comprise either 1 or 5% of the heron's diet.
- **Treatment of non-detects for EPCs –** The concentrations of individual compounds or congeners that were not detected were assumed to be zero when calculating sums for total HPAHs, total DDx, and total PCB congeners. The effect on HQs of using one-half the DL or the full DL (rather than zero) was evaluated for calculating the sums of these organic compounds. For TEQ sums, USEPA's TEQ calculator (USEPA 2014) was used, which incorporates the Kaplan-Meier method. The effects of using zero, one-half, or the full DL on HQ calculations, rather than the Kaplan-Meier method, was also evaluated.

The differences in HQs due to the uncertainties in exposure assumptions discussed above were calculated for only the Scenario 1 diets and site-wide exposure assumptions. The results are similar for the other potential dietary/exposure assumption combinations; they are not shown herein because of the large number of values that would need to be presented. The uncertainty calculations were also conducted only for chemicals that had LOAEL HQs ≥ 1.0 (Table 8-15 for copper, lead, and methylmercury; Table 8-16 for PCB TEQ - bird, PCDD/PCDF TEQ - bird, and total TEQ - bird; and Table 8-17 for total HPAHs, total PCBs, and total DDx).

Table 8-15. Bird dietary HQs for copper, lead, and methylmercury based on uncertainty evaluation

	Parameter Values/ Assumptions		Range of LOAEL HQs[a]											
			Copper				Lead				Methyl Mercury			
			HQ Based on TRV-A[b]		HQ Based on TRV-B[c]		HQ Based on TRV-A[b]		HQ Based on TRV-B[c]		HQ Based on TRV-A[b]		HQ Based on TRV-B[c]	
Uncertainty	Orig.	Adj.	Orig.	Adj.	Orig.	Adj.	Orig.	Adj.	Orig.	Adj.	Orig.	Adj.	Orig.	Adj.
Spotted sandpiper (site wide)														
Body weight	0.0425 kg	0.0471 kg	0.50	0.51	**2.0**	**1.8**	0.5	0.51	**7.3**	**7.5**	0.021	0.021	0.08	0.08
		0.0379 kg		0.51		**2.3**		0.51		**7.5**		0.021		0.08
SIR	18% of FIR	7.3% of FIR		0.31		**1.3**		0.28		**4.1**		0.020		0.07
		30% of FIR		0.71		**2.9**		0.74		**11**		0.022		0.08
FIR	64% of bw	62% of bw		0.49		**2.0**		0.49		**7.2**		0.020		0.07
		66% of bw		0.53		**2.0**		0.53		**7.4**		0.022		0.08
Great blue heron (Scenario 1, site wide)														
Body weight	2.3 kg	2.2 kg	0.033	0.033	0.13	0.14	0.018	0.019	0.27	0.27	0.10	0.10	0.37	0.38
		2.6 kg		0.033		0.13		0.019		0.27		0.10		0.37
SIR	1% of FIR	0% of FIR		0.029		0.12		0.014		0.21		0.10		0.37
		2% of FIR		0.036		0.15		0.022		0.33		0.10		0.37
FIR	18% of bw	16% of bw		0.030		0.12		0.017		0.25		0.09		0.34
		20% of bw		0.038		0.15		0.021		0.31		0.11		0.42
Proportion of crab in diet	0%	1%		0.035		0.14		0.018		0.27		0.10		0.37
		5%		0.043		0.17		0.018		0.26		0.11		0.39
SUF	1	0.5		0.016		0.07		0.009		0.14		0.05		0.18
Prey size	preference among fish sizes	no fish size preference		0.049		0.20		0.009		0.14		0.27		**1.0**

Table 8-15. Bird dietary HQs for copper, lead, and methylmercury based on uncertainty evaluation

Uncertainty	Parameter Values/ Assumptions		Range of LOAEL HQs[a]											
			Copper				Lead				Methyl Mercury			
			HQ Based on TRV-A[b]		HQ Based on TRV-B[c]		HQ Based on TRV-A[b]		HQ Based on TRV-B[c]		HQ Based on TRV-A[b]		HQ Based on TRV-B[c]	
	Orig.	Adj.	Orig.	Adj.	Orig.	Adj.	Orig.	Adj.	Orig.	Adj.	Orig.	Adj.	Orig.	Adj.
Belted kingfisher (Scenario 1, site wide)														
Body weight	0.147 kg	0.136 kg	0.18	0.18	0.72	0.74	0.044	0.049	0.72	0.72	0.25	0.25	0.92	0.92
		0.158 kg		0.18		0.74		0.049		0.72		0.25		0.92
SIR	1% of FIR	0% of FIR		0.17		0.71		0.039		0.58		0.25		0.92
		2% of FIR		0.19		0.78		0.059		0.86		0.25		0.92
FIR	50% of bw	48% bw		0.18		0.71		0.047		0.69		0.24		0.89
		52% bw		0.19		0.76		0.050		0.74		0.24		0.95
Prey size	preference among fish sizes	no fish size preference		0.21		0.83		0.04		0.54		0.42		**1.6**

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

[b] TRVs were derived from the primary literature review based on process identified in Section 8.1.3.1.

[c] TRVs were based on USEPA's revised draft of the LPR restoration project FFS (Louis Berger et al. 2014) or first draft of the LPR restoration project FFS (Malcolm Pirnie 2007b).

BERA – baseline ecological risk assessment
bw – body weight
FFS – focused feasibility study
FIR – food ingestion rate
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
LPR – Lower Passaic River
LPRSA – Lower Passaic River study Area
NJDEP – New Jersey Department of Environmental Protection
NOAEL – no-observed-adverse-effect level
SIR – sediment ingestion rate
SUF – site use factor
TRV – toxicity reference value
USEPA – US Environmental Protection Agency

Table 8-16. Bird dietary HQs for PCB TEQ - bird, PCDD/PCDF TEQ - bird, and total TEQ - bird based on uncertainty evaluation

Uncertainty	Parameter Values/ Assumptions		Range of LOAEL HQs[a]											
			PCB TEQ - Bird				PCDD/PCDF TEQ - Bird				Total TEQ - Bird			
			HQ Based on TRV-A[b]		HQ Based on TRV-B[c]		HQ Based on TRV-A[b]		HQ Based on TRV-B[c]		HQ Based on TRV-A[b]		HQ Based on TRV-B[c]	
	Orig.	Adj.	Orig.	Adj.	Orig.	Adj.	Orig.	Adj.	Orig.	Adj.	Orig.	Adj.	Orig.	Adj.
Spotted sandpiper (site wide)														
Body weight	0.0425 kg	0.0471 kg	0.31	0.32	**1.5**	**1.6**	0.91	0.93	**4.5**	**4.7**	**1.1**	**1.1**	**5.4**	**5.6**
		0.0379 kg		0.31		**1.6**		0.93		**4.6**		**1.1**		**5.6**
SIR	18% of FIR	7.3% of FIR		0.21		**1.0**		0.52		**2.6**		0.62		**3.1**
		30% of FIR		0.43		**2.1**		1.3		**6.7**		**1.6**		**8.1**
FIR	64% of bw	62% of bw		0.30		**1.5**		0.90		**4.5**		**1.1**		**5.4**
		66% of bw		0.33		**1.6**		1.0		**4.8**		**1.2**		**5.8**
Treatment of non-detects	use of the Kaplan-Meier method in USEPA's TEQ calculator (USEPA 2014)	use of DL = 0, one-half the DL or the full DL for non-detects		0.31		1.6		0.91		4.5		1.1		5.4

Table 8-16. Bird dietary HQs for PCB TEQ - bird, PCDD/PCDF TEQ - bird, and total TEQ - bird based on uncertainty evaluation

			Range of LOAEL HQs[a]											
			PCB TEQ - Bird				PCDD/PCDF TEQ - Bird				Total TEQ - Bird			
	Parameter Values/ Assumptions		HQ Based on TRV-A[b]		HQ Based on TRV-B[c]		HQ Based on TRV-A[b]		HQ Based on TRV-B[c]		HQ Based on TRV-A[b]		HQ Based on TRV-B[c]	
Uncertainty	**Orig.**	**Adj.**	**Orig.**	**Adj.**	**Orig.**	**Adj.**	**Orig.**	**Adj.**	**Orig.**	**Adj.**	**Orig.**	**Adj.**	**Orig.**	**Adj.**
Great blue heron (Scenario 1, site wide)														
Body weight	2.3 kg	2.2 kg		0.068		0.34		0.068		0.34		0.13		0.65
		2.6 kg		0.068		0.34		0.068		0.34		0.13		0.65
SIR	1% of FIR	0% of FIR		0.065		0.32		0.060		0.30		0.12		0.60
		2% of FIR		0.068		0.34		0.074		0.37		0.14		0.68
FIR	18% of bw	16% of bw		0.062		0.31		0.062		0.31		0.12		0.59
		20% of bw		0.076		0.38		0.077		0.38		0.15		0.74
Proportion of crab in diet	0%	1%		0.067		0.34		0.067		0.34		0.13		0.65
		5%		0.069		0.35		0.068		0.34		0.13		0.66
SUF	1	0.5	0.067	0.033	0.33	0.17	0.067	0.033	0.33	0.17	0.13	0.060	0.64	0.32
Treatment of non-detects	use of the Kaplan-Meier method in USEPA's TEQ calculator (USEPA 2014)	use of DL = 0 one-half the DL or the full DL for non-detects		0.067		0.33		0.067		0.33		0.13		0.64
Prey size	preference among fish sizes	no fish size preference		0.14		0.69		0.23		**1.2**		0.36		**1.8**

Table 8-16. Bird dietary HQs for PCB TEQ - bird, PCDD/PCDF TEQ - bird, and total TEQ - bird based on uncertainty evaluation

	Parameter Values/ Assumptions		Range of LOAEL HQs[a]											
			PCB TEQ - Bird				PCDD/PCDF TEQ - Bird				Total TEQ - Bird			
			HQ Based on TRV-A[b]		HQ Based on TRV-B[c]		HQ Based on TRV-A[b]		HQ Based on TRV-B[c]		HQ Based on TRV-A[b]		HQ Based on TRV-B[c]	
Uncertainty	Orig.	Adj.	Orig.	Adj.	Orig.	Adj.	Orig.	Adj.	Orig.	Adj.	Orig.	Adj.	Orig.	Adj.
Belted kingfisher (Scenario 1, site wide)														
Body weight	0.147 kg	0.136 kg	0.25	0.25	**1.3**	**1.2**	0.27	0.27	**1.4**	**1.4**	0.53	0.51	**2.7**	**2.5**
		0.158 kg		0.25		**1.2**		0.27		**1.4**		0.51		**2.5**
SIR	1% of FIR	0% of FIR		0.24		**1.2**		0.26		**1.3**		0.49		**2.4**
		2% of FIR		0.25		**1.3**		0.29		**1.5**		0.53		**2.6**
FIR	50% of bw	48% bw		0.24		**1.2**		0.26		**1.3**		0.49		**2.4**
		52% bw		0.28		**1.3**		0.28		**1.4**		0.52		**2.6**
Treatment of non-detects	use of the Kaplan-Meier method in USEPA's TEQ calculator (USEPA 2014)	use of DL = 0, one-half the DL or the full DL for non-detects		0.25		1.3		0.27		1.4		0.53		2.7
Prey size	preference among fish sizes	no fish size preference		0.51		**2.6**		0.41		**2.1**		0.93		**4.6**

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

[b] TRVs were derived from the primary literature review based on process identified in Section 8.1.3.1.

[c] TRVs were based on USEPA's revised draft of the LPR restoration project FFS (Louis Berger et al. 2014) or first draft of the LPR restoration project FFS (Malcolm Pirnie 2007b).

BERA – baseline ecological risk assessment
bw – body weight
DL – detection limit
FFS – focused feasibility study
FIR – food ingestion rate
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
LPR – Lower Passaic River Study Area
LPRSA – Lower Passaic River study Area
NJDEP – New Jersey Department of Environmental Protection
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
PCDD – polychlorinated dibenzo-*p*-dioxin
PCDF – polychlorinated dibenzofuran
SIR – sediment ingestion rate
SUF – site use factor
TEQ – toxic equivalent
TRV – toxicity reference value
USEPA – US Environmental Protection Agency

Table 8-17. Bird dietary HQs for total HPAHs, total PCB congeners, and total DDx based on uncertainty evaluation

Uncertainty	Parameter Values/Assumptions		Range of LOAEL HQs[a]											
			Total HPAHs				Total PCB Congeners				Total DDx			
			HQ Based on TRV-A[b]		HQ Based on TRV-B[c]		HQ Based on TRV-A[b]		HQ Based on TRV-B[c]		HQ Based on TRV-A[b]		HQ Based on TRV-B[c]	
	Orig.	Adj.	Orig.	Adj.	Orig.	Adj.	Orig.	Adj.	Orig.	Adj.	Orig.	Adj.	Orig.	Adj.
Spotted sandpiper (site wide)														
Body weight	0.0425 kg	0.0471 kg	na	na	**4.9**	**5.0**	0.17	0.18	0.48	0.49	0.069	0.70	0.64	0.65
		0.0379 kg		na		**4.9**		0.17		0.49		0.70		0.65
SIR	18% of FIR	7.3% of FIR		na		**3.5**		0.13		0.37		0.051		0.48
		30% of FIR		na		**6.4**		0.22		0.61		0.088		0.82
FIR	64% of bw	62% of bw		na		**4.8**		0.17		0.47		0.067		0.62
		66% of bw		na		**5.1**		0.18		0.51		0.072		0.67
Treatment of non-detects	DL = 0 for non-detects	use of one-half the DL or the full DL for non-detects		na		4.9		0.17		0.48		0.092		0.85

Table 8-17. Bird dietary HQs for total HPAHs, total PCB congeners, and total DDx based on uncertainty evaluation

Uncertainty	Parameter Values/Assumptions		Range of LOAEL HQs[a]											
			Total HPAHs				Total PCB Congeners				Total DDx			
			HQ Based on TRV-A[b]		HQ Based on TRV-B[c]		HQ Based on TRV-A[b]		HQ Based on TRV-B[c]		HQ Based on TRV-A[b]		HQ Based on TRV-B[c]	
	Orig.	Adj.	Orig.	Adj.	Orig.	Adj.	Orig.	Adj.	Orig.	Adj.	Orig.	Adj.	Orig.	Adj.
Great Blue Heron (Scenario 1, site wide)														
Body weight	2.3 kg	2.2 kg	na	na	0.19	0.19	0.070	0.072	0.20	0.20	0.043	0.044	0.40	0.41
		2.6 kg		na		0.19		0.071		0.20		0.044		0.41
SIR	1% of FIR	0% of FIR		na		0.16		0.070		0.19		0.043		0.40
		2% of FIR		na		0.21		0.071		0.20		0.044		0.41
FIR	18% of bw	16% of bw		na		0.17		0.07		0.18		0.04		0.37
		20% of bw		na		0.21		0.08		0.23		0.05		0.46
Proportion of crab in diet	0%	1%		na		0.18		0.070		0.20		0.043		0.40
		5%		na		0.18		0.069		0.19		0.044		0.40
SUF	1	0.5		na		0.09		0.04		0.10		0.02		0.20
Treatment of non-detects	DL = 0 for non-detects	use of one-half the DL or the full DL for non-detects		na		0.18		0.040		0.200		0.044		0.41
Prey size	preference among fish sizes	no fish size preference		na		0.070		0.30		0.84		0.16		**1.5**

Table 8-17. Bird dietary HQs for total HPAHs, total PCB congeners, and total DDx based on uncertainty evaluation

			Range of LOAEL HQs[a]											
			Total HPAHs				Total PCB Congeners				Total DDx			
	Parameter Values/Assumptions		HQ Based on TRV-A[b]		HQ Based on TRV-B[c]		HQ Based on TRV-A[b]		HQ Based on TRV-B[c]		HQ Based on TRV-A[b]		HQ Based on TRV-B[c]	
Uncertainty	Orig.	Adj.	Orig.	Adj.	Orig.	Adj.	Orig.	Adj.	Orig.	Adj.	Orig.	Adj.	Orig.	Adj.
Belted Kingfisher (Scenario 1, site wide)														
Body weight	0.147 kg	0.136 kg	na	na	0.36	0.41	0.25	0.25	0.71	0.70	0.16	0.16	**1.4**	**1.5**
		0.158 kg		na		0.41		0.25		0.70		0.16		**1.5**
SIR	1% of FIR	0% of FIR		na		0.31		0.25		0.70		0.16		**1.5**
		2% of FIR		na		0.50		0.25		0.71		0.16		**1.5**
FIR	50% of bw	48% bw		na		0.39		0.24		0.68		0.15		**1.4**
		52% bw		na		0.42		0.26		0.73		0.16		**1.5**
Treatment of non-detects	DL = 0 for non-detects	use of one-half the DL or the full DL for non-detects		na		0.37		0.25		0.71		0.16		1.5
Prey size	preference among fish sizes	no fish size preference		na		0.34		0.5		**1.5**		0.2		**2.1**

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

[b] TRVs were derived from the primary literature review based on process identified in Section 8.1.3.1.

[c] TRVs were based on USEPA's revised draft of the LPR restoration project FFS (Louis Berger et al. 2014), first draft of the LPR restoration project FFS (Malcolm Pirnie 2007b), or LPR restoration project PAR (Battelle 2005).

BERA – baseline ecological risk assessment
bw – body weight
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DL – detection limit
DDT – dichlorodiphenyltrichloroethane
FFS – focused feasibility study
FIR – food ingestion rate
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
LPR – Lower Passaic River
LPRSA – Lower Passaic River Study Area
na – not applicable (no TRV available)
NJDEP – New Jersey Department of Environmental Protection
PAR – pathways analysis report
PCB – polychlorinated biphenyl
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT and 4,4′-DDT)
SIR – sediment ingestion rate
SUF – site use factor
TRV – toxicity reference value
USEPA – US Environmental Protection Agency

The results of this evaluation are summarized in Table 8-18 as well as in the following bullets:

- **Body weight and food ingestion rate –** Differences in these parameters generally had a relatively small effect on the risk estimates, with the difference in HQ values being a maximum of ±0.4 units.
- **Sediment ingestion rate –** Varying the SIR for heron and belted kingfisher resulted in a maximum difference in HQ values of ±0.14 units. However, uncertainty in the sandpiper SIR is greater than for great blue heron and belted kingfisher, and resulted in a maximum difference in HQ values of ±3.7 units.
- **Site use factor –** The use of an SUF of 0.5 rather than 1.0 for great blue heron decreased all HQs by one-half. There were no site-wide HQs ≥ 1.0 for great blue heron; however, the HQ exceedances by reach (see Tables 8-15 through Table 8-17) would be half as much using an SUF of 0.5.
- **Prey size –** As a result of not assigning prey preferences to fish consumption in the diets of great blue heron and belted kingfisher, HQ values differed by a maximum of ±1.9 units, and resulted in LOAEL HQs that were < 1.0 that became ≥ 1.0 in some cases.
- **Crab consumption by great blue heron –** Inclusion of crab in the diet of great blue heron resulted in minimal changes to the HQs (maximum of ±0.04 units).
- **Treatment of non-detects for EPCs –** Different treatments of non-detects for calculating EPCs for organic compounds and TEQs – bird resulted in only small changes to the HQs (maximum of ±0.1 units).

Table 8-18. Summary of uncertainties evaluated for bird dietary evaluation

Species	General Uncertainty Evaluated	Specific Evaluation Conducted	Rationale	Difference in HQ[a] Values
Spotted sandpiper	average body weight	Include the minimum and maximum male and female body weights reported in USEPA (1993).	Evaluate effect on risk estimates based on minimum and maximum body weights.	≤ 0.3 (±)
Great blue heron				≤ 0.01 (±)
Belted kingfisher				≤ 0.2 (±)
Spotted sandpiper	SIR of 18% based on best professional judgement	Include SIRs of 7.3 and 30%.	Evaluate effect on risk estimates based on reasonable range to bracket the original estimate.	≤ 3.7 (±)

Table 8-18. Summary of uncertainties evaluated for bird dietary evaluation

Species	General Uncertainty Evaluated	Specific Evaluation Conducted	Rationale	Difference in HQ[a] Values
Great blue heron	SIR of 1% based on best professional judgement	Include SIRs of 0 and 2%.	Evaluate effect on risk estimates based on reasonable range to bracket the original estimate.	≤ 0.06 (±)
Belted kingfisher				≤ 0.14 (±)
Spotted sandpiper	FIR of 64% of the body weight	62% and 66% of bw	Evaluate effect on risk estimates based on the minimum and maximum FIRs.	≤ 0.4 (±)
Great blue heron	FIR of 18% of the body weight	16% and 20% of bw	Evaluate effect on risk estimates based on the minimum and maximum FIRs.	≤ 0.10 (±)
Belted kingfisher	FIR of 50% of the body weight	48% and 52% of bw	Evaluate effect on risk estimates based on the minimum and maximum FIRs.	≤ 0.3 (±)
Great blue heron	SUF of 1	SUF of 0.5	Evaluate effect on risk of assuming a lower SUF.	≤ 0.32 (-)
Great blue heron	prey consumption limited by size	no size preference for prey consumption	Evaluate effect on risk of a different scenario for prey consumption.	≤ 1.2 (±)
Belted kingfisher				≤ 1.9 (±)
Great blue heron	no crab in the diet	Include crab ingestion rates of 1 and 5% in diet.	Evaluate effect of including crab in the diet of great blue heron.	≤ 0.04 (±)
Spotted sandpiper	DL = 0 for non-detects for total HPAHs, total PCB congeners, and total DDx	Include use of one-half of DL or the full DL for non-detects	Evaluate effect of using different treatments of non-detects in calculating sums for organic compounds.	≤ 0.1 (±)
Great blue heron				≤ 0.01 (±)
Belted kingfisher				≤ 0.1 (±)
Spotted sandpiper	use of the Kaplan-Meier method for TEQ sums	Include use of zero, one-half of DL, or the full DL for non-detects	Evaluate effect of using different treatments of non-detects in calculating sums for TEQs - bird.	≤ 0.1 (+)
Great blue heron				0
Belted kingfisher				0

[a] Differences in HQs (based on a LOAEL TRV) were calculated from the data presented in Tables 8-15 through 8-17 and are based on the site-wide exposure area and diet Scenario 1 for spotted sandpiper (100% benthic invertebrates), great blue heron (100% 0–13 cm fish), and belted kingfisher (15% blue crab and 85% 0–9 cm fish). Direction of the HQ change is provided in parentheses.

bw – body weight
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DL – detection limit
DDT – dichlorodiphenyltrichloroethane
EPC – exposure point concentration
FIR – food ingestion rate
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
SIR – sediment ingestion rate
SUF – site use factor
PCB – polychlorinated biphenyl
TEQ – toxic equivalent
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT and 4,4′-DDT)
TRV – toxicity reference value

TEQ Uncertainty

TEQs represent uncertain estimates because they are calculated using TEFs that are highly variable. The four most-potent dioxin-like PCBs in birds are PCBs 77, 81, 126, and 169 (i.e., those with the highest TEFs and thus contributing most to the TEQ). For PCB 77, five studies produced TEFs ranging over three orders of magnitude (< 0.0003 to 0.15) for the various bird species tested (Van den Berg et al. 1998). For PCB 81, two studies tested several species and identified TEFs ranged from 0.001 to 0.5. For PCBs 126 and 169, data were available from only one study, so the associated uncertainty has not yet been quantified.

The high variability in TEFs may be due, in part, to differences in species sensitivity to dioxin-like compounds. Bird species can be grouped into three general classes of sensitivity to dioxin-like compounds, based on documented species differences in the amino acid sequences at the Ah receptor. These species differences affect not only overall sensitivity, but also the relative potency (i.e., TEFs) of individual dioxin-like compounds (Farmahin et al. 2013). Another issue with bird TEFs is their relevance for assessing dietary exposure risks. TEFs are estimated based on ethoxyresorufin-O-deethylase induction or *in ovo* studies. Such studies are most accurate for assessing effects based on tissue congener concentrations in whole embryos (USEPA 2008), not dietary exposure studies.

Ring-necked pheasants and chickens (the two species in studies used to derive TEQ TRVs) are in the moderate- and high-sensitivity groups, respectively, as classified by Farmahin et al. (2013). The derivation of TRVs using moderately to highly sensitive species indicates that the risk calculations are more likely to overestimate risk than to underestimate risk.

TRV Uncertainty

General TRV uncertainties, including the derivation of TRVs using SSDs, are discussed in Sections 8.1.3.2 and 6.3.3.1. For the COPECs with TRVs based on 5th percentile LOAELs determined from SSDs (i.e., methylmercury and total DDx), the range of the empirical LOAELs and number of data points (i.e., number of species included in the SSD) are shown in Table 8-19 to provide a context of uncertainty for SSD-derived values.

Table 8-19. Uncertainty evaluation of bird diet TRVs based on SSDs

COPEC	TRV Unit	NOAEL	LOAEL	No. of species (count of LOAELs in SSD)	Empirical LOAEL Range	Notes on Key Uncertainties
Methylmercury	µg/kg bw/day	50	96	n = 6	64–8,780	• SSD-derived LOAEL is within range of measured LOAELs
Total DDx	µg/kg bw/day	190	250	n = 10	150–71,000	• SSD-derived LOAEL is within range of measured LOAELs

Note: TRVs included in this table are based on SSDs that are based on TRVs derived from the general literature search.

COPEC – chemical of potential ecological concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
SSD – species sensitivity distribution
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT and 4,4′-DDT)
TRV – toxicity reference value
ww – wet weight

8.1.4.3 Comparison to background

Consistent with USEPA guidance (USEPA 2002c), this section presents background concentrations for prey for bird dietary COPECs with LOAEL HQs ≥ 1 (copper, lead, methylmercury, total HPAHs, total PCBs, PCB TEQ - bird, PCDD/PCDF TEQ - bird, total TEQ - bird, and total DDx). Three background datasets were developed for use in this BERA using available data from the following areas: 1) the Upper Passaic River upstream of Dundee Dam, to represent freshwater urban habitat; 2) Jamaica Bay/Lower Harbor, to represent estuarine urban habitat; and 3) Mullica River/Great Bay, to represent estuarine/freshwater rural habitat. These datasets are summarized in Section 4.2, and details on how background values were determined from these datasets are presented in Appendix J. Whole-body data were limited to mummichog and other killifish in the Jamaica Bay/Lower Harbor and Mullica River/Great Bay background areas for comparison to LPRSA species. Table 8-20 presents the comparison of LPRSA fish tissue concentrations to background areas, where data are available, for fish COPECs with LOAEL HQs ≥ 1.0.

Fish tissue EPCs for LPRSA fish compared to those above Dundee Dam are summarized as follows:

- For copper, LPRSA EPCs were less than maximum concentrations for 4 of 10 species above Dundee Dam, and less than UCLs for 3 of 4 fish species for which UCLs above Dundee Dam could be calculated. LPRSA EPCs for mummichog were less than the killifish maximum concentrations and UCLs above Dundee Dam.
- For lead, LPRSA EPCs were less than maximum concentrations for 6 of 10 species above Dundee Dam, and less than UCLs for 2 of 4 fish species for which

UCLs above Dundee Dam could be calculated. Mummichog LPRSA EPCs were greater than maximum concentrations and UCLs in mummichog from the Mullica River/Great Bay.

- For methylmercury, LPRSA EPCs were less than maximum concentrations for 7 of 10 species above Dundee Dam, and similar to or less than UCLs for 4 of 4 fish species for which UCLs above Dundee Dam could be calculated. Mummichog LPRSA EPCs were less than maximum concentrations in mummichog from the Mullica River/Great Bay. For mercury, LPRSA EPCs were less than maximum concentrations for 8 of 10 species above Dundee Dam, and similar to or less than UCLs for 3 of 4 fish species for which UCLs above Dundee Dam could be calculated. Mummichog LPRSA EPCs were less than maximum concentrations in mummichog from the Mullica River/Great Bay.
- For total HPAHs, LPRSA EPCs were less than maximum concentrations for 4 of 10 species above Dundee Dam, and less than UCLs for 2 of 4 fish species for which UCLs above Dundee Dam could be calculated. For mummichog/killifish, LPRSA total HPAH EPCs were greater than the maximum concentrations and UCLs from Jamaica Bay/Lower Harbor.
- In general, EPCs for total PCB congeners, PCB TEQ - bird, PCDD/PCDF TEQ - bird, total TEQ - bird, and total DDx in whole-body fish tissue were higher on a fish species basis in fish from the LPRSA than UCLs and maximum concentrations in fish from above Dundee Dam.

In comparison to regional areas, EPCs for organic compounds in whole-body fish tissue were higher in mummichog from the LPRSA than in mummichog from the Mullica River/Great Bay. However, UCLs and maximum concentrations of total PCBs, PCB TEQ - bird, total TEQ - bird (driven by the PCB component of TEQ - bird), and total DDx were higher in mummichog from Jamaica Bay/Lower Harbor than in mummichog from the LPRSA.

The mean mummichog lipid content was higher in Jamaica Bay/Lower Harbor mummichog (3.1%) than in LPRSA mummichog (2.0%). Although the higher mean mummichog lipid content in Jamaica Bay/Lower Harbor could indicate better fish condition, there are other factors that may affect lipid content in fish, such as size, age, sex, reproductive status, genetic background, diet, water temperature, and seasonality (Mraz 2012; Iverson et al. 2002).

The lipid-normalized maximum concentrations were greater for mummichog from Jamaica Bay/Lower Harbor than for mummichog from the LPRSA:

- **Total PCBs**: approximately 2.6 times greater in mummichog from Jamaica Bay/Lower Harbor (94 mg/kg lipid) than in mummichog from the LPRSA (36 mg/kg lipid)

- **PCB TEQ - bird:** approximately 4.2 times greater in mummichog from Jamaica Bay/Lower Harbor (0.011 mg/kg lipid) than in mummichog from the LPRSA (0.0026 mg/kg lipid)
- **Total TEQ - bird:** approximately 1.5 times greater in mummichog from Jamaica Bay/Lower Harbor (0.011 mg/kg lipid) than in mummichog from the LPRSA (0.0074 mg/kg lipid)
- **Total DDx:** approximately 1.7 times greater in mummichog from Jamaica Bay/Lower Harbor (7.1 mg/kg lipid) than in mummichog from the LPRSA (4.2 mg/kg lipid)

Table 8-20. LPRSA fish tissue compared to background tissue for bird dietary COPECs with LOAEL HQs ≥ 1.0

Species	LPRSA				Above Dundee Dam				Jamaica Bay/Lower Harbor				Mullica River/Great Bay			
	N	EPC	Min. Detect	Max. Detect	N	UCL	Min. Detect	Max. Detect	N	UCL	Min. Detect	Max. Detect	N	UCL	Min. Detect	Max. Detect
Copper																
American eel	21	2.6	0.52	7.8	16	0.747	0.415	0.818	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]
Brown bullhead	6	0.86	0.55	0.94	6	1.79	0.487	2.17	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]
Common carp	12	1.1	0.6	1.6	10	1.32	0.926	1.65	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]
Channel catfish	11	1.3	0.31	2.3	4	na	0.39	0.745	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]
Mummichog/killifish[a]	18	3.1	2	4.3	1	na	1.49	1.49	na[b]	na[b]	na[b]	na[b]	10	4.4	2.7	6.0
Northern pike	1	0.57	0.57	0.57	1	na	0.481	0.481	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]
Other forage fish	10	4.1	0.87	5.4	2	na	0.655	0.916	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]
Smallmouth bass	3	0.80	0.4	0.8	3	na	0.315	0.396	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]
White perch	22	14	1.6	50.9	8	14.4	4.94	16.2	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]
White sucker	5	1.1	0.77	1.1	5	na	0.736	1.5	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]
Lead																
American eel	14	0.87	0.18	1.4	16	0.36	0.73	0.702	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]
Brown bullhead	6	0.80	0.15	0.83	6	2.08	0.288	3.63	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]
Common carp	12	0.79	0.21	0.96	10	0.692	0.256	0.859	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]
Channel catfish	11	0.30	0.056	0.37	4	na	0.11	0.32	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]
Mummichog/killifish[a]	18	2.4	0.38	3.9	1	na	0.35	0.35	na[b]	na[b]	na[b]	na[b]	10	0.23	0.16	0.23
Northern pike	1	0.033	0.033	0.033	1	na	0.052	0.052	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]
Other forage fish	10	3.0	0.15	4.9	2	na	0.209	0.476	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]
Smallmouth bass	3	0.098	0.052	0.098	3	na	0.045	0.053	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]
White perch	22	0.44	0.17	0.96	8	0.87	0.26	1.22	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]
White sucker	5	0.30	0.15	0.3	5	na	0.1	1.1	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]

Table 8-20. LPRSA fish tissue compared to background tissue for bird dietary COPECs with LOAEL HQs ≥ 1.0

Species	LPRSA				Above Dundee Dam				Jamaica Bay/Lower Harbor				Mullica River/Great Bay			
	N	EPC	Min. Detect	Max. Detect	N	UCL	Min. Detect	Max. Detect	N	UCL	Min. Detect	Max. Detect	N	UCL	Min. Detect	Max. Detect
Methylmercury																
American eel	21	280	92	470	16	190	121	255	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]
Brown bullhead	6	92	39	120	6	203	29.7	276	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]
Common carp	12	62	39	90	10	110	47.5	131	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]
Channel catfish	11	140	30	230	4	na	140	559	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]
Mummichog/killifish[a]	18	53	19	69	1	na	34.5	34.5	2	na	69.2	71.4	na[b]	na[b]	na[b]	na[b]
Northern pike	1	180	180	180	1	na	316	316	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]
Other forage fish	10	70	14	150	2	na	61.7	110	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]
Smallmouth bass	3	220	140	220	3	na	139	162	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]
White perch	22	170	25	330	8	270	120	373	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]
White sucker	5	130	71	130	5	na	51.3	196	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]
Total HPAHs																
American eel	21	24	3.1	49	16	8.4	0.95	13	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]
Brown bullhead	6	110	21	110	6	416	10.6	732	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]
Common carp	12	76	24	120	10	68.9	23.3	83	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]
Channel catfish	11	55	20	96	4	na	5.08	43.4	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]
Mummichog/killifish[a]	18	260	29	540	1	na	29.2	29.2	7	23	12	17	10	15[d]	nd	nd
Northern pike	1	42	42	42	1	na	20.2	20.2	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]
Other forage fish	10	670	78	1000	2	na	25.8	82.4	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]
Smallmouth bass	3	390	7.3	390	3	na	6.52	10.8	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]
White perch	22	140	21	340	8	144	25.5	218	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]
White sucker	5	48	21	48	5	na	7.4	230	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]

Table 8-20. LPRSA fish tissue compared to background tissue for bird dietary COPECs with LOAEL HQs ≥ 1.0

Species	LPRSA				Above Dundee Dam				Jamaica Bay/Lower Harbor				Mullica River/Great Bay			
	N	EPC	Min. Detect	Max. Detect	N	UCL	Min. Detect	Max. Detect	N	UCL	Min. Detect	Max. Detect	N	UCL	Min. Detect	Max. Detect
Total PCBs (µg/kg ww)																
American eel	21	2,000	420	5,700	16	1,080	206	1,880	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]
Brown bullhead	6	1,400	260	1,700	6	519	183	614	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]
Common carp	12	5,200	1,500	7,900	10	2,100	755	2,560	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]
Channel catfish	11	1,700	350	2,700	4	na	948	2,130	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]
Mummichog/killifish[a]	18	600	240	930	1	na	219	219	7	1,900	55	3,200	na[b]	na[b]	na[b]	na[b]
Northern pike	1	2,000	2,000	2,000	1	na	1,880	1,880	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]
Other forage fish	10	220	170	870	2	na	107	853	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]
Smallmouth bass	3	1,400	630	1,400	3	na	1,000	1,310	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]
White perch	22	2,500	290	5,100	8	834	408	1,130	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]
White sucker	5	2,900	540	2,900	5	na	327	872	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]
PCB TEQ - bird																
American eel	21	15	2.9	23	16	13.6	2.51	17.5	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]
Brown bullhead	6	65	20	84	6	23	9.04	27.3	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]
Common carp	12	200	67	260	10	132	77.2	163	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]
Channel catfish	11	41	12	55	4	na	33.5	60.5	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]
Mummichog/killifish[a]	18	47	17	73	1	na	10.9	10.9	7	630	0.11	410	10	4.9	3	5.8
Northern pike	1	160	160	160	1	na	138	138	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]
Other forage fish	10	69	20	95	2	na	15.6	136	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]
Smallmouth bass	3	67	37	67	3	na	90.3	113	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]
White perch	22	230	31	400	8	85.1	53.5	99.6	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]
White sucker	5	170	64	170	5	na	31.4	104	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]

Table 8-20. LPRSA fish tissue compared to background tissue for bird dietary COPECs with LOAEL HQs ≥ 1.0

Species	LPRSA				Above Dundee Dam				Jamaica Bay/Lower Harbor				Mullica River/Great Bay			
	N	EPC	Min. Detect	Max. Detect	N	UCL	Min. Detect	Max. Detect	N	UCL	Min. Detect	Max. Detect	N	UCL	Min. Detect	Max. Detect
PCDD/PCDF TEQ - bird																
American eel	21	25	0.73	48	16	1.5	0.136	2.6	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]
Brown bullhead	6	160	9.6	210	6	3.39	1.73	3.67	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]
Common carp	12	630	11	1,400	10	10.8	5.84	13.5	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]
Channel catfish	11	100	23	170	4	na	4.28	10.1	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]
Mummichog/killifish[a]	18	54	12	110	1	na	0.858	0.858	7	23	10	30	12	0.5	0.14	0.7
Northern pike	1	120	120	120	1	na	16.1	16.1	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]
Other forage fish	10	53	9.1	100	2	na	0.452	10.7	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]
Smallmouth bass	3	82	9.8	82	3	na	3.19	3.82	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]
White perch	22	210	21	280	8	7.15	4.11	9.39	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]
White sucker	5	140	8	140	5	na	1.51	9.16	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]
Total TEQ - bird																
American eel	21	42	7.8	62	16	14.8	2.47	19.2	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]
Brown bullhead	6	230	31	290	6	26.3	10.8	30.6	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]
Common carp	12	830	77	1,700	10	142	84	171	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]
Channel catfish	11	150	35	230	4	na	37.8	70.3	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]
Mummichog/killifish[a]	18	98	30	170	1	na	11.8	11.8	7	640	28	430	10	5.6	3.4	6.5
Northern pike	1	280	280	280	1	na	154	154	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]
Other forage fish	10	120	32	200	2	na	16.1	147	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]
Smallmouth bass	3	140	57	140	3	na	93.4	117	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]
White perch	22	400	52	690	8	92.2	57.6	109	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]
White sucker	5	310	74	310	5	na	32.9	111	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]

Table 8-20. LPRSA fish tissue compared to background tissue for bird dietary COPECs with LOAEL HQs ≥ 1.0

Species	LPRSA				Above Dundee Dam				Jamaica Bay/Lower Harbor				Mullica River/Great Bay			
	N	EPC	Min. Detect	Max. Detect	N	UCL	Min. Detect	Max. Detect	N	UCL	Min. Detect	Max. Detect	N	UCL	Min. Detect	Max. Detect
Total DDx																
American eel	21	260	32	470	16	270	62	490	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]
Brown bullhead	3	160	20	200	6	67	27	76	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]
Common carp	12	650	110	1100	10	220	87	280	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]
Channel catfish	11	280	48	490	4	na	120	340	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]
Mummichog/killifish[a]	18	66	26	100	1	na	45	45	7	180	10	240	na[b]	na[b]	na[b]	na[b]
Northern pike	1	280	280	280	1	na	230	230	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]
Other forage fish	10	75	22	140	2	na	30	120	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]
Smallmouth bass	3	230	100	230	3	na	140	150	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]
White perch	22	240	38	490	8	150	85	170	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]
White sucker	5	150	63	150	5	na	33	170	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]	na[b]

Note: The maximum detected concentration for background areas exclude outlier concentrations, as described in Appendix J.

a The mummichog/killifish group consists of mummichog from the LPRSA, Jamaica Bay/Lower Harbor, and Mullica/Great Bay, and banded killifish from above Dundee Dam.

b No data available.

DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
EPC – exposure point concentration
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
LPRSA – Lower Passaic River Study Area
na – not applicable
PCB – polychlorinated biphenyl
PCDD – polychlorinated dibenzo-*p*-dioxin
PCDF – polychlorinated dibenzofuran
TEQ – toxic equivalent
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT and 4,4′-DDT)
UCL – upper confidence limit on the mean
ww – wet weight

In comparison to regional areas, EPCs for organic compounds in whole-body fish tissue were higher in mummichog from the LPRSA than in mummichog from the Mullica River/Great Bay. However, UCLs and maximum concentrations of total PCBs, PCB TEQ - bird, total TEQ - bird (driven by the PCB component of TEQ - bird), and total DDx were higher in mummichog from Jamaica Bay/Lower Harbor than in mummichog from the LPRSA.

The mean mummichog lipid content was higher in Jamaica Bay/Lower Harbor mummichog (3.1%) than in LPRSA mummichog (2.0%). Although the higher mean mummichog lipid content in Jamaica Bay/Lower Harbor could indicate better fish condition, there are other factors that may affect lipid content in fish, such as size, age, sex, reproductive status, genetic background, diet, water temperature, and seasonality (Mraz 2012; Iverson et al. 2002).

The lipid-normalized maximum concentrations were greater for mummichog from Jamaica Bay/Lower Harbor than for mummichog from the LPRSA:

- **Total PCBs**: approximately 2.6 times greater in mummichog from Jamaica Bay/Lower Harbor (94 mg/kg lipid) than in mummichog from the LPRSA (36 mg/kg lipid)
- **PCB TEQ - bird:** approximately 4.2 times greater in mummichog from Jamaica Bay/Lower Harbor (0.011 mg/kg lipid) than in mummichog from the LPRSA (0.0026 mg/kg lipid)
- **Total TEQ - bird:** approximately 1.5 times greater in mummichog from Jamaica Bay/Lower Harbor (0.011 mg/kg lipid) than in mummichog from the LPRSA (0.0074 mg/kg lipid)
- **Total DDx:** approximately 1.7 times greater in mummichog from Jamaica Bay/Lower Harbor (7.1 mg/kg lipid) than in mummichog from the LPRSA (4.2 mg/kg lipid)

8.1.5 Summary of key uncertainties

The primary uncertainty associated with the bird dietary risk characterization is the selection of TRVs used in the risk calculations. Uncertainties associated with TRVs are discussed in Section 8.1.3.1. Two TRVs were derived based on SSDs: the TRV for methylmercury and the TRV for total DDx. Both of the SSD-derived LOAELs were within the range of LOAELs measured in the reviewed studies.

The adjustments in the dietary composition to include greater consumption of larger fish (i.e., Scenario 2) resulted in slightly higher HQs for great blue heron and belted kingfisher than did the scenario in which only small fish were consumed (i.e., Scenario 1). Given the uncertainties associated with quantifying prey composition, prey fractions under Scenario 2 could overestimate exposure for great blue heron and belted kingfisher. In addition, the evaluation of risk by reach resulted in slightly higher HQs in some specific reaches. This approach may overestimate risk to great blue heron because the species has a relatively large home range.

Based on the analysis of varying exposure parameters and EPC calculations (i.e., treatment of non-detects in sums) it was found that most of these adjustments did not affect the HQs substantially. The SIR for spotted sandpiper HQs and uncertainties in this value could result in over- or underestimation of risk. In addition, the assumption that great blue heron feed solely from the LPRSA (SUF = 1) could overestimate risk, and the assumption that they obtain half of their food from the LPRSA (SUF = 0.5) could underestimate risk.

Other uncertainties in the bird dietary risk assessment—such as TEQ methodology, the use of laboratory toxicity studies to predict effects, and the use of tissue data from laboratory bioaccumulation studies—could result in under- or overestimation of risks. HQs are more likely to represent an overestimation of risk because of the conservative assumptions used in the risk evaluation. These conservative assumptions include the use of the lowest LOAEL among all species or endpoints as the TRV, the use of an upper exposure value (i.e., UCL) to calculate dietary exposure concentrations, and the assumption that a species feeds exclusively from the LPRSA (i.e., SUF = 1).

8.1.6 Summary

Sixteen dietary COPECs were evaluated for birds. Table 8-21 provides the range in LOAEL HQs for all dietary and exposure area scenarios, using a range of TRVs for the COPECs with LOAEL HQs ≥ 1.0. The following bird species and COPECs had LOAEL HQs ≥ 1.0 on a site-wide basis:

- Spotted sandpiper – total TEQ - bird, PCB TEQ - bird, copper, lead, and total HPAHs
- Belted kingfisher – PCDD/PCDF TEQ - bird, total TEQ - bird, PCB TEQ - bird, and total DDx

Additional bird species and COPECs with LOAEL HQs ≥ 1.0 when risk was evaluated by reach were:

- Spotted sandpiper – total PCBs and total DDx
- Great blue heron – PCDD/PCDF TEQ - bird, total TEQ - bird, total PCBs, PCB TEQ - bird, methylmercury, and total DDx
- Belted kingfisher – lead

Evaluation by reach is likely to overestimate risks to great blue heron because the species has a relatively large home range.

Table 8-21. Summary of bird dietary LOAEL HQs

COPEC[b]	Exposure Area	Range of HQs[a]						Key Uncertainty
		Spotted Sandpiper		Great Blue Heron		Belted Kingfisher		
		HQ Based on TRV-A[c]	HQ Based on TRV-B[d]	HQ Based on TRV-A[c]	HQ Based on TRV-B[d]	HQ Based on TRV-A[c]	HQ Based on TRV-B[d]	
Copper	site wide/RM ≥ 6	0.50	**2.0**	0.033–0.94	0.13–0.38	0.18–0.20	0.72–0.80	• TRVs based on slight reductions in growth (TRV-A based on 10% reduction in chicken growth and TRV-B based on 4% reduction in turkey growth); may overestimate potential adverse effects on LPRSA populations
	by reach	0.3.0–0.88	**1.2–3.6**	0.029–0.33	0.12–**1.3**	0.15–0.24	0.61–0.97	
Lead	site wide/RM ≥ 6	0.49	**7.3**	0.010–0.018	0.15–0.27	0.044–0.056	0.65–0.83	• TRV-A based on 10% reduction in quail growth; may overestimate potential adverse effects on LPRSA populations • TRV-B based on reduced quail egg production; not evident that effects on Japanese quail egg production rates would reflect adverse effects on reproduction in wild birds
	by reach	0.20–0.59	**3.0–10**	0.0041–0.036	0.06–0.52	0.015–0.075	0.22–**1.1**	
Methylmercury	site wide/RM ≥ 6	0.021	0.77	0.10–0.25	0.37–0.94	0.22–0.35	0.81–**1.3**	• TRV-A based on SSD within the range of measured LOAELs • TRV-B derived using interspecies extrapolation factor of 3 based on exposure to methylmercury dicyandiamide, a fungicide that is not a form of mercury expected to be associated with the LPRSA
	by reach	0.0072–0.027	0.027–0.10	0.031–0.42	0.11–**1.6**	0.13–0.43	0.48–**1.6**	
Total HPAHs	site wide/RM ≥ 6	ne[e]	**4.9**	ne[e]	0.10–0.19	ne[e]	0.36–0.61	• TRV-B based on weekly injection study of pigeons with single PAH (benzo[a]pyrene) with interspecies extrapolation factor of 3 applied
	by reach	ne[e]	**1.9–10**	ne[e]	0.04–0.34	ne[e]	0.11–0.80	
Total PCBs	site wide/RM ≥ 6	0.17	0.48	0.24–0.70	0.20–0.66	0.21–0.26	0.59–0.71	• TRV-A based on non-chicken reproduction • TRV-B based on interpolated value from chicken hatchability data
	by reach	0.047–0.41	0.13–**1.2**	0.03–0.41	0.09–**1.1**	0.093–0.32	0.26–0.89	
PCB TEQ - bird	site wide/RM ≥ 6	0.31	**1.5**	0.067–0.18	0.33–0.90	0.23–0.28	**1.2–1.4**	• TRV-A and TRV-B based on same literature source using weekly injection of pheasants • TRV-B extrapolated from study using an interspecies extrapolation factor of 5
	by reach	0.0073–0.78	0.37–**3.9**	0.030–0.33	0.13–**1.6**	0.10–0.31	0.49–**1.5**	
PCDD/PCDF TEQ - bird	site wide/RM ≥ 6	0.91	**4.5**	0.067–0.19	0.33–**1.0**	0.21–0.29	**1.0–1.4**	
	by reach	0.014–**4.2**	0.0171–**21**	0.024–0.37	0.10–**1.9**	0.090–0.38	0.45–**1.9**	

Table 8-21. Summary of bird dietary LOAEL HQs

COPEC[b]	Exposure Area	Spotted Sandpiper: HQ Based on TRV-A[c]	Spotted Sandpiper: HQ Based on TRV-B[d]	Great Blue Heron: HQ Based on TRV-A[c]	Great Blue Heron: HQ Based on TRV-B[d]	Belted Kingfisher: HQ Based on TRV-A[c]	Belted Kingfisher: HQ Based on TRV-B[d]	Key Uncertainty
		Range of HQs[a]						
Total TEQ - bird	site wide/RM ≥ 6	**1.1**	**5.4**	0.067–0.37	0.33–**1.8**	0.43–0.53	**2.1–2.7**	• High variability of bird TEFs and differences in species sensitivities to dioxin-like compounds
	by reach	0.089–**5.0**	0.44–**25**	0.045–0.71	0.22–**3.5**	0.11–0.63	0.91–**3.1**	
Total DDx	site wide/RM ≥ 6	0.069	0.64	0.043–0.14	0.40–**1.3**	0.14–0.16	**1.3–1.5**	• TRV-A based on SSD within range of measured LOAELs evaluated • TRV-B based on field study of eggshell thinning in pelicans
	by reach	0.018–0.15	0.16–**1.4**	0.020–0.26	0.19–**2.4**	0.066–0.20	0.61–**1.8**	

Bold identifies HQs ≥ 1.0.

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

[b] Only COPECs with HQs ≥ 1.0 based on a LOAEL TRV are included in the table.

[c] TRVs were derived from the primary literature review based on process identified in Section 8.1.3.1.

[d] TRVs were based on USEPA's revised draft of the LPR restoration project FFS (Louis Berger et al. 2014), first draft of the LPR restoration project FFS (Malcolm Pirnie 2007b), or LPR restoration project PAR (Battelle 2005).

[e] TRV-A set did not include HPAHs for evaluation; benzo(a)pyrene was evaluated as an individual PAH and LOAEL HQs were < 1 based on this TRV.

BERA – baseline ecological risk assessment
COPEC – chemical of potential ecological concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
FFS – focused feasibility study
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
LPR – Lower Passaic River
LPRSA – Lower Passaic River Study Area
ne – not evaluated
NJDEP – New Jersey Department of Environmental Protection
NOAEL – no-observed-adverse-effect level
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
PCDD – polychlorinated dibenzo-*p*-dioxin
PCDD/PCDF – polychlorinated dibenzofuran
RM – river mile
SSD – species sensitivity distribution
TEQ – toxic equivalent
TRV – toxicity reference value
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT and 4,4′-DDT)

8.2 EGG TISSUE ASSESSMENT

As an additional assessment of reproduction in birds, potential risks to bird eggs from maternal dietary exposure were evaluated in two species: great blue heron and belted kingfisher. In this assessment, biota (prey) tissue data were converted into modeled egg tissue data based on biomagnification assumptions from the literature. Assessing the potential risks to birds based on early life stage (i.e., reproductive) effects was also evaluated using a dietary dose approach (Section 8.1).

8.2.1 COPECs

COPECs for piscivorous bird egg tissue were identified in the SLERA in cases where the maximum modeled egg concentration exceeded TSVs (Appendix A). The bird egg COPECs identified for belted kingfisher and great blue heron are provided in Table 8-22.

Table 8-22 Bird egg COPECs

COPEC	
Metals	
Methylmercury/mercury	
PCBs	
Total PCBs	PCB TEQ - bird
PCDDs/PCDFs	
PCDD/PCDF TEQ - bird	Total TEQ - bird
Pesticides	
Total DDx	Dieldrin

Note: COPECs are those COIs for which the maximum concentration exceeded its TSV.

COI – chemical of interest
COPEC – chemical of potential ecological concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
PCB – polychlorinated biphenyl
PCDD – polychlorinated dibenzo-*p*-dioxin
PCDF – polychlorinated dibenzofuran
TEQ – toxic equivalent
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT and 4,4′-DDT)
TSV – toxicity screening value

For the COPECs in Table 8-22, exposure-based concentrations (Section 8.2.2) were compared with toxicity-based values (Section 8.2.3) to derive bird egg HQs (Section 8.2.4).

8.2.2 Exposure

COPEC concentrations in bird eggs were estimated using the following equation:

$$EPC_{egg} = EPC_{prey} \times BMF \qquad \textbf{Equation 8-5}$$

Where:

EPC_{egg} = exposure point concentration in bird egg tissue(s) (mg/kg ww)
EPC_{prey} = exposure point concentration in prey tissue (mg/kg ww)
BMF = biomagnification factor

8.2.2.1 Prey tissue concentrations

The prey composition and exposure area assumptions for belted kingfisher and great blue heron were consistent with the prey composition scenarios and exposure areas used for the dietary assessment (Section 8.1.2.3). Table 8-23 summarizes the prey compositions and exposure areas used to derive prey tissue concentrations for the two scenarios.

Table 8-23. Summary of prey composition scenarios and exposure areas for bird species

Species Exposure Area	Prey Type	% in Diet	Exposure Areas	Rationale
Great blue heron: mudflats				
Scenario 1	fish 0–13 cm	100	site-wide mudflats	Evaluate great blue heron diet based on the fish size class expected to make up most of their diet.
Scenario 2	fish 0–13 cm	17	site-wide mudflats	Evaluate larger prey as part of great blue heron diet, including very large fish such as common carp.
	fish 13–18 cm	29		
	fish 18–30 cm	40		
	fish >30 cm	14		
Great blue heron: reach specific				
Scenario 1	fish 0–13 cm	100	by reach	Evaluate HQs on reach-specific basis
Scenario 2	fish 0–13 cm	17	by reach	Evaluate HQs on reach-specific basis.
	fish 13–18 cm	29		
	fish 18–30 cm	40		
	fish >30 cm	14		

Species Exposure Area	Prey Type	% in Diet	Exposure Areas	Rationale
Belted kingfisher: RM ≥ 6				
Scenario 1	fish 0–9 cm	85	RM ≥ 6	Evaluate belted kingfisher diet based on the fish size class expected to make up most of their diet in the exposure area where they are most likely to forage (RM ≥ 6).
	blue crab	15		
Scenario 2	blue crab	15	RM ≥ 6	Evaluate larger prey as part of the belted kingfisher diet.
	fish 0–9 cm	31.5		
	fish 9–13 cm	51		
	fish 13–18 cm	2.5		
Belted kingfisher: site wide				
Scenario 1	fish 0–9 cm	85	site wide	Evaluate HQs on site-wide basis.
	blue crab	15		
Scenario 2	blue crab	15	site wide	Evaluate HQs on site-wide basis.
	fish 0–9 cm	31.5		
	fish 9–13 cm	51		
	fish 13–18 cm	2.5		
Belted kingfisher: by reach				
Scenario 1	fish 0–9 cm	85	by reach	Evaluate HQs on reach-specific basis.
	blue crab	15		
Scenario 2	blue crab	15	by reach	Evaluate HQs on reach-specific basis.
	fish 0–9 cm	31.5		
	fish 9–13 cm	51		
	fish 13–18 cm	2.5		

EPC – exposure point concentration
HQ – hazard quotient
RM – river mile
USEPA – US Environmental Protection Agency

This assessment assumes that great blue heron obtain all of their food from the LPRSA. However, as discussed in Section 8.1.2.3, great blue heron have relatively large home ranges, and it is likely that they forage in areas outside of the LPRSA. Also, because of their migration patterns, great blue heron populations may not use the LPRSA year round. The use of an SUF of 1 provides conservative estimates of the potential risks. The effect on the HQs of varying the SUF is addressed in Section 8.2.4.2, as are additional uncertainties associated with exposure assumptions:

- **Exclusion of blue crab as a portion of the great blue heron diet**. While blue crab are expected to make up only a small fraction (< 5%) of the great blue heron diet, the effect of including blue crab in the great blue heron diet was evaluated.

- **Selected prey portions for each fish size class for both great blue heron and belted kingfisher**. Prey portions for the various fish size classes were assigned based on the general literature, and EPCs were based on those portions multiplied by EPCs derived for each fish size class. An evaluation was also conducted to determine the difference in risk estimates when fish size classes were not assigned prey portions, but instead were grouped into a single "all fish" EPC.

8.2.2.2 BMFs

COPEC-specific BMFs were obtained from the literature. BMFs are estimated as the average ratio of bird prey tissue and bird egg tissue concentrations. Table 8-24 presents the BMFs obtained from the published literature. BMFs were available for six species: belted kingfisher, osprey, great blue heron, bald eagle, herring gull, and brown pelican.

Table 8-24. Literature-based bird egg BMFs

COPEC	BMF	Bird Species	Fish Tissue	Location	Source
Mercury					
Mercury	1	bald eagle	various whole-body fish tissue prey items by percentage of diet from inland and coastal areas	Great Lakes, Michigan	Giesy et al. (1995)
Mercury	2.8[a] (1.9–2.9)[b]	bald eagle	common carp, peamouth chub, and sucker collected in 1991 by Watson et al. (1991), as cited in Buck (2004)	Lower Columbia River, Washington and Oregon	Buck (2004)
	2.2[a] (1.6–2.6)[b]		common carp, peamouth chub, and largescale sucker collected as part of the Bi-State Program (Tetra Tech 1993, as cited in Buck 2004)		
Geomean	**1.8**				
PCBs					
Total PCBs	5[c]	great blue heron	gizzard shad (liver excluded)	Crab Orchard National Wildlife Refuge, Illinois	Straub et al. (2007)
Total PCBs	11[b]	osprey	largescale sucker, mountain whitefish, and northern pikeminnow	Willamette River, Oregon	Henny et al. (2003)
Total PCBs	10[a] (4–73)[b,f]	great blue heron	fish tissue regurgitated or rejected cast from nests; prey items collected opportunistically; fish species not weighted as proportion in diet	Lower Columbia River, Willamette River, and Puget Sound; Oregon and Washington	Thomas and Anthony (1999)
Total PCBs	11[a,g] (8–22)[b]	osprey	channel catfish, shad, white perch, menhaden, and flounder	Delaware Bay, Maurice River, and Atlantic coast area, New Jersey	Clark et al. (2001)
PCB-118	14[h]	belted kingfisher	site-specific diet equal to 90.2% forage fish, 5.4% crayfish, and 4.4% amphibian (frog) tissue	Tittabawassee River, Michigan	Seston et al. (2012)
PCB-126	12[h]				
Total PCBs	28	bald eagle	various whole-body fish tissue prey items by percentage of diet from inland and coastal areas	Great Lakes, Michigan	Giesy et al. (1995)

Table 8-24. Literature-based bird egg BMFs

COPEC	BMF	Bird Species	Fish Tissue	Location	Source
Total PCBs	32[i]	herring gull	alewife	Lake Ontario	Braune and Norstrom (1989)
Total PCBs	32	bald eagle	unknown	Great Lakes, Michigan	Kubiak and Best (1991) as cited in Clark et al. (2001)
Total PCBs	45[a] (38–52)[b]	bald eagle	common carp, peamouth chub, and largescale sucker collected as part of the Bi-State Program (Tetra Tech 1993, as cited in Buck 2004)	Lower Columbia River, Washington and Oregon	Buck (2004)
	113[a,j] (90–155)[b]		common carp, peamouth chub, and sucker collected in 1991 by Watson et al. (1991), as cited in Buck (2004)		
Geomean	**19**				
PCDDs/PCDFs					
2,3,7,8-TCDD	4.3[h]	belted kingfisher	site-specific diet equal to 90.2% forage fish, 5.4% crayfish, and 4.4% amphibian (frog) tissue	Tittabawassee River, Michigan	Seston et al. (2012)
2,3,7,8-TCDD	7[a] (2–23)[b]	great blue heron	fish tissue regurgitated or rejected cast from nests; prey items collected opportunistically; fish species not weighted as proportion in diet	Lower Columbia River and Willamette River, Oregon and Washington	Thomas and Anthony (1999)
2,3,7,8-TCDD	10[d]	osprey	largescale sucker, mountain whitefish, and northern pikeminnow	Willamette River, Oregon	Henny et al. (2003)
2,3,7,8-TCDD[j]	15[a] (14–17)[b]	bald eagle	common carp, peamouth chub, and sucker collected in 1991 by Watson et al. (1991), as cited in Buck (2004)	Lower Columbia River, Washington and Oregon	Buck (2004)
	20[a] (15–30)[b]		common carp, peamouth chub, and largescale sucker collected as part of the Bi-State Program (Tetra Tech 1993, as cited in Buck 2004)		

Table 8-24. Literature-based bird egg BMFs

COPEC	BMF	Bird Species	Fish Tissue	Location	Source
TEQ - bird	19	bald eagle	unknown	Great Lakes, Michigan	Kubiak and Best (1991) as cited in Giesy et al. (1995)
2,3,7,8-TCDD	21[i]	herring gull	alewife	Lake Ontario	Braune and Norstrom (1989)
Geomean	**12**				
Dieldrin					
Dieldrin	7.1[h]	herring gull	alewife	Lake Ontario	Braune and Norstrom (1989)
Dieldrin	6.7[d]	osprey	largescale sucker, mountain whitefish, and northern pikeminnow	Willamette River, Oregon	Henny et al. (2003)
Geomean	**6.9**				
Total DDx					
p,p′-DDE	20[a] (9–143)[b,e]	great blue heron	fish tissue regurgitated or rejected cast from nests; prey items collected opportunistically; fish species not weighted as proportion in diet	Lower Columbia River, Willamette River, and Puget Sound; Oregon and Washington	Thomas and Anthony (1999)
p,p′-DDE	22	bald eagle	various whole-body fish tissue prey items by percentage of diet from inland and coastal areas	Great Lakes, Michigan	Giesy et al. (1995)
DDE	31	brown pelican	Menhaden fish tissue	South Carolina	Blus et al. (1977)
DDE	34[h]	herring gull	alewife	Lake Ontario	Braune and Norstrom (1989)

Table 8-24. Literature-based bird egg BMFs

COPEC	BMF	Bird Species	Fish Tissue	Location	Source
DDE	75[a] (61–78)[b]	bald eagle	common carp, peamouth chub, and sucker collected in 1991 by Watson et al. (1991), as cited in Buck (2004)	Lower Columbia River, Washington and Oregon	Buck (2004)
	141[a] (122–157)[b]		common carp, peamouth chub, and largescale sucker collected as part of the Bi-State Program (Tetra Tech 1993, as cited in Buck 2004)		
DDE	87[d]	osprey	largescale sucker, mountain whitefish, and northern pikeminnow	Willamette River, Oregon	Henny et al. (2003)
Geomean	**46**				

a Geometric mean BMF calculated from data in literature from multiple geographic areas.

b Range of BMFs reported in multiple geographic areas.

c Average BMF calculated from data in literature from two sampling years.

d Average lipid content equal to 4.3% for eggs and 5.0% for fish.

e Study reported markedly low residue concentrations detected in prey in the region with the highest reported BMF (143). BMFs in the other five regions ranged from 3 to 41.

f Study reported markedly low residue concentrations detected in prey in the region with the highest reported BMF (73). BMFs in the other five regions ranged from 4 to 13.

g Study reported a BMF of 32 in the text; however, data presented in the study result in BMFs for total PCBs of approximately 8, 8, and 22 in three Delaware Bay, Maurice River, and the Atlantic Coast, respectively.

h A BMF based on lipid-normalized egg and prey concentration was reported in the study (1.7 for 2,3,7,8-TCDD and up to 5.4 for individual PCB congeners). A BMF based on wet weight egg and prey concentrations was derived assuming 2.4% lipids in belted kingfisher prey and an egg lipid of 6% (see discussion below for belted kingfisher BMFs).

i Average lipid content equal to 7.7% for gull eggs and 2.8% for fish.

j Study reported low detection frequency of total PCBs in some fish, which may explain the high BMF of 113.

BMF – biomagnification factor
COPEC – chemical of potential ecological concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
PCB – polychlorinated biphenyl
PCDD – pentachlorodibenzo-*p*-dioxin
PCDF – pentachlorodibenzofuran
TCDD – tetrachlorodibenzo-*p*-dioxin
TEQ – toxic equivalent
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT and 4,4′-DDT)

As with any modeling, the use of BMFs to predict bird egg tissue concentrations is uncertain as a result of multiple factors, including the assumptions used in the derivation of the BMFs (i.e., the spatial area over which the average BMF is calculated), the limited validation of such empirical models by actual datasets for different species and locations, and the broad assumptions made regarding a linear relationship between populations of prey in the environment and egg concentrations for birds that feed on some portion of those prey. However, given the absence of empirical bird egg concentrations from the LPRSA, the use of modeled bird egg concentrations allows for a comparison to bird egg-specific TRVs for the evaluation of potential risks.

Selected species-specific BMFs are presented in Table 8-25 and were selected as described in the sections that follow.

Table 8-25. Selected literature-based bird egg BMFs

COPEC	Range of BMFs					
	Species-specific BMF[a]			Alternative BMFs[b]		
	BMF	Species	Source	BMF Min.	BMF Max.	BMF Geometric Mean
Great blue heron						
Mercury	1.8	bald eagle	geomean of values in Table 8-24	1	2.8	1.8
Total PCBs	5[c]	great blue heron	Straub et al. (2007)	5	113	19
PCB TEQ - bird	7[d]	great blue heron	Thomas and Anthony (1999)	4.3	21	12
PCDD/PCDF TEQ - bird						
Total TEQ - bird						
Total DDx	20	great blue heron	Thomas and Anthony (1999)	20	141	46
Dieldrin	6.9	multiple	geomean of values in Table 8-24	6.7	7.1	6.9
Belted kingfisher						
Mercury	1.8	bald eagle	geomean of values in Table 8-24	1	2.8	1.8
Total PCBs	14[c]	belted kingfisher	Seston et al. (2012)	5	113	19
PCB TEQ - bird	4.3[e]	belted kingfisher		4.3	21	12
PCDD/PCDF TEQ - bird						
Total TEQ - bird						
Total DDx	46	multiple	geomean of values in Table 8-24	20	141	46
Dieldrin	6.9	multiple	geomean of values in Table 8-24	6.7	7.1	6.9

a BMFs were derived based on process identified in Section 8.2.2.2.
b BMFs were derived based on process identified by USEPA (2015b), USEPA (2015c), and USEPA (2016g).
c Average BMF calculated from data in literature from two sampling years.
d Geometric mean BMF calculated from data in literature from multiple geographic areas.
e A BMF based on lipid-normalized egg and prey PCB 118 concentrations was reported in the literature (1.7 for 2,3,7,8-TCDD and up to 5.4 for individual PCB congeners). A BMF based on wet weight egg and prey concentrations was derived assuming 2.4% lipids in belted kingfisher prey and an egg lipid of 6% (see discussion below for belted kingfisher BMFs).

BMF – biomagnification factor
COPEC – chemical of potential ecological concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
PCB – polychlorinated biphenyl
PCDD – pentachlorodibenzo-*p*-dioxin
PCDF – pentachlorodibenzofuran
TCDD – tetrachlorodibenzo-*p*-dioxin
TEQ – toxic equivalent
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT and 4,4′-DDT)
USEPA – US Environmental Protection Agency

Due to differences among species in diet and uptake and transfer of various COPECs from diet to egg tissue, species-specific BMFs were selected for great blue heron and belted kingfisher, when BMFs were available from the literature. When species-specific

BMFs were not available, the geometric mean BMF across all species reported in the general literature was used. In addition, a range of BMFs (i.e., the minimum and maximum BMFs) and the geometric mean of the BMFs across all species were evaluated (USEPA 2015b, c, 2016g). BMFs for great blue heron and belted kingfisher are described in more detail below.

Great Blue Heron BMFs

For mercury and dieldrin, no species-specific BMFs were available for great blue heron and limited data were reported in the literature. For these two COPECs, geomeans of the available values were selected: 1.8 was selected for mercury (BMFs ranged from 1 to 2.8) and 6.9 was selected for dieldrin (BMFs ranged from 6.7 to 7.1).

There is high variability among total PCB BMFs reported for great blue heron (Table 8-25). BMFs ranged from 4 to 73 in 6 regions of the Pacific Northwest (in Washington and Oregon) (Thomas and Anthony 1999). Higher BMFs reported in this region were associated with lower prey tissue concentrations: at the three locations with the lowest prey tissue concentrations (20 to 40 µg/kg ww), the BMFs ranged from 13 to 73. At the three locations with the highest prey concentrations (94 to 627 µg/kg ww), the BMFs ranged from 4 to 5. BMFs based on the higher range of PCB concentrations in fish were consistent with the BMF of 5 based on fish and heron eggs collected from the Crab Orchard National Wildlife Refuge in Illinois. There, fish tissue concentrations ranged from 365 to 711 µg/kg ww based on the reported lipid fraction of 2.82% (Straub et al. 2007). Bioaccumulation has been found to decrease in aquatic tissues with increased sediment concentrations of organic chemicals, including PCBs (Burkhard et al. 2013). Because the LPRSA fish total PCB UCL tissue concentrations were similar to the higher range of total PCB concentrations (LPRSA fish EPCs for total PCBs were all ≥ 250 µg/kg ww), a BMF of 5 was selected for great blue heron for total PCBs.

For 2,3,7,8-TCDD, a geometric mean BMF of 7 was reported based on three regional areas in the Pacific Northwest where BMFs ranged from 2 to 23 (Thomas and Anthony 1999). This is the only great blue heron BMF available from the literature. Fish tissue concentrations of 2,3,7,8-TCDD in this study ranged from 0.23 to 0.75 ng/kg ww, several orders of magnitude less than the fish tissue concentrations from the LPRSA (LPRSA fish EPCs for great blue heron were all ≥ 34 ng/kg ww); thus, the selected BMF of 7 is uncertain for estimating uptake of PCDDs/PCDFs into LPRSA bird egg tissues.

There is high variability among total DDx BMFs reported for great blue heron (Table 8-25). BMFs ranged from 9 to 143 in 6 regions of the Pacific Northwest (in Washington and Oregon) (Thomas and Anthony 1999). Higher BMFs reported in this region were associated with lower prey tissue concentrations: at the two locations with the lowest prey tissue concentrations (3.64 and 6.0 µg/kg ww), the BMFs ranged from 41 to 143. At the four locations with the higher prey concentrations (22.2 to 71 µg/kg ww), the BMFs ranged from 3 to 24. Bioaccumulation has been found to decrease in aquatic tissues with increased sediment concentrations of organic chemicals, including DDx (Burkhard et al. 2013). Because the LPRSA fish total DDx UCL tissue concentrations were similar to the higher range of total DDx concentrations (LPRSA fish EPCs for great blue heron were all > 29 µg/kg ww), a BMF of 20 was selected for great blue heron for total DDx.

Belted Kingfisher BMFs

For mercury, total DDx, and dieldrin, no species-specific BMFs for belted kingfisher were reported in the literature. For these three COPECs, geomeans of the available values were selected: 1.8 was selected for mercury (BMFs ranged from 1 to 2.8), 46 was selected for total DDx (BMFs ranged from 20 to 141), and 6.9 was selected for dieldrin (BMFs ranged from 6.7 to 7.1).

Total PCBs and TEQ - bird BMFs for belted kingfisher were available from data collected in the Tittabawassee River floodplain in Midland, Michigan, where historical contamination from PCDDs/PCDFs has been documented (Seston et al. 2012). In this study, PCDDs/PCDFs and PCBs tended to have low bioaccumulation concentrations from prey tissue to kingfisher egg tissue. The range of average concentrations in forage fish tissue from three reaches within the study area was 12 to 25 µg/kg ww for total PCBs, and 33 to 180 ng/kg ww for PCDD/PCDF TEQs - bird. The PCDD/PCDF range in fish tissue from this study is within with the range of PCDD/PCDF concentrations in LPRSA fish tissue (LPRSA fish EPCs for belted kingfisher and total TEQs - bird ranged from 34 to 560 ng/kg ww). LPRSA total PCB concentrations are an order of magnitude greater than those reported in this study (LPRSA fish EPCs for belted kingfisher and total PCBs are all ≥ 250 µg/kg ww), so the selected BMF is uncertain in estimating uptake of total PCBs into LPRSA bird egg tissues.

Reported BMFs were based on reported lipid-normalized egg and prey concentrations; a lipid-based BMF of 1.7 was reported for 2,3,7,8-TCDD, and BMFs of 5.4 and 4.9 were reported for PCB 118 and PCB 126, respectively (Seston et al. 2012). BMFs based on wet weight egg and prey concentrations were derived assuming a prey lipid of 2.4% (based on the average lipid percent in belted kingfisher diet [85% small fish and 15% crab using LPRSA data]) and an egg lipid of 6% (based on the average lipid percent reported for other bird species,[124] including herring gull [7.7%], osprey [4.3%], and great blue heron [5.7%]) (Braune and Norstrom 1989; Henny et al. 2003; Straub et al. 2007). The resulting wet weight-based BMFs were 4.3 for 2,3,7,8-TCDD, 14 for PCB 118, and 12 for PCB 126. These BMFs (4.3 for 2,3,7,8-TCDD and 14 for PCBs [the higher of the values for the two PCB congeners]) were selected for modeling belted kingfisher egg tissue concentrations in the LPRSA.

8.2.2.3 Calculated egg concentrations

The bird egg tissue concentrations calculated using Equation 8-5 are presented in Table 8-26. Uncertainties associated with the use of non-detects in calculations of total PCBs and TEQs- bird are discussed in Section 8.2.4.2.

[124] A lipid value for belted kingfisher eggs was not identified from the literature.

Table 8-26. Modeled LPRSA piscivorous bird egg concentrations

COPEC		Unit (ww)	Egg Concentration							
			Belted Kingfisher				Great Blue Heron			
			Range of BMFs				Range of BMFs			
			Species-specific BMF[a]	Alternative BMFs[b]			Species-specific BMF[a]	Alternative BMFs[b]		
				BMF Min.	BMF Max.	BMF Geometric Mean		BMF Min.	BMF Max.	BMF Geometric Mean
Mercury										
1	RM ≥ 6	µg/kg	119	66	198	119	na	na	na	na
	site wide		120	67	201	120	106	59	177	106
	by reach		96–152	53–84	160–253	88–152	77–149	43–83	129–213	77–149
2	RM ≥ 6		130	72	217	130	na	na	na	na
	site wide		133	74	222	133	263	146	439	263
	by reach		93–153	52–85	155–255	93–153	254–417	141–232	423–695	254–417
Total PCBs										
1	RM ≥ 6	µg/kg	9,898	3,535	79,891	13,433	na	na	na	na
	site wide		9,779	3,493	78,931	13,272	2,800	2,800	63,280	10,640
	by reach		3,584–12,285	1,280–4,388	28,928–99,158	4,864–16,673	1,250–4,650	1,250–4,650	28,250–105,090	4,750–17,670
2	RM ≥ 6		8,647	3,088	69,794	11,735	na	na	na	na
	site wide		8,103	2,894	65,404	10,997	9,461	9,461	213,819	35,952
	by reach		4,442–9,881	2,086–3,529	35,849–79,750	6,028–13,409	4,388–16,319	4,388–16,319	99,169–368,798	16,674–62,010
PCB TEQ - bird										
1	RM ≥ 6	ng/kg	298	298	1,456	832	na	na	na	na
	site wide		291	291	1,421	812	364	223.6	1092	624
	by reach		143–364	116–364	565–1,779	323–1,016	140–616	86–378.4	420–1,533	240–1056
2	RM ≥ 6		328	328	1,602	916	na	na	na	na
	site wide		277	277	1,353	773	998	613	2995	1712
	by reach		140–325	140–325	686–1,801	392–1,029	628–1,810	207–1,112	1012–5,431	578–3,103

Table 8-26. Modeled LPRSA piscivorous bird egg concentrations

COPEC		Unit (ww)	Egg Concentration							
			Belted Kingfisher				Great Blue Heron			
			Range of BMFs				Range of BMFs			
			Species-specific BMF[a]	Alternative BMFs[b]			Species-specific BMF[a]	Alternative BMFs[b]		
				BMF Min.	BMF Max.	BMF Geometric Mean		BMF Min.	BMF Max.	BMF Geometric Mean
PCDD/PCDF TEQ - bird										
1	RM ≥ 6	ng/kg	331	331	1,616	923	na	na	na	na
	site wide		309	309	1,509	862	336	206.4	1008	576
	by reach		95–442	95–442	511–2,159	265–1,234	105–770	65–361	315–1764	180–1,320
2	RM ≥ 6		272	272	1,327	758	na	na	na	na
	site wide		239	239	1,169	668	1036	636	3107	1,776
	by reach		105–358	105–358	511–1750	292–1000	290–1867	178–1147	871–5600	498–3200
Total TEQ - bird										
1	RM ≥ 6	ng/kg	621	621	3,035	1,734	na	na	na	na
	site wide		585	585	2,856	1,632	672	412.8	2016	1,152
	by reach		121–714	212–714	1,034–3,486	591–1,968	245–1,190	172–731	735–3,570	420–2040
2	RM ≥ 6		579	579	2,829	1,616	na	na	na	na
	site wide		497	497	2,425	1,386	2,012	1,236	6,036	3,449
	by reach		254–659	249–659	1,242–3,220	696–1,840	881–3,718	541–2,284	2642–11,155	1,510–6,374
Total DDx										
1	RM ≥ 6	µg/kg	3,558	1,547	10,906	3,558	na	na	na	na
	site wide		3,636	1,581	11,146	3,636	1,240	1,240	8,742	2,852
	by reach		1,507–4,455	655–1,937	4,618–13,656	1,507–4,455	580–1,720	580–1720	4,089–14,100	1,334–4,600
2	RM ≥ 6		3,287	1429	10,075	3,287	na	na	na	na
	site wide		3,093	1345	9,480	3,093	3,999	3,999	28,192	9,197
	by reach		1,707–3,769	742–1,639	5,231–11,554	1,707–3,769	2,087–7,515	2,087–7,515	14,714–52,979	4,800–17,284

Table 8-26. Modeled LPRSA piscivorous bird egg concentrations

COPEC		Unit (ww)	Egg Concentration							
			Belted Kingfisher				Great Blue Heron			
			Range of BMFs				Range of BMFs			
			Species-specific BMF[a]	Alternative BMFs[b]			Species-specific BMF[a]	Alternative BMFs[b]		
				BMF Min.	BMF Max.	BMF Geometric Mean		BMF Min.	BMF Max.	BMF Geometric Mean
Dieldrin										
1	RM ≥ 6	µg/kg	118	115	122	118	na	na	na	na
	site wide		101	98	104	101	76	74	78	76
	by reach		26–172	25–167	27–177	26–172	24–193	23–188	25–199	24–193
2	RM ≥ 6		107	104	110	107	na	na	na	na
	site wide		83	80	85	83	221	214	227	221
	by reach		26–131	25–127	27–134	26–131	91–416	89–404	94–428	91–416

[a] BMFs were derived based on process identified in Section 8.2.2.2.

[b] BMFs were derived based on process identified by USEPA (2015b), USEPA (2015c), and USEPA (2016g); egg concentrations calculated based on these BMFs.

BMF – biomagnification factor
COPEC – chemical of potential ecological concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
LPRSA – Lower Passaic River Study Area
na – not applicable
PCB – polychlorinated biphenyl
PCDD – polychlorinated dibenzo-*p*-dioxin
PCDF – polychlorinated dibenzofuran
RM – river mile
TEQ – toxic equivalent
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT and 4,4′-DDT)
ww – wet weight
USEPA – US Environmental Protection Agency

8.2.3 Effects

Bird egg tissue TRVs for evaluating potential effects on early life stages of piscivorous birds were developed in the same manner as dietary TRVs (Section 8.1.3.1). A range of TRVs was evaluated. The selection was based on a comprehensive review of the primary literature and an assessment of acceptability. TRVs were also based on previous documents developed by USEPA Region 2 for the LPRSA. Selected TRVs are consistent with the comments, responses, and directives received from USEPA on June 30, 2017 (USEPA 2017b), September 18, 2017 (USEPA 2017e), July 10, 2018 (USEPA 2018), January 2, 2019 (CDM 2019), and March 5, 2019 (USEPA 2019); during face-to-face meetings or conference calls on July 24, September 27, October 3, November 6, 2017, July 24, 2018, and August 16, 2018; and via additional deliverables and communications between the Cooperating Parties Group (CPG) and USEPA from August through December 2017, July through September 2018, and January through June 2019.

8.2.3.1 Methods for selecting TRVs

Two sets of bird egg tissue TRVs were used for the derivation of HQs in this BERA. One set of TRVs was based on previous documents developed by USEPA Region 2 for the LPRSA:

- USEPA's revised draft of the LPR restoration project FFS (Louis Berger et al. 2014)
- USEPA's first draft of the LPRSA FFS (Malcolm Pirnie 2007b)

The second set of TRVs was selected by first conducting a literature search for relevant toxicological studies. These studies were then evaluated for acceptability of use. For those studies considered acceptable, as described in Appendix E, NOAEL and LOAEL daily doses were derived. TRVs were then selected for each COPEC-bird species pair based on an evaluation of all the acceptable NOAEL and LOAEL TRVs. Details regarding the literature search and acceptability of the studies are presented in Appendix E. The TRV selection process and general uncertainties regarding the use of TRVs to estimate risk are the same as for bird dietary TRVs, as described in Section 8.1.3.1. COPEC-specific uncertainties associated with bird egg tissue TRVs are discussed in the following section (Section 8.2.3.2).

8.2.3.2 Selected TRVs for bird eggs

The bird egg TRVs used in this BERA are summarized in Table 8-27.

Table 8-27. Bird egg tissue TRVs

COPEC	Units (ww)	Range of TRVs[a]								
		TRV-A[b]				TRV-B[c]				
		NOAEL	LOAEL	Endpoint	Source	NOAEL	LOAEL	Endpoint	Source	Document
Metals										
Methylmercury/ mercury	µg/kg	180[d]	1,800	reproduction (mallard)	geomean of LOAELs reported for 4 studies	no value[e]	no value[e]	na	na	na
PCDDs/PCDFs										
PCDD/PCDF TEQ - bird Total TEQ - bird	ng/kg	25[d]	250	reproduction (5 species)	SSD-derived 5th percentile value	59	150	reproduction (various species)	SSD-derived 5th percentile value USEPA (2003c)	revised FFS (Louis Berger et al. 2014)
PCBs										
Total PCBs	µg/kg	1,600[d]	16,000	reproduction (ringed turtle dove)	Peakall et al. (1972); Peakall and Peakall (1973)	700	1,300	chicken (hatchability)	Chapman (2003)	revised FFS (Louis Berger et al. 2014)
PCB TEQ - bird	ng/kg	25[d]	250	reproduction (5 species)	SSD-derived 5th percentile value	59	150	reproduction (various species)	SSD-derived 5th percentile value USEPA (2003c)	revised FFS (Louis Berger et al. 2014)
Pesticides										
Dieldrin	µg/kg	300[d]	3,000	reproduction (pheasant)	Genelly and Rudd (1956)	200	8,100	reproduction (barn own)	Mendenhall et al. (1983)	revised FFS (Louis Berger et al. 2014)
Total DDx	µg/kg	410[d]	4,100	reproduction (7 species)	SSD-derived 5th percentile value	500	3,000	reproduction (brown pelican)	Blus (1984)	revised FFS (Louis Berger et al. 2014)

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

b TRVs were derived from the primary literature review based on process identified in Section 8.2.3.1.

[c] TRVs were based on USEPA's revised draft of the LPR restoration project FFS (Louis Berger et al. 2014) or first draft of the LPR restoration project FFS (Malcolm Pirnie 2007b).

[d] NOAEL extrapolated from LOAEL using an uncertainty factor of 10.

[e] No TRVs were selected by USEPA in the revised FFS (Louis Berger et al. 2014) or draft FFS (Malcolm Pirnie 2007b).

BERA – baseline ecological risk assessment
COPEC – chemical of potential ecological concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
FFS – focused feasibility study
LOAEL – lowest-observed-adverse-effect level
LPR – Lower Passaic River
LPRSA – Lower Passaic River study Area
na – not available
NJDEP – New Jersey Department of Environmental Protection
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
PCDD – polychlorinated dibenzo-*p*-dioxin
PCDF – polychlorinated dibenzofuran
TEQ – toxic equivalent
SSD – species sensitivity distribution
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT and 4,4′-DDT)
TRV – toxicity reference value
USEPA – US Environmental Protection Agency
ww – wet weight

Methylmercury/Mercury

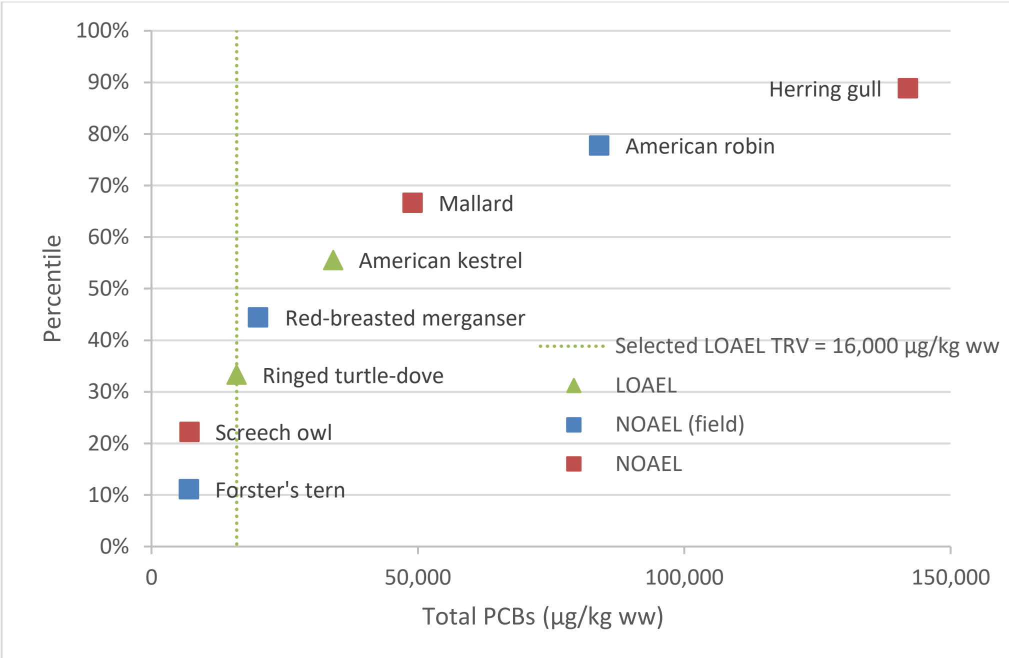
A total of six acceptable studies were found that evaluated mercury bird egg tissue residues. LOAELs were reported for only one species (mallards) (Heinz 1979; Heinz and Hoffman 2003; Heinz 1976, 1974), so data were insufficient for the development of an SSD. No TRVs were available for the piscivorous bird species selected for evaluation (i.e., belted kingfisher and great blue heron). A geometric mean of 1,800 µg/kg ww was derived based on the five reported values; the reproductive endpoints for mallards for these five LOAELs were embryo development, offspring survival, hatchability, avoidance response behavior, and egg/young production. The geometric mean LOAEL was selected as the LOAEL TRV. There was no NOAEL from this study, so one was extrapolated using an uncertainty factor of 10; the selected NOAEL was 180 µg/kg ww. There is uncertainty associated with the selected TRVs due to the limited toxicity dataset. There is also uncertainty associated with the use of an extrapolation factor to derive a NOAEL.

No TRVs were available for mercury in the draft FFS (Malcolm Pirnie 2007b) or revised FFS (Louis Berger et al. 2014), so no additional TRVs were used for mercury.

Total PCBs

A total of 12 acceptable studies were found that evaluated total PCB bird egg tissue residues and effect thresholds. LOAELs from these studies were reported for only two species (ringed turtle doves and American kestrels) (Peakall and Peakall 1973; Peakall et al. 1972; Fernie et al. 2000; Fernie et al. 2001), so data were insufficient for the development of an SSD. The lowest LOAEL of 16,000 µg/kg ww was reported in ringed turtle dove egg tissue following two generations of exposure, resulting in reduced hatchability and embryo survival (Peakall and Peakall 1973; Peakall et al. 1972). This LOAEL was selected as the LOAEL TRV. There was no NOAEL from this study, so one was extrapolated using an uncertainty factor of 10; the selected NOAEL was 1,600 µg/kg ww. There is uncertainty associated with the use of an extrapolation factor to derive a NOAEL.

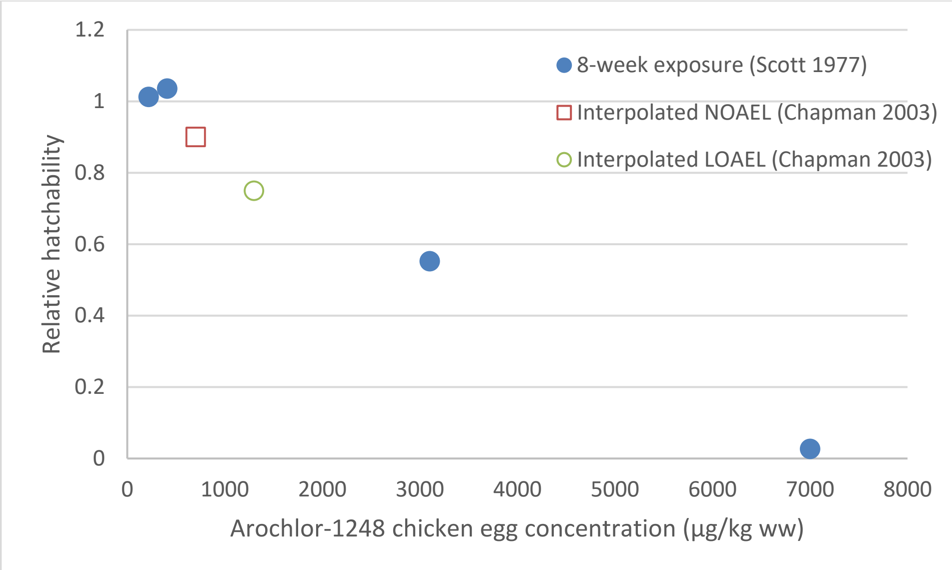
The selected LOAEL was based on a limited dataset; only two LOAELs based on controlled studies were available from the literature. The LOAEL not selected was 34.1 mg/kg ww, a dose at which the reproductive success of American kestrels had been found to be reduced (Fernie et al. 2000; Fernie et al. 2001). A number of NOAELs for bird eggs and total PCBs that were greater than the selected LOAEL were available for several species (Figure 8-4); however, no NOAELs were available for the piscivorous bird species selected for evaluation (i.e., belted kingfisher and great blue heron).



Note: All TRVs based on reproductive success (e.g., hatchability, embryo survival, fledgling survival).

Figure 8-4.Bird egg tissue total PCB toxicity data

NOAEL and LOAEL TRVs of 700 and 1,300 µg/kg ww, respectively, were also selected for total PCBs (Louis Berger et al. 2014). These TRVs were based on an interpolated no-effect value (based on a 10% decrease relative to control) and a low-effect value (based on a 25% decrease relative to control) using chicken hatchability data as described by Chapman (2003). The interpolated value from Chapman (2003) was based on data reported by Scott (1977), wherein an empirical LOAEL and NOAEL of 410 and 3,100 µg/kg ww, respectively, were determined following eight weeks of exposure to Aroclor 1248; at the LOAEL, chicken hatchability was significantly reduced (55% of control). These data are presented in Figure 8-5.



Source: Chapman (2003)

Figure 8-5.Interpolated bird egg tissue total PCB data

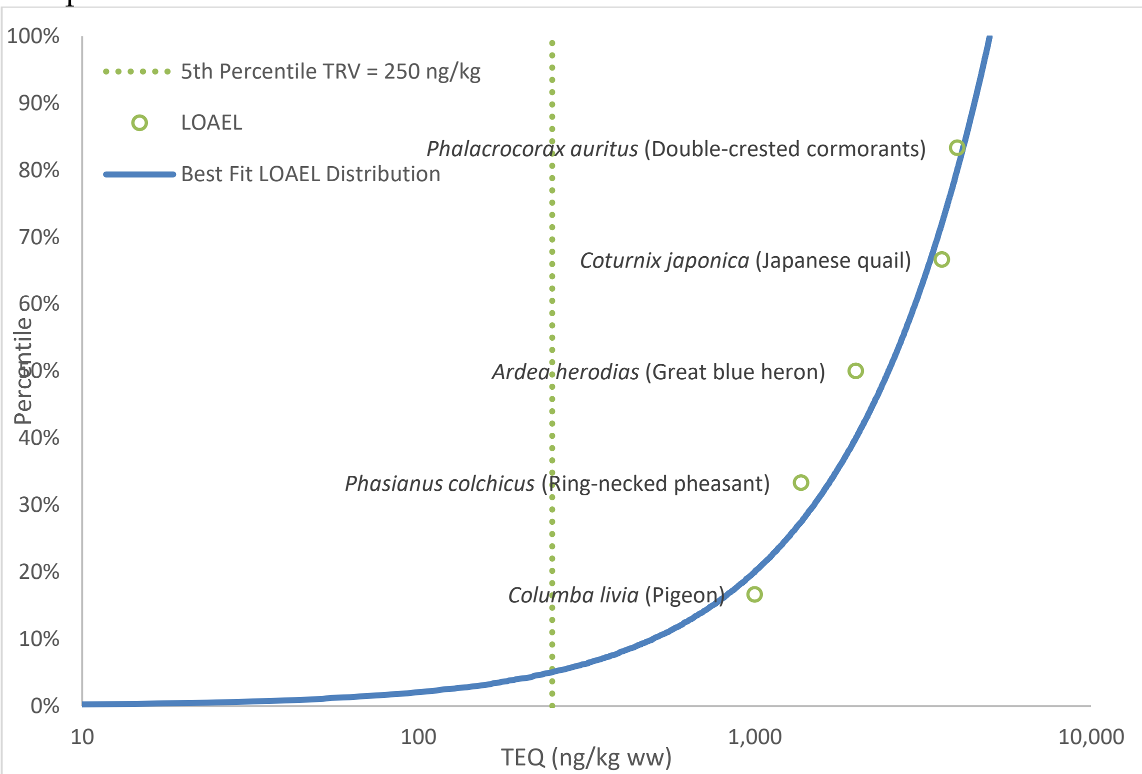
The interpolated values are associated with uncertainty, given the range of data over which the Chapman (2003)-interpolated values were determined based on the empirical data; the empirical LOAEL (based on hatchability reduced to 55% of control) from Scott (1977) was 3,100 µg/kg ww, whereas the interpolated LOAEL (based on a low-effect threshold of 25% relative to control) was 1,300 µg/kg ww.

TRVs based on domestic reproductive endpoints are uncertain because domesticated species are indeterminate layers with altered egg-laying rates compared to wild bird species. It is not known how an effect threshold of 25% reduction in egg hatchability of chickens is predictive of potential population-level effects in wild birds.

PCDDs/PCDFs - Bird

Five acceptable toxicity studies were reviewed that evaluated bird egg tissue 2,3,7,8-TCDD and effect levels following exposure to 2,3,7,8-TCDD from injection (Janz and Bellward 1996; Powell et al. 1997; Powell et al. 1998; Nosek et al. 1992; Cohen-Barnhouse et al. 2011). LOAELs based on embryo survival and hatchability were reported for five bird species (i.e., double-crested cormorant, great blue heron, Japanese quail, pigeon, and ring-necked [or common] pheasant) and ranged from 1,000 to 40,000 ng/kg ww. An SSD was developed to derive a TRV (Figure 8-6), and the 5th percentile determined from the SSD was 250 ng/kg ww; this LOAEL TRV was selected. The SSD-derived LOAEL was less than the lowest measured LOAEL reported from the literature: a dose of 1,000 mg/kg bw/day associated with embryo mortality in ringed necked pheasants (Nosek et al. 1992) and hatchability in pigeons (Janz and Bellward 1996) following a single egg injection of 2,3,7,8-TCDD (Appendix E). Thus, the SSD-derived LOAEL represents a conservatively extrapolated value that is less than those empirically measured in the reviewed toxicity studies. The

NOAEL TRV (25 ng/kg ww) was extrapolated from the LOAEL TRV using an uncertainty factor of 10. There is uncertainty associated with the use of an extrapolation factor to derive a NOAEL.



Note: All TRVs based on reproductive success (hatchability or embryo survival).

Figure 8-6.Bird egg tissue 2,3,7,8–TCDD SSD toxicity data

Chicken toxicity data were excluded in the development of the SSD, given the greater sensitivity of chickens to PCDDs/PCDFs relative to wild bird species. LOAELs based on chicken reproduction were reported in several studies, ranging from 10 to 320 ng/kg ww with a geometric mean of 130 ng/kg ww. The 5th percentile SSD would not have changed significantly had the chicken data been included (i.e., decreased from 250 to 240 ng/kg ww).

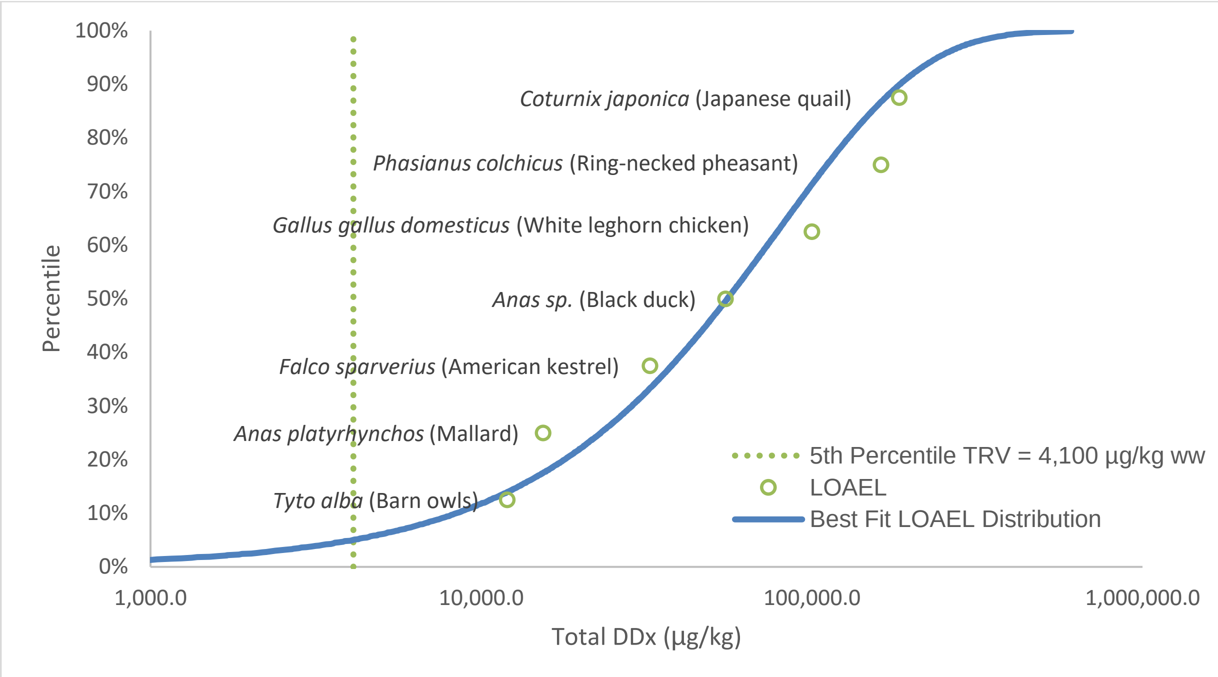
NOAEL and LOAEL values of 59 and 150 ng/kg ww, respectively, were also selected for TEQ - bird (Louis Berger et al. 2014) based on an SSD 5th percentile from USEPA guidance (2003c). These values included chicken reproduction. The 5th percentile SSD from USEPA guidance (2003c) was slightly less than that derived using the studies shown in Figure 8-6 and including chicken toxicity data, as described above (i.e., a 5th percentile LOAEL SSD of 240 ng/kg ww). Domesticated species have unnaturally high egg-laying rates and toxicological and reproductive sensitivities that are very different from those of wild bird species. Comparing toxic threshold effects on reproductive endpoints for domesticated species with those for non-domesticated species is uncertain because of differences in reproductive physiology.

As discussed in Section 8.1.3.2, recent studies have found that avian sensitivity to the toxic effects of dioxin-like compounds may vary up to 1,000-fold among bird species, and is associated with differences in the structural characteristics of the Ah receptor (Farmahin et al. 2013; Cohen-Barnhouse et al. 2011; Head et al. 2008). Genetic differences in the ligand-binding domain of the Ah receptor have been correlated to differences in avian sensitivities, such as embryo survival (Head et al. 2008). Using the amino acid sequences of the ligand-binding domain of the Ah receptor in individual bird species, a number of birds have been grouped into three classifications of sensitivity to dioxin-like compounds: 1) high sensitivity, 2) moderate sensitivity, and 3) low sensitivity (Farmahin et al. 2013). Chickens are in the high-sensitivity group and likely over-predict the PCDD/PCDF sensitivity of LPRSA species, such as great blue heron, which are in the low-sensitivity group. If a great blue heron-specific LOAEL is used, the results suggest that this species is much less sensitive to PCBs than are chickens and other birds (pigeons and pheasants); hatchability is reduced by 18% relative to control in great blue heron, with egg concentrations of 2,000 ng/kg ww (Janz and Bellward 1996) (compared with the SSD-derived value of 250 ng/kg ww). Therefore, the selected LOAEL may over-predict the toxicity of PCDDs/PCDFs to belted kingfisher and great blue heron. An HQ based on the use of a species-specific toxicity value of 2,000 ng/kg ww was calculated as part of the uncertainty evaluation (Section 8.2.4.2).

Total DDx

Eight acceptable toxicity studies were reviewed that evaluated bird egg tissue DDx or DDx metabolites and effect levels (Wiemeyer and Porter 1970; Mendenhall et al. 1983; Longcore et al. 1971; Longcore and Samson 1973; Bryan et al. 1989; Haegele and Hudson 1974; Dunachie and Fletcher 1969; Genelly and Rudd 1956). LOAELs based on eggshell thickness, embryo and offspring survival, and hatchability were reported for seven bird species (i.e., American kestrel, barn owl, American black duck, Japanese quail, mallard, ring-necked [or common] pheasant, and white leghorn chicken), with LOAELs ranging from 12,000 to 658,000 µg/kg ww. An SSD was developed to derive a TRV (Figure 8-7). The 5th percentile determined from the SSD was 4,100 µg/kg ww; this TRV was selected.

The 5th percentile LOAEL of 4,100 µg/kg ww is similar to LOAELs for eggshell thinning and reproductive success reported in various field studies, wherein field-based LOAELs ranged from 3,500 to 84,500 µg/kg ww (Appendix E). However, the SSD-derived LOAEL is less than the lowest measured LOAEL reported from the literature: an egg residue of 12,000 µg/kg ww associated with nestling mortality in barn owls following dietary exposure to DDE (Mendenhall et al. 1983) (Appendix E). Thus, the SSD-derived LOAEL represents a conservatively extrapolated value that is less than those empirically measured in the reviewed toxicity studies. The NOAEL TRV (410 µg/kg ww) was extrapolated from the LOAEL TRV using an uncertainty factor of 10. There is uncertainty associated with the use of an extrapolation factor to derive a NOAEL.



Note: All TRVs based on reproductive success (eggshell thinning, hatchability, embryo survival, or offspring survival).

Figure 8-7.Bird egg tissue total DDx SSD toxicity data

NOAEL and LOAEL TRVs of 500 and 3,000 µg/kg ww, respectively, were selected for total DDx (Louis Berger et al. 2014) based on data reported for DDx egg residues for brown pelican (Blus 1984). Blus (1984) established a critical value of 3,000 µg/kg for black pelicans "if it prevailed through most of the breeding population for several years." However, the data supporting this value were not consistent; nesting success was 30% at values ranging from non-detected concentrations to 1,000 µg/kg,[125] 50% at 1,000 to 2,000 µg/kg, and 30 to 50% at 3,000 µg/kg. Reproductive failure was noted at a concentration of 3,700 µg/kg.

The use of field-collected egg data created uncertainty in establishing a LOAEL, given the other factors in the field that could potentially influence to reproductive success (e.g., other contaminants and non-chemical stressors). Furthermore, Blus (1984) provided data for black-crowned night-heron, a species present in the LPRSA and directly relevant to great blue heron. Blus (1984) established a critical tissue residue (CTR) value (based on nesting success) of 12,000 µg/kg for black-crowned night

[125] Non-detected concentrations ranged up to 100 µg/kg for DDx (Blus 1984). The USEPA-recommended NOAEL of 500 µg/kg was used to represent the range of concentrations from the non-detected values to 1,000 µg/kg.

heron, indicating the greater sensitivity of brown pelicans. Also reported by Blus (1984) were critical values based on eggshell thinning (an endpoint that has been linked directly with DDx), and field-measured DDx values for the brown pelican and black-crowned night-heron. A range of 5,000 to 8,000 µg/kg was reported for brown pelican (associated with 18 to 20% eggshell thinning), and a range of 36,000 to 54,000 µg/kg was reported for black-crowned night-heron (associated with 18 to 20% eggshell thinning). The 18 to 20% range of eggshell thinning is associated with the critical level at which populations may be affected. Thus, there is uncertainty associated with these selected TRVs.

Dieldrin

A LOAEL of 3,000 µg/kg ww for dieldrin was selected based on pheasant reproduction (Genelly and Rudd 1956). A total of three acceptable studies were found that evaluated dieldrin tissue residues in bird eggs. There was no NOAEL from these studies, so a NOAEL of 300 µg/kg ww was extrapolated using an uncertainty factor of 10. There is uncertainty associated with the use of an extrapolation factor to derive a NOAEL. There is uncertainty associated with the TRVs, as two LOAELs based on controlled studies (3,000 and 33,600 µg/kg ww) were available from the literature, both for pheasant reproduction. No TRVs were available for the piscivorous bird species selected for evaluation (i.e., belted kingfisher and great blue heron). There is also uncertainty associated with the use of an extrapolation factor to derive a NOAEL.

NOAEL and LOAEL TRVs of 200 and 8,100 µg/kg ww, respectively, were also selected for dieldrin (Louis Berger et al. 2014) based on values reported by Mendenhall et al. (1983). Eggs with concentrations of 8,100 µg/kg ww were reported to have eggshell thickness reduced by 5.5%; however, Mendenhall et al. (1983) also reported that no reduction in breeding success was noted in that exposure group. Thus, there is uncertainty associated with using this LOAEL to predict the potential for adverse effects in wild populations. The NOAEL of 200 µg/kg ww is based on the control concentration in the study; the use of a control for a no-effect threshold is uncertain.

8.2.4 Risk characterization

This section presents the bird egg HQs (Section 8.2.4.1), as well as uncertainties associated with the HQ calculations (Section 8.2.4.2). In addition to the original HQ calculations, this section presents alternate HQs calculated based on the identified uncertainties. These alternates were calculated to determine if any of the uncertainties could result in risk conclusions that were different from those determined by the original HQs. For COPECs with HQs ≥ 1.0 when compared with LOAEL TRVs, a comparison of background data to site data is also presented, consistent with USEPA guidance (USEPA 2002c).

8.2.4.1 Bird egg HQs

HQs were calculated for the seven bird egg COPECs (Tables 8-28 and 8-29). HQs were < 1.0 for methylmercury/mercury and for dieldrin. HQs for the remaining COPECs had HQs ≥ 1 (i.e., total PCBs, PCB TEQ - bird, PCDD/PCDF TEQ - bird, total TEQ - bird, and total DDx).

Table 8-28. Bird egg tissue LOAEL HQs

			Range of LOAEL HQs for Great Blue Heron[a]				Range of LOAEL HQs for Belted Kingfisher[a]			
			HQ Based on TRV-A[b]	HQ Based on TRV-B[c]			HQ Based on TRV-A[b]	HQ Based on TRV-B[c]		
				Alternative BMF[e]				Alternative BMF[e]		
COPEC	Diet Scenario	Area	Species-specific BMF[d]	BMF Min.	BMF Max.	BMF Geomean	Species-specific BMF[d]	BMF Min.	BMF Max.	BMF Geomean
Total PCBs	1[f]	site wide	0.18	**2.2**	**49**	**8.2**	0.62	**2.7**	**61**	**10**
		RM ≥ 6	na	na	na	na	0.61	**2.7**	**61**	**10**
		by reach	0.078–0.29	**1.0–3.6**	**22–81**	**3.7–14**	0.22–0.77	**1.0–3.4**	**22–76**	**3.7–13**
	2[g]	site wide	0.59	**7.3**	**164**	**28**	0.51	**2.2**	**50**	**8.5**
		RM ≥ 6	na	na	na	na	0.54	**2.4**	**54**	**9.0**
		by reach	0.27–**1.0**	**3.4–13**	**76–284**	**13–48**	0.28–0.62	**1.2–2.7**	**28–61**	**4.6–10**
PCB TEQ - bird	1[f]	site wide	**1.5**	**1.5**	**7.3**	**4.2**	**1.2**	**1.9**	**9.5**	**5.4**
		RM ≥ 6	na	na	na	na	**1.2**	**2.0**	**9.7**	**5.5**
		by reach	0.56–**2.5**	0.57–**2.5**	**3.2–12**	**1.6–7.0**	0.46–**1.5**	0.77–**2.4**	**3.8–11**	**2.2–6.8**
	2[g]	site wide	**4.0**	**4.1**	**20**	**11**	**1.1**	**1.8**	**9.0**	**5.2**
		RM ≥ 6	na	na	na	na	**1.3**	**2.2**	**11**	**6.1**
		by reach	**1.3–7.2**	**1.4–7.4**	**6.7–36**	**3.9–21**	0.56–**1.5**	0.94–**2.5**	**4.6–12**	**2.6–6.9**
PCDD/ PCDF TEQ - bird	1[f]	site wide	**1.3**	**1.4**	**6.7**	**3.8**	**1.2**	**2.1**	**10**	**5.7**
		RM ≥ 6	na	na	na	na	**1.3**	**2.2**	**11**	**6.2**
		by reach	0.42–**3.1**	0.43–**3.2**	**2.1–15**	**1.2–8.8**	0.38–**1.8**	0.63–**2.9**	**3.1–14**	**1.8–8.2**
	2[g]	site wide	**4.1**	**4.2**	**21**	**12**	0.96	**1.6**	**7.8**	**4.5**
		RM ≥ 6	na	na	na	na	**1.1**	**1.8**	**8.8**	**5.1**
		by reach	**1.2–7.5**	**1.2–7.6**	**5.8–37**	**3.3–21**	0.42–**1.4**	0.70–**2.4**	**3.4–12**	**1.9–6.7**

Table 8-28. Bird egg tissue LOAEL HQs

COPEC	Diet Scenario	Area	Range of LOAEL HQs for Great Blue Heron[a]				Range of LOAEL HQs for Belted Kingfisher[a]			
			HQ Based on TRV-A[b]	HQ Based on TRV-B[c]			HQ Based on TRV-A[b]	HQ Based on TRV-B[c]		
				Alternative BMF[e]				Alternative BMF[e]		
			Species-specific BMF[d]	BMF Min.	BMF Max.	BMF Geomean	Species-specific BMF[d]	BMF Min.	BMF Max.	BMF Geomean
Total TEQ - bird	1[f]	site wide	**2.7**	**2.8**	**13**	**7.7**	**2.3**	**3.9**	**19**	**11**
		RM ≥ 6	na	na	na	na	**2.5**	**4.1**	**20**	**12**
		by reach	**1.0–4.8**	**1.0–4.9**	**4.9–24**	**2.8–14**	0.85–**2.9**	**1.4–4.8**	**6.9–23**	**3.9–13**
	2[g]	site wide	**8.0**	**8.2**	**40**	**23**	**2.0**	**3.3**	**16**	**9.2**
		RM ≥ 6	na	na	na	na	**2.3**	**3.9**	**19**	**11**
		by reach	**3.5–15**	**3.6–15**	**18–74**	**10–42**	**1.0–2.6**	**1.7–4.4**	**8.1–21**	**4.6–12**
Total DDx	1[f]	site wide	0.30	0.41	**2.9**	**0.95**	0.89	0.53	**3.7**	**1.2**
		RM ≥ 6	na	na	na	na	0.87	0.52	**3.6**	**1.2**
		by reach	0.14–0.49	0.19–0.67	**1.4–4.7**	0.44–**1.5**	0.37–**1.1**	0.22–0.65	**1.5–4.6**	0.50–**1.5**
	2[g]	site wide	0.98	**1.3**	**9.4**	**3.1**	0.75	0.45	**3.2**	**1.0**
		RM ≥ 6	na	na	na	na	0.80	0.48	**3.4**	**1.1**
		by reach	0.51–**1.8**	0.70–**2.5**	**4.9–18**	**1.6–5.8**	0.42–0.92	0.25–0.55	**1.7–3.9**	0.57–**1.3**

Bold identifies HQs ≥ 1.0.

Shaded cells identify HQs ≥ 1.0 based on LOAEL TRVs.

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

b TRVs were derived from the primary literature based on the process identified in Section 8.2.3.1.

c TRVs were based on USEPA's revised draft of the LPR restoration project FFS (Louis Berger et al. 2014) or first draft of the LPR restoration project FFS (Malcolm Pirnie 2007b).

d BMFs were derived based on the process identified in Section 8.2.2.2.

[e] BMFs were based on the process identified by USEPA (2015b), USEPA (2015c), and USEPA (2016g).

[f] Diet Scenario 1 includes 100% 0–13-cm fish for great blue heron and 85% 0–9-cm fish and 15% blue crab for belted kingfisher.

[g] Diet Scenario 2 for great blue heron includes 17% 0–13-cm fish, 29% 13–18-cm fish, 40% 18–30-cm fish, and 14% > 30-cm fish. Diet Scenario 2 for belted kingfisher includes 15% blue crab, 31.5% 0–9-cm fish, 51% 9–13-cm fish, and 2.5% 13–18-cm fish.

BERA – baseline ecological risk assessment
BMF – biomagnification factor
COPEC – chemical of potential ecological concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
FFS – focused feasibility study
HQ – hazard quotient
LOAEL – lowest–observed–adverse–effect level
LPR – Lower Passaic River
LPRSA – Lower Passaic River study Area
na – not applicable (no TRV available)
NJDEP – New Jersey Department of Environmental Protection
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
PCDD – polychlorinated dibenzo-*p*-dioxin
PCDF – polychlorinated dibenzofuran
RM – river mile
TEQ – toxic equivalent
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT and 4,4′-DDT)
TRV – toxicity reference value
USEPA – US Environmental Protection Agency

Table 8-29. Bird egg tissue NOAEL HQs

COPEC	Diet Scenario	Area	Range of NOAEL HQs for Great Blue Heron[a]				Range of NOAEL HQs for Belted Kingfisher[a]			
			HQ Based on TRV-A[b]	HQ Based on TRV-B[c]			HQ Based on TRV-A[b]	HQ Based on TRV-B[c]		
				Alternative BMF[e]				Alternative BMF[e]		
			Species-specific BMF[d]	BMF Min.	BMF Max.	BMF Geomean	Species-specific BMF[d]	BMF Min.	BMF Max.	BMF Geomean
Total PCBs	1[f]	site wide	**1.8**	**4.0**	**90**	**15**	**6.1**	**5.0**	**110**	**19**
		RM ≥ 6	na	na	na	na	**6.2**	**5.1**	**110**	**19**
		by reach	0.78–**2.9**	**1.8–6.6**	**40–150**	**6.8–25**	**2.2–7.7**	**1.8–6.3**	**41–140**	**6.9–24**
	2[g]	site wide	**5.9**	**14**	**310**	**51**	**5.1**	**4.1**	**93**	**16**
		RM ≥ 6	na	na	na	na	**5.4**	**4.4**	**100**	**17**
		by reach	**2.7–10**	**6.3–23**	**140–530**	**24–89**	**2.8–6.2**	**2.3–5.0**	**51–110**	**8.6–19**

Table 8-29. Bird egg tissue NOAEL HQs

COPEC	Diet Scenario	Area	Range of NOAEL HQs for Great Blue Heron[a]				Range of NOAEL HQs for Belted Kingfisher[a]			
			HQ Based on TRV-A[b]	HQ Based on TRV-B[c]			HQ Based on TRV-A[b]	HQ Based on TRV-B[c]		
				Alternative BMF[e]				Alternative BMF[e]		
			Species-specific BMF[d]	BMF Min.	BMF Max.	BMF Geomean	Species-specific BMF[d]	BMF Min.	BMF Max.	BMF Geomean
PCB TEQ - bird	1[f]	site wide	**15**	**3.8**	**19**	**11**	**12**	**4.9**	**24**	**14**
		RM ≥ 6	na	na	na	na	**12**	**5.1**	**25**	**14**
		by reach	**5.6–25**	**1.5–6.4**	**7.1–31**	**4.1–18**	**4.6–15**	**2.0–6.2**	**10–30**	**5.5–17**
	2[g]	site wide	**40**	**10**	**51**	**29**	**11**	**4.7**	**23**	**13**
		RM ≥ 6	na	na	na	na	**13**	**5.6**	**27**	**16**
		by reach	**13–72**	**3.5–19**	**17–92**	**10–53**	**5.6–15**	2.4–**6.2**	**12–31**	**6.6–17**
PCDD/PCDF TEQ - bird	1[f]	site wide	**13**	**3.5**	**17**	**9.8**	**12**	**5.2**	**26**	**15**
		RM ≥ 6	na	na	na	na	**13**	**5.6**	**27**	**16**
		by reach	**4.2–31**	**1.1–8.0**	**5.3–39**	**3.1–22**	**3.8–18**	**1.6–7.5**	**7.8–37**	**4.5-21**
	2[g]	site wide	**41**	**11**	**53**	**30**	**9.6**	**4.1**	**20**	**11**
		RM ≥ 6	na	na	na	na	**11**	**4.6**	**22**	**13**
		by reach	**12–75**	**3.0–19**	**15–95**	**8.4–54**	**4.2–14**	**1.8–6.1**	**8.7–30**	**5.0–17**

Table 8-29. Bird egg tissue NOAEL HQs

COPEC	Diet Scenario	Area	Great Blue Heron[a]: HQ Based on TRV-A[b] – Species-specific BMF[d]	Great Blue Heron[a]: HQ Based on TRV-B[c] – Alternative BMF[e] – BMF Min.	Great Blue Heron[a]: HQ Based on TRV-B[c] – Alternative BMF[e] – BMF Max.	Great Blue Heron[a]: HQ Based on TRV-B[c] – Alternative BMF[e] – BMF Geomean	Belted Kingfisher[a]: HQ Based on TRV-A[b] – Species-specific BMF[d]	Belted Kingfisher[a]: HQ Based on TRV-B[c] – Alternative BMF[e] – BMF Min.	Belted Kingfisher[a]: HQ Based on TRV-B[c] – Alternative BMF[e] – BMF Max.	Belted Kingfisher[a]: HQ Based on TRV-B[c] – Alternative BMF[e] – BMF Geomean
Total TEQ - bird	1[f]	site wide	**27**	**7.0**	**34**	**20**	**23**	**9.9**	**48**	**28**
		RM ≥ 6	na	na	na	na	**25**	**11**	**51**	**29**
		by reach	**9.8–48**	**2.6–12**	**12–61**	**7.1–35**	**8.5–29**	**3.6–12**	**18–59**	**10–34**
	2[g]	site wide	**81**	**21**	**100**	**58**	**20**	**8.4**	**41**	**23**
		RM ≥ 6	na	na	na	na	**23**	**9.8**	**48**	**27**
		by reach	**35–150**	**9.2–39**	**45–190**	**26–110**	**10–26**	**4.2–11**	**21–55**	**12–31**
Total DDx	1[f]	site wide	**3.0**	**2.5**	**17**	**5.7**	**8.8**	**3.2**	**22**	**7.3**
		RM ≥ 6	na	na	na	na	**8.7**	**3.1**	**22**	**7.1**
		by reach	**1.4–4.9**	**1.2–4.0**	**8.2–28**	**2.7–9.2**	**3.7–10.9**	**1.3–3.9**	**9.2–27**	**3.0–8.9**
	2[g]	site wide	**9.8**	**8.0**	**56**	**18**	**7.5**	**2.7**	**19**	**6.2**
		RM ≥ 6	na	na	na	na	**8.0**	**2.9**	**20**	**6.6**
		by reach	**5.1–18**	**4.2–15**	**29–110**	**10–35**	**4.2–9.2**	**1.5–3.3**	**10–23**	**3.4–7.5**

Bold identifies HQs ≥ 1.0.

[a] The NJDEP acknowledges that the BERA for the LPRSA identifies unacceptable risk. However, the *NJDEP's Ecological Evaluation Technical Guidance*, published in August 2018 (NJDEP 2018), does not advocate the use of more than one set of TRVs for individual contaminant-receptor pairs. It is the NJDEP's position that a single TRV set (NOAEL and LOAEL) that evaluates the more sensitive species and endpoints to characterize risk to invertebrates, fish, birds, and wildlife should be selected in a BERA, not two sets of TRVs, as presented in this document. It is also the NJDEP's position that, for the LPRSA, use of

one conservative TRV set derived for sensitive receptors and sensitive endpoints most clearly demonstrates the degree of risk for individual contaminant-receptor pairs and ensures protection of threatened, endangered, and species of special concern.

[b] TRVs were derived from the primary literature based on the process identified in Section 8.2.3.1.

[c] TRVs were based on USEPA's revised draft of the LPR restoration project FFS (Louis Berger et al. 2014) or first draft of the LPR restoration project FFS (Malcolm Pirnie 2007b).

[d] BMFs were derived based on the process identified in Section 8.2.2.2.

[e] BMFs were based on the process identified by USEPA (2015b), USEPA (2015c), and USEPA (2016g).

[f] Diet Scenario 1 includes 100% 0–13-cm fish for great blue heron and 85% 0–9-cm fish and 15% blue crab for belted kingfisher.

[g] Diet Scenario 2 for great blue heron includes 17% 0–13-cm fish, 29% 13–18-cm fish, 40% 18–30-cm fish, and 14% > 30-cm fish. Diet Scenario 2 for belted kingfisher includes 15% blue crab, 31.5% 0–9-cm fish, 51% 9–13-cm fish, and 2.5% 13–18-cm fish.

BERA – baseline ecological risk assessment
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COPEC – chemical of potential ecological concern
DDD – dichlorodiphenyldichloroethane
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DDT – dichlorodiphenyltrichloroethane
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NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
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PCDF – polychlorinated dibenzofuran
RM – river mile
TEQ – toxic equivalent
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT and 4,4′-DDT)
TRV – toxicity reference value
USEPA – US Environmental Protection Agency

Total PCBs

For great blue heron, LOAEL HQs for total PCBs ranged from 0.18 to 164 on a site-wide basis, and ranged from 0.078 to 284 by reach. For belted kingfisher, LOAEL HQs ranged from 0.51 to 10 on a site-wide basis, from 0.54 to 10 for RM ≥ 6, and from 0.22 to 13 by reach. LOAEL HQs were generally greater using diet Scenario 2 (includes fish > 13 cm) compared to diet Scenario 1 (only fish ≤ 13 cm).

PCB TEQ - Bird

For great blue heron, LOAEL HQs for PCB TEQ ranged from 1.5 to 20 on a site-wide basis, and ranged from 0.56 to 36 by reach. For belted kingfisher, LOAEL HQs for PCB TEQ ranged from 1.1 to 9.5 on a site-wide basis, from 1.2 to 11 for RM ≥ 6, and from 0.46 to 12 by reach. LOAEL HQs were generally greater using diet Scenario 2 (includes fish > 13 cm) compared to diet Scenario 1 (only fish ≤ 13 cm).

PCDD/PCDF TEQ - Bird

For great blue heron, LOAEL HQs ranged from 1.3 to 21 on a site-wide basis, and ranged from 0.42 to 37 by reach. For belted kingfisher, LOAEL HQs ranged from 1.0 to 10 on a site-wide basis, from 1.1 to 11 for RM ≥ 6, and from 0.38 to 14 by reach.

Total TEQ - Bird

For great blue heron, LOAEL HQs ranged from 2.7 to 40 on a site-wide basis, and ranged from 1.0 to 74 by reach. For belted kingfisher, LOAEL HQs ranged from 2.0 to 19 on a site-wide basis, from 2.3 to 20 for RM ≥ 6, and from 0.85 to 23 by reach.

Total DDx

For great blue heron, LOAEL HQs ranged from 0.30 to 4.7 on a site-wide basis, and ranged from 0.14 to 18 by reach. For belted kingfisher, LOAEL HQs ranged from 0.75 to 3.7 on a site-wide basis, from 0.80 to 3.6 for RM ≥ 6, and from 0.22 to 4.6 by reach.

8.2.4.2 Uncertainties in risk characterization

This section discusses uncertainties associated with exposure assumptions, EPCs, and selected TRVs that could affect HQ calculations for bird eggs. Uncertainties associated with the TEFs pertaining to TEQ calculations are the same as those discussed in Section 8.1.4.2 for the bird diet.

General TRV uncertainties, including the derivation of TRVs using SSDs, are discussed in Sections 8.2.3.2 and 6.3.3.1. Uncertainties associated with dietary assumptions are discussed in Section 8.1.4.3. Uncertainties associated with selected BMFs cannot be quantified. However, a range in BMFs was evaluated. Species-specific BMFs were used, when available from the literature. When species-specific BMFs were not available, a geometric mean BMF across all species reported in the general literature was used. In addition, a range of BMFs (i.e., the minimum and maximum BMF) and the geometric mean of the BMFs across all species were evaluated. This section discusses and presents an analysis of the uncertainties associated with exposure area assumptions, EPC calculations, and TRVs.

Exposure Assumptions and EPCs Uncertainties

The uncertainties addressed in this section are as follows:

- **Crab consumption by great blue heron –** No crab was included in the great blue heron diet. The effect of using 1 and 5% crab in the diet was evaluated.
- **Site use factor –** An SUF of 1 was used for belted kingfisher and great blue heron exposure. The effect on HQs of using an alternative SUF of 0.5 was evaluated for great blue heron.
- **Prey size –** The original HQ calculations for great blue heron and belted kingfisher assumed that these species consumed only fish less than a certain size limit in Scenario 1 for each species. This evaluation considered the possibility that great blue heron and belted kingfisher consume fish of any size and without any preference for a particular size class.
- **Treatment of non-detects for EPCs –** The concentrations of congeners that were not detected were assumed to be zero when calculating total PCBs, and TEQs were calculated using USEPA's TEQ calculator (USEPA 2014) using the Kaplan-Meier method. The effect on HQs of using one–half the DL or the full DL was evaluated for total PCBs. The effect on TEQ HQs of using zero, one-half the DL, or the full DL was evaluated for total TEQ.

The effect of these uncertainties on HQ calculations is presented in Tables 8-30 through 8-34 for one diet scenario for each bird species evaluated.

Table 8-30. Bird egg tissue HQs based on uncertainties in exposure parameters, EPCs, and selected TRVs for PCB congeners

	Parameter Values/Assumptions		Total PCBs Range of LOAEL HQs[a]							
			HQ Based on TRV-A[b]		HQ Based on TRV-B[c]					
					Alternative BMF[e]					
			Species-specific BMF[d]		BMF Min.		BMF Max.		BMF Geomean	
Uncertainty	Original	Adjusted	Original	Adjusted	Original	Adjusted	Original	Adjusted	Original	Adjusted
Great blue heron (Scenario 1, site wide)										
Proportion of crab in diet	0%	1%	0.18	0.17	**2.2**	**2.1**	**49**	**48**	**8.2**	**8.2**
		5%		0.17		**2.1**		**48**		**8.0**
SUF	1	0.5		0.088		**1.1**		**24**		**4.1**
Prey size	preference among fish sizes	no fish size preference		0.75		**9.2**		**209**		**35**
Treatment of non-detects	DL = 0 for non-detects	use of one-half the DL or the full DL for non-detects[f]		0.18		**2.2**		**49**		**8.2**
Belted kingfisher (Scenario 1, site wide)										
Prey size	preference among fish sizes	no fish size preference	0.61	**1.3**	**2.7**	**5.8**	**61**	**130**	**10**	**22**
Treatment of non-detects	DL = 0 for non-detects	use of one-half the DL or the full DL for non-detects[f]		0.61		**2.7**		**61**		**10**

Bold identifies HQs ≥ 1.0 based on a LOAEL TRV.

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

b TRVs were derived from the primary literature based on process identified in Section 8.2.3.1.

c TRVs were based on USEPA's revised draft of the LPR restoration project FFS (Louis Berger et al. 2014) or first draft of the LPR restoration project FFS (Malcolm Pirnie 2007b).

d BMFs were derived based on process identified in Section 8.2.2.2.

[e] BMFs were based on process identified by USEPA (2015b), USEPA (2015c), and USEPA (2016g).

[f] HQs are the same regardless of treatment of non-detected values as one-half the DL or as the full DL.

BERA – baseline ecological risk assessment
BMF – biomagnification factor
DL – detection limit
EPC – exposure point concentration
FFS – focused feasibility study
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
LPR – Lower Passaic River
LPRSA – Lower Passaic River study Area
NJDEP – New Jersey Department of Environmental Protection
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
SUF – site use factor
TRV – toxicity reference value
USEPA – US Environmental Protection Agency

Table 8-31. Bird egg tissue HQs based on uncertainties in exposure parameters, EPCs, and selected TRVs for total DDx

Uncertainty	Parameter Values/Assumptions		Total DDx Range of LOAEL HQs[a]							
			HQ Based on TRV-A[b]		HQ Based on TRV-B[c]					
					Alternative BMF[e]					
			Species-specific BMF[d]		BMF Min.		BMF Max.		BMF Geomean	
	Original	Adjusted	Original	Adjusted	Original	Adjusted	Original	Adjusted	Original	Adjusted
Great blue heron (Scenario 1, site wide)										
Proportion of crab in diet	0%	1%	0.30	0.30	0.41	0.41	**2.9**	**2.9**	0.95	0.95
		5%		0.30		0.42		**2.9**		0.96
SUF	1	0.5		0.15		0.21		**1.5**		0.48
Prey size	preference among fish sizes	no fish size preference		**1.1**		**1.5**		**11**		**3.5**
Treatment of non-detects	DL = 0 for non-detects	use of one-half the DL or the full DL for non-detects[f]		0.31		0.42		**3.0**		0.97
Belted kingfisher (Scenario 1, site wide)										
Prey size	preference among fish sizes	no fish size preference	0.89	**1.3**	0.53	0.75	**3.7**	**5.3**	**1.2**	**1.7**
Treatment of non-detects	DL = 0 for non-detects	use of one-half the DL or the full DL for non-detects[f]		0.90		0.54		**3.8**		**1.2**

Bold identifies HQs ≥ 1.0.

Shaded cells identify HQs ≥ 1.0 based on a LOAEL TRV.

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

b TRVs were derived from the primary literature based on the process identified in Section 8.2.3.1.

c TRVs were based on USEPA's revised draft of the LPR restoration project FFS (Louis Berger et al. 2014) or first draft of the LPR restoration project FFS (Malcolm Pirnie 2007b).

d BMFs were derived based on the process identified in Section 8.2.2.2.

e BMFs were based on the process identified by USEPA (2015b), USEPA (2015c), and USEPA (2016g).

[f] HQs are the same regardless of treatment of non-detected values as one-half the DL or as the full DL.

BERA – baseline ecological risk assessment
BMF – biomagnification factor
DL – detection limit
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
EPC – exposure point concentration
FFS – focused feasibility study
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
LPR – Lower Passaic River
LPRSA – Lower Passaic River study Area
NJDEP – New Jersey Department of Environmental Protection
NOAEL – no-observed-adverse-effect level
SUF – site use factor
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT and 4,4′-DDT)
TRV – toxicity reference value
USEPA – US Environmental Protection Agency

Table 8-32. Bird egg tissue HQs based on uncertainties in exposure parameters, EPCs, and selected TRVs for total PCB TEQ - bird

	Parameter Values/Assumptions		PCB TEQ - Bird Range of LOAEL HQs[a]							
			HQ Based on TRV-A[b]		HQ Based on TRV-B[c]					
					Alternative BMF[e]					
			Species-specific BMF[d]		BMF Min.		BMF Max.		BMF Geomean	
Uncertainty	Original	Adjusted	Original	Adjusted	Original	Adjusted	Original	Adjusted	Original	Adjusted
Great blue heron (Scenario 1, site wide)										
Proportion of crab in diet	0%	1%	**1.5**	**1.5**	**1.5**	**1.5**	**7.3**	**7.3**	**4.2**	**4.2**
		5%		**1.5**		**1.6**		**7.6**		**4.3**
SUF	1	0.5		0.73		0.75		**3.6**		**2.1**
Prey size	preference among fish sizes	no fish size preference		**3.1**		**3.2**		**15**		**8.8**
Treatment of non-detects	DL = 0 for non-detects	use of one-half the DL or the full DL for non-detects[f]		**1.5**		**1.5**		**7.3**		**4.2**
Belted kingfisher (Scenario 1, site wide)										
Prey size	preference among fish sizes	no fish size preference	**1.2**	**2.4**	**1.9**	**4.1**	**9.5**	**20**	**5.4**	**11**
Treatment of non-detects	use of Kaplan-Meier method in USEPA's TEQ calculator (USEPA 2014) for TEQs	use of DL = 0, one-half the DL, or the full DL for non-detects[f]		**1.2**		**1.9**		**9.5**		**5.4**

Bold identifies HQs ≥ 1.0 based on a LOAEL TRV.

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

[b] TRVs were derived from the primary literature based on the process identified in Section 8.2.3.1.

[c] TRVs were based on USEPA's revised draft of the LPR restoration project FFS (Louis Berger et al. 2014) or first draft of the LPR restoration project FFS (Malcolm Pirnie 2007b).

[d] BMFs were derived based on the process identified in Section 8.2.2.2.

[e] BMFs were based on the process identified by USEPA (2015b), USEPA (2015c), and USEPA (2016g).

[f] HQs are the same regardless of treatment of non-detected values as equal to zero, one-half the DL, or as the full DL.

BERA – baseline ecological risk assessment
BMF – biomagnification factor
DL – detection limit
EPC – exposure point concentration
FFS – focused feasibility study
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
LPR – Lower Passaic River
LPRSA – Lower Passaic River study Area
NJDEP – New Jersey Department of Environmental Protection
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
SUF – site use factor
TEQ – toxic equivalent
TRV – toxicity reference value
USEPA – US Environmental Protection Agency

Table 8-33. Bird egg tissue HQs based on uncertainties in exposure parameters, EPCs, and selected TRVs for total PCDD/PCDF TEQ - bird

Uncertainty	Parameter Values/Assumptions		PCDD/PCDF TEQ - Bird Range of LOAEL HQs[a]							
			HQ Based on TRV-A[b]		HQ Based on TRV-B[c]					
					Alternative BMF[e]					
			Species-specific BMF[d]		BMF Min.		BMF Max.		BMF Geomean	
	Original	Adjusted	Original	Adjusted	Original	Adjusted	Original	Adjusted	Original	Adjusted
Great Blue Heron (Scenario 1, site wide)										
Proportion of crab in diet	0%	1%		**1.4**		**1.4**		**6.8**		**3.9**
		5%		**1.4**		**1.4**		**6.9**		**3.9**
SUF	1	0.5		0.67		0.69		**3.4**		**1.9**
Prey size	preference among fish sizes	no fish size preference	**1.3**	**5.0**	**1.4**	**5.2**	**6.7**	**25**	**3.8**	**14**
Treatment of non-detects	DL = 0 for non-detects	use of one-half the DL or the full DL for non-detects[f]		**1.3**		**1.4**		**6.7**		**3.8**
Belted Kingfisher (Scenario 1, site wide)										
Prey size	preference among fish sizes	no fish size preference		**1.9**		**3.2**		**16**		**9.0**
Treatment of non-detects	use of Kaplan-Meier method in USEPA's TEQ calculator (USEPA 2014) for TEQs	use of DL = 0, one-half the DL, or the full DL for non-detects[f]	**1.2**	**1.2**	**2.1**	**2.1**	**10**	**10**	**5.7**	**5.7**

Bold identifies HQs ≥ 1.0 based on a LOAEL TRV.

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

b TRVs were derived from the primary literature based on the process identified in Section 8.2.3.1.

d TRVs were based on USEPA's revised draft of the LPR restoration project FFS (Louis Berger et al. 2014) or first draft of the LPR restoration project FFS (Malcolm Pirnie 2007b).

d BMFs were derived based on the process identified in Section 8.2.2.2.

e BMFs were based on the process identified by USEPA (2015b), USEPA (2015c), and USEPA (2016g).

[f] HQs are the same regardless of treatment of non-detected values as equal to zero, one-half the DL, or as the full DL.

BERA – baseline ecological risk assessment
BMF – biomagnification factor
DL – detection limit
EPC – exposure point concentration
FFS – focused feasibility study
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
LPR – Lower Passaic River
LPRSA – Lower Passaic River study Area
NJDEP – New Jersey Department of Environmental Protection
NOAEL – no-observed-adverse-effect level
PCDD - polychlorinated dibenzo-*p*-dioxin
PCDF – polychlorinated dibenzofuran
SUF – site use factor
TEQ – toxic equivalent
TRV – toxicity reference value
USEPA – US Environmental Protection Agency

Table 8-34. Bird egg tissue HQs based on uncertainties in exposure parameters, EPCs, and selected TRVs for total TEQ - bird

Uncertainty	Parameter Values/Assumptions		Total TEQ - Bird Range of LOAEL HQs[a]							
			HQ Based on TRV-A[b]		HQ Based on TRV-B[c]					
			Species-specific BMF[d]		Alternative BMF[e]					
					BMF Min.		BMF Max.		BMF Geomean	
	Original	Adjusted	Original	Adjusted	Original	Adjusted	Original	Adjusted	Original	Adjusted
Great Blue Heron (Scenario 1, site wide)										
Proportion of crab in diet	0%	1%	**2.7**	**2.7**	**2.8**	**2.8**	**13**	**14**	**7.7**	**7.7**
		5%		**2.8**		**2.9**		**14**		**8.0**
SUF	1	0.5		**1.3**		**1.4**		**6.7**		**3.8**
Prey size	preference among fish sizes	no fish size preference		**7.8**		**8.0**		**39**		**22**
Treatment of non-detects	DL = 0 for non-detects	use of one-half the DL or the full DL for non-detects[f]		**2.7**		**2.8**		**13**		**7.7**
Belted Kingfisher (Scenario 1, site wide)										
Prey size	preference among fish sizes	no fish size preference	**2.3**	**4.4**	**3.9**	**7.3**	**19**	**36**	**11**	**20**
Treatment of non-detects	use of Kaplan-Meier method in USEPA's TEQ calculator (USEPA 2014) for TEQs	use of DL = 0, one-half the DL or the full DL for non-detects[f]		**2.3**		**3.9**		**19**		**11**

Bold identifies HQs ≥ 1.0 based on a LOAEL TRV.

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

[b] TRVs were derived from the primary literature based on the process identified in Section 8.2.3.1.

[c] TRVs were based on USEPA's revised draft of the LPR restoration project FFS (Louis Berger et al. 2014) or first draft of the LPR restoration project FFS (Malcolm Pirnie 2007b).

[d] BMFs were derived based on the process identified in Section 8.2.2.2.

[e] BMFs were based on the process identified by USEPA (2015b), USEPA (2015c), and USEPA (2016g).

[f] HQs are the same regardless of treatment of non-detected values as one-half the DL or as the full DL.

BERA – baseline ecological risk assessment
BMF – biomagnification factor
DL – detection limit
EPC – exposure point concentration
FFS – focused feasibility study
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
LPR – Lower Passaic River
LPRSA – Lower Passaic River study Area
NJDEP – New Jersey Department of Environmental Protection
NOAEL – no-observed-adverse-effect level
SUF – site use factor
TEQ – toxic equivalent
TRV – toxicity reference value
USEPA – US Environmental Protection Agency

The results of this evaluation are summarized in Table 8-35 and are also summarized below:

- **Crab consumption by great blue heron –** Inclusion of crab in the diet of great blue heron resulted in relatively small changes to the HQs (maximum of ± 1.0 units).
- **Site use factor –** The use of an SUF of 0.5 rather than 1.0 for great blue heron decreased all the HQs by one-half and resulted in some LOAEL HQs that had been ≥ 1.0 for PCB TEQ - bird and PCDD/PCDF TEQ - bird becoming < 1.0.
- **Prey size –** As a result of not assigning prey preferences for fish consumption in the diets of great blue heron and belted kingfisher, HQ values increased by a maximum of 160 units for great blue heron and a maximum of 69 units for kingfisher.
- **Treatment of non-detects for EPCs –** Different treatments of non-detects for calculating EPCs for organic compounds and TEQs - bird resulted in only small changes to the HQs (maximum of ± 0.1 units).

Table 8-35. Summary of uncertainties evaluated for bird egg tissue

Species	General Uncertainty Evaluated	Specific Evaluation Conducted	Rationale	Difference in HQ[a]
Great blue heron	exclusion of blue crab	Include crab as 1 and 5% of overall diet (and fish as 99 and 95%, respectively).	Evaluate effect on risk estimates based on inclusion of blue crab in the great blue heron diet.	≤ 1.0 (±)
Great blue heron	assumption of 100% site use	Evaluate SUF of 0.5 (rather than 1).	Evaluate the effect on risk estimates when assuming use of the LPRSA only seasonally.	≤ 25 (-)
Great blue heron	selected portions of fish prey size classes	Group all fish prey as a single size class (0 to < 30 cm) rather than dividing by size class.	Evaluate difference in risk estimates based on grouping all fish prey together to derive EPCs vs. dividing into size classes with portions assigned based on the general literature.	≤ 160 (+)
Great blue heron	treatment of non-detects (DL = 0 for PCB congeners and Kaplan-Meier method for TEQ)	Include DL = 0 (for TEQ only), one-half the DL, or the full DL for non-detects.	Evaluate the effect on risk estimates based on treatment of non-detects.	≤ 0.1 (+)

Table 8-35. Summary of uncertainties evaluated for bird egg tissue

Species	General Uncertainty Evaluated	Specific Evaluation Conducted	Rationale	Difference in HQ[a]
Belted kingfisher	selected portions of fish prey size classes	Group all fish prey as a single size class (0 to 18 cm) rather than dividing by size class.	Evaluate difference in risk estimates based on grouping all fish prey together to derive EPCs vs. dividing into size classes with portions assigned based on the general literature.	≤ 69 (+)
Belted kingfisher	treatment of non-detects (DL = 0 for PCB congeners and Kaplan-Meier method for TEQ)	Include DL = 0 (for TEQ only), one-half the DL, or the full DL for non-detects.	Evaluate the effect on risk estimates based on treatment of non-detects.	≤ 0.1 (+)

[a] Differences in HQs (based on a LOAEL TRV) were calculated from the data presented in Tables 8-30 through Table 8-34, and are based on the site-wide exposure area and diet Scenario 1 great blue heron (i.e., 100% 0–13-cm fish) and belted kingfisher (i.e., 15% blue crab and 85% 0–9 cm-fish). Direction of the HQ change is provided in parentheses.

DL – detection limit
EPC – exposure point concentration
HQ – hazard quotient
LPRSA – Lower Passaic River Study Area
LOAEL – lowest-observed-adverse-effect level
PCB – polychlorinated biphenyl
SUF – site use factor
TEQ – toxic equivalent
TRV – toxicity reference value
USEPA – US Environmental Protection Agency

TEQ Uncertainty

TEQs represent uncertain estimates because they are calculated using TEFs that are highly variable. The four most-potent dioxin-like PCBs in birds are PCBs 77, 81, 126, and 169 (i.e., those with the highest TEFs and thus contributing most to TEQs). For PCB 77, 5 studies produced TEFs ranging over 3 orders of magnitude (< 0.0003 to 0.15) for the various bird species tested (Van den Berg et al. 1998). For PCB 81, two studies tested several species and identified TEFs ranging from 0.001 to 0.5. For PCBs 126 and 169, data were available from only one study, so the associated uncertainty has not yet been quantified.

The high variability in TEFs may be due, in part, to differences in species sensitivity to dioxin-like compounds. Bird species can be grouped into three general classes of sensitivity to dioxin-like compounds, based on documented species differences in the amino acid sequences at the Ah receptor. These species differences affect not only overall sensitivity, but also the relative potency (i.e., TEFs) of individual dioxin-like compounds (Farmahin et al. 2013). Another issue with bird TEFs is their relevance for assessing dietary exposure risks. TEFs are estimated based on ethoxyresorufin-O-deethylase induction or *in ovo* studies. Such studies are most accurate for assessing effects based on tissue congener concentrations in whole embryos (USEPA 2008), not dietary exposure studies.

Ring-necked pheasants and chickens (the two species in studies used to derive TEQ TRVs) are in the moderate- and high-sensitivity groups, respectively, as classified by Farmahin et al. (2013). The derivation of TRVs using moderately to highly sensitive species indicates that the risk calculations are more likely to overestimate than to underestimate risk.

TRV Uncertainty

General TRV uncertainties, including the derivation of TRVs using SSDs, are discussed in Sections 8.2.3.2 and 6.3.3.1. For the COPECs with TRVs based on 5th percentile LOAELs determined from SSDs (i.e., TEQ - bird and total DDx), the range of the empirical LOAELs and number of data points (i.e., number of species included in the SSD) are shown in Table 8-36 to provide a context of uncertainty for SSD-derived values.

Table 8-36. Uncertainty evaluation of bird egg tissue TRVs based on SSDs

COPEC	TRV Unit (ww)	NOAEL	LOAEL	No. of species (count of LOAELs in SSD)	Empirical LOAEL Range	Notes on Key Uncertainties
TEQ - bird	ng/kg	100	250	n = 5	1,000–40,000	SSD-derived LOAEL < lowest measured LOAEL
Total DDx	µg/kg	3,900	4,100	n = 7	12,000–658,000	SSD-derived LOAEL < lowest measured LOAEL

Note: TRVs included in this table are based on SSDs that are based on TRVs derived from the general literature search.

COPEC – chemical of potential ecological concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
SSD – species sensitivity distribution
TEQ – toxic equivalency
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT and 4,4′-DDT)
TRV – toxicity reference value
ww – wet weight

8.2.4.3 Comparison to background

Consistent with USEPA guidance (USEPA 2002c), Section 8.1.4.3 presents background concentrations for prey for bird species and COPECs with LOAEL HQs ≥ 1 (total PCBs, PCB TEQ - bird, PCDD/PCDF TEQ - bird, total TEQ - bird, and total DDx). Three background datasets were developed for use in this BERA using data available from the following areas: 1) upstream of Dundee Dam, to represent freshwater urban habitat, 2) Jamaica Bay/Lower Harbor, to represent estuarine urban habitat, and 3) Mullica River/Great Bay, to represent estuarine/freshwater rural habitat. These datasets are summarized in Section 4.2, and details on how background values were determined from these datasets are presented in Appendix J.

8.2.5 Summary of key uncertainties

The uncertainty associated with the bird egg risk characterization pertains to the use of BMFs to model bird egg concentrations. Although uncertainties associated with selected BMFs cannot be quantified, a range in BMFs was evaluated. There are also uncertainties associated with the selected TRVs used in risk calculation.

The adjustments in the dietary composition to include a greater consumption of larger fish in the diet (i.e., Scenario 2) resulted in slightly higher HQs for great blue heron and belted kingfisher than the scenario in which only small fish are consumed (i.e., Scenario 1). In addition, the evaluation of risk by reach resulted in slightly higher HQs in some specific reaches. This approach may overestimate risk to great blue heron because they have a relatively large home range.

Other uncertainties relate to the TEQ methodology and the use of laboratory toxicity data to predict effects, which could be either over- or underestimated. HQs are more likely to represent an overestimation of risk because of the conservative assumptions used in the risk evaluation. These assumptions include the use of the lowest LOAEL among all species or endpoints as the TRV, the use of an upper exposure value (i.e., UCL) to calculate dietary exposure concentrations, and the assumption that a species feeds exclusively from the LPRSA (i.e., SUF = 1).

8.2.6 Summary

Seven COPECs were evaluated based on modeled bird egg concentrations. LOAEL HQs were ≥ 1.0 for total PCBs, TEQ - bird, and DDx. Table 8-37 provides the range of LOAEL HQs for all exposure area scenarios, using a range of TRVs and BMFs for the COPECs with LOAEL HQs ≥ 1.

Table 8-37. Summary of bird egg tissue LOAEL HQs

COPEC[b]	Exposure Area	Range of LOAEL HQs[a]								Key Uncertainties
		Belted Kingfisher				Great Blue Heron				
		HQ Based on TRV-A[c]	HQ Based on TRV-B[d]			HQ Based on TRV-A[c]	HQ Based on TRV-B[d]			
			Alternative BMF[f]				Alternative BMF[f]			
		Species-specific BMF[e]	BMF Min.	BMF Max.	BMF Geo-mean	Species-specific BMF[e]	BMF Min.	BMF Max.	BMF Geo-mean	
Total PCBs	site wide/ RM ≥ 6	0.51–0.62	**2.2–2.7**	**50–61**	**8.5–10**	0.18– 0.59	**2.2–7.3**	**49–164**	**8.2–28**	• TRV-A based on non-chicken reproduction and limited dataset (two studies) • TRV-B based on interpolated value from chicken hatchability data based on a 25% decrease relative to control; TRV-B approximately three times less than measured LOAEL (for which hatchability was reduced by 55%) • Uncertainty associated with use of literature-based BMFs used to predict bird egg concentrations; species-specific BMF for heron and kingfisher based heron and kingfisher data, respectively, for comparison to TRV-A, and range of BMFs evaluated for comparison to TRV-B
	by reach	0.22–0.77	**1.0–3.4**	**22–76**	**3.7–13**	0.078–**1.0**	**1.0–13**	**22–284**	**3.7–48**	
PCB TEQ - bird	site wide/ RM ≥ 6	**1.1–1.3**	**1.8–2.2**	**9.0–11**	**5.2–6.1**	**1.5–4.0**	**1.5–4.1**	**7.3–20**	**4.2–11**	• TRV-A based on SSD with no chicken reproduction data (SSD not expected to have changed significantly with inclusion of chicken data) and less than lowest measured LOAEL • TRV-B based on SSD inclusive of chicken reproduction data • TEQ sensitivities vary with Ah receptor; chicken in high-sensitivity group and great blue heron in low-sensitivity group • Species-specific BMF for heron and kingfisher based heron and kingfisher data, respectively
	by reach	0.46–**1.5**	0.77–**2.5**	**3.8–12**	**2.2–6.9**	0.56–**7.2**	0.57–**7.4**	**3.2–36**	**1.6–21**	
PCDD/ PCDF TEQ - bird	site wide/ RM ≥ 6	0.96–**1.3**	**1.6–2.2**	**7.8–11**	**4.5–6.2**	**1.3–4.1**	**1.4–4.2**	**6.7–21**	**3.8–12**	
	by reach	0.38–**1.8**	0.63–**2.9**	**3.4–14**	**1.8–8.2**	0.42–**7.5**	0.43–**7.6**	**2.1–37**	**1.2–21**	
Total TEQ - bird	site wide/ RM ≥ 6	**2.0–2.5**	**3.3–4.1**	**16–20**	**9.2–12**	**2.7–8.0**	**2.8–8.2**	**13–40**	**7.7–23**	
	by reach	0.85–**2.9**	**1.4–4.8**	**6.9–23**	**3.9–13**	**1.0–15**	**1.0–15**	**4.9–74**	**2.8–42**	

COPEC[b]	Exposure Area	Range of LOAEL HQs[a]								Key Uncertainties
		Belted Kingfisher				Great Blue Heron				
		HQ Based on TRV-A[c]	HQ Based on TRV-B[d]			HQ Based on TRV-A[c]	HQ Based on TRV-B[d]			
			Alternative BMF[f]				Alternative BMF[f]			
		Species-specific BMF[e]	BMF Min.	BMF Max.	BMF Geo-mean	Species-specific BMF[e]	BMF Min.	BMF Max.	BMF Geo-mean	
Total DDx	site wide/ RM ≥ 6	0.75–0.89	0.45–0.53	**3.2–3.7**	**1.0–1.2**	0.30–0.98	0.41–**1.3**	**2.9–9.4**	0.95–**3.1**	• TRV-A based on SSD not inclusive of chicken reproduction data • TRV-B based on SSD inclusive of chicken reproduction data • Uncertainty associated with use of literature-based BMFs used to predict bird egg concentrations; species-specific BMF for heron based heron data, and species-specific BMF for kingfisher based on geomean of 5 species for comparison to TRV-A and range of BMFs evaluated for comparison to TRV-B
	by reach	0.37–**1.1**	0.22–0.65	**1.5–4.6**	0.50–**1.5**	0.14–**1.8**	0.19–**2.5**	**1.4–18**	0.44–**5.8**	

Bold identifies HQs ≥ 1.0.

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

[b] Only COPECs with HQs ≥ 1.0 based on a LOAEL TRV are included in the table.

[c] TRVs were derived from the primary literature based on the process identified in Section 8.2.3.1.

[d] TRVs were based on USEPA's revised draft of the LPR restoration project FFS (Louis Berger et al. 2014) or first draft of the LPR restoration project FFS (Malcolm Pirnie 2007b).

[e] BMFs were derived based on process identified in Section 8.2.2.2.

[f] BMFs were derived based on the process identified by USEPA (2015b), USEPA (2015c), and USEPA (2016g).

Ah – aryl hydrocarbon
BERA – baseline ecological risk assessment
BMF – biomagnification factor
COPEC – chemical of potential ecological
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
HQ – hazard quotient
FFS – focused feasibility study
LOAEL – lowest-observed-adverse-effect level
LPR – Lower Passaic River
LPRSA – Lower Passaic River study Area
NJDEP – New Jersey Department of Environmental Protection
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
PCDD – polychlorinated dibenzo-*p*-dioxin
PCDF – polychlorinated dibenzofuran
RM – river mile
SSD – species sensitivity distribution
TEQ – toxic equivalent
TRV – toxicity reference value
USEPA – US Environmental Protection Agency
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT and 4,4′-DDT)

8.3 SUMMARY OF PRELIMINARY COCS FOR BIRDS

The potential for unacceptable risk to aquatic birds from COPECs in the LPRSA was evaluated based on the CSM presented in Section 3. Specifically, the risk assessment for birds evaluated Assessment Endpoint No. 6:

- Protection and maintenance (i.e., survival, growth, and reproduction) of herbivorous, omnivorous, sediment–probing, and piscivorous bird populations

The potential for risk to three bird species (spotted sandpiper, great blue heron, and belted kingfisher) was characterized using LPRSA tissue, sediment, and water chemistry to estimate dietary doses. In addition, risks to great blue heron and belted kingfisher were characterized using chemical concentrations in bird egg tissue as a secondary LOE. Dietary doses and modeled bird egg concentrations were compared to a range of TRVs to derive risk estimates (HQs) in the risk characterization.

Bird species-COPEC pairs with effect-level HQs ≥ 1.0 (based on a LOAEL TRV) in at least one LOE were identified as preliminary COCs (listed in Table 8-38). Based on these criteria, the following preliminary COCs were identified:

- **Spotted sandpiper:** PCDD/PCDF TEQ - bird, total TEQ - bird, copper, lead, total HPAHs, total PCBs, PCB TEQ - bird, PCB TEQ - bird, and total DDx were identified as preliminary COCs based on the dietary LOE.
- **Great blue heron:** PCDD/PCDF TEQ - bird, total TEQ - bird, copper, methylmercury, total PCBs, PCB TEQ - bird, and total DDx were identified as preliminary COCs based on the dietary LOE; total PCBs, PCB TEQ - bird, PCDD/PCDF TEQ - bird, total TEQ - bird, and total DDx were identified as preliminary COCs based on the egg tissue LOE.
- **Belted kingfisher:** PCDD/PCDF TEQ - bird, total TEQ - bird, lead, methylmercury, PCB TEQ - bird, and total DDx were identified as preliminary COCs based on the dietary LOE; PCDD/PCDF TEQ - bird, total TEQ - bird, total PCBs, PCB TEQ - bird, and total DDx were identified as preliminary COCs based on the egg tissue LOE.

Table 8-38. Summary of preliminary COCs for birds

Preliminary COC and Species[b]	Exposure Area[c]	Range of LOAEL HQs[a]					
		HQ based on TRV-A[d]		HQ based on TRV-B[e]			
			Bird Egg Tissue LOE		Bird Egg Tissue LOE		
					Alternative BMFs[g]		
		Dietary Dose LOE	Species-specific BMF[f]	Dietary Dose LOE	Min.	Max.	Geomean
Copper							
Spotted sandpiper	site wide	0.5	na	**2.0**	na	na	na
	by reach	0.30–0.88		**1.2–3.6**			
Great blue heron	site wide	0.033–0.094	na	0.13–0.38	na	na	na
	by reach	0.029–0.33		0.12–**1.3**			
Belted kingfisher	site wide	0.18	na	0.72–0.73	na	na	na
	RM ≥ 6	0.18–0.20		0.72–0.80			
	by reach	0.15–0.24		0.61–0.97			
Lead							
Spotted sandpiper	site wide	0.49	na	**7.3**	na	na	na
	by reach	0.20–0.68		**3.0–10**			
Great blue heron	site wide	0.010–0.018	na	0.15–0.27	na	na	na
	by reach	0.0041–0.036		0.060–0.52			
Belted kingfisher	site wide	0.044–0.048	na	0.65–0.71	na	na	na
	RM ≥ 6	0.049–0.056		0.72–0.83			
	by reach	0.015–0.075		0.22–**1.1**			
Methylmercury							
Spotted sandpiper	site wide	0.021	na	0.077	na	na	na
	by reach	0.0072–0.027		0.027–0.10			
Great blue heron	site wide	0.10–0.25	0.055–0.14	0.37–0.94	na	na	na
	by reach	0.031–0.42	0.017–0.23	0.11–**1.6**	na	na	na
Belted kingfisher	site wide	0.25–0.35	0.048–0.066	0.92–**1.3**	na	na	na
	RM ≥ 6	0.22–0.23	0.042–0.043	0.81–0.84	na	na	na
	by reach	0.13–0.43	0.025–0.081	0.48–**1.6**	na	na	na

Table 8-38. Summary of preliminary COCs for birds

Preliminary COC and Species[b]	Exposure Area[c]	Range of LOAEL HQs[a]					
		HQ based on TRV-A[d]		HQ based on TRV-B[e]			
			Bird Egg Tissue LOE		Bird Egg Tissue LOE		
					Alternative BMFs[g]		
		Dietary Dose LOE	Species-specific BMF[f]	Dietary Dose LOE	Min.	Max.	Geomean
Total HPAHs							
Spotted sandpiper	site wide	na	na	**4.9**	na	na	na
	by reach			**1.9–10**			
Great blue heron	site wide			0.10–0.19			
	by reach			0.043–0.40			
Belted kingfisher	site wide			0.36–0.50			
	RM ≥ 6			0.40–0.61			
	by reach			0.11–0.80			
Total PCB Congeners							
Spotted sandpiper	site wide	0.17	na	0.48	na	na	na
	by reach	0.047–0.41	na	0.13–**1.2**			
Great blue heron	site wide	0.070–0.24	0.18–0.59	0.20–0.66	**2.2–7.3**	**49–164**	**8.2–28**
	by reach	0.031–0.41	0.078–**1.0**	0.087–**1.1**	**1.0–13**	**22–284**	**3.7–48**
Belted kingfisher	site wide	0.21–0.25	0.51–0.61	0.59–0.71	**2.2–2.7**	**50–61**	**8.5–10**
	RM ≥ 6	0.22–0.26	0.54–0.62	0.62–0.71	**2.4–2.7**	**54–61**	**9.0–10**
	by reach	0.093–0.32	0.22–0.77	0.26–0.89	**1.0–3.4**	**22–76**	**3.7–13**
PCB TEQ - bird							
Spotted sandpiper	site wide	0.31	na	**1.5**	na	na	na
	by reach	0.073–0.78		0.37–**3.9**			
Great blue heron	site wide	0.067–0.18	**1.5–4.0**	0.33–0.90	**1.5–4.1**	**7.3–20**	**4.2–11**
	by reach	0.030–0.33	0.56–**7.2**	0.13–**1.6**	0.57–**7.4**	**3.2–36**	**1.0–21**
Belted kingfisher	site wide	0.23–0.25	**1.1–1.2**	**1.2**	**1.8–1.9**	**9.0–9.5**	**5.2–5.4**
	RM ≥ 6	0.25–0.28	**1.2–1.3**	**1.3–1.4**	**2.0–2.2**	**9.7–11**	**5.5–6.1**
	by reach	0.10–0.31	0.46–**1.5**	0.49–**1.5**	0.77–**2.5**	**3.8–12**	**2.2–6.9**
PCDD/PCDF TEQ - bird							
Spotted sandpiper	site wide	0.91	na	**4.5**	na	na	na
	by reach	0.014–**4.2**		0.071–**21**			
Great blue heron	site wide	0.067–0.19	**1.3–4.1**	0.33–0.96	**1.4–4.2**	**6.7–21**	**3.8–12**
	by reach	0.020–0.37	0.42–**7.5**	0.10–**1.9**	0.43–**7.6**	**2.1–37**	**1.2–21**

Table 8-38. Summary of preliminary COCs for birds

Preliminary COC and Species[b]	Exposure Area[c]	Range of LOAEL HQs[a]					
		HQ based on TRV-A[d]		HQ based on TRV-B[e]			
			Bird Egg Tissue LOE		Bird Egg Tissue LOE		
					Alternative BMFs[g]		
		Dietary Dose LOE	Species-specific BMF[f]	Dietary Dose LOE	Min.	Max.	Geomean
Belted kingfisher	site wide	0.21–0.27	0.96–**1.2**	**1.0–1.3**	**1.6–2.1**	**7.8–10**	**4.5–5.7**
	RM ≥ 6	0.24–0.29	**1.1–1.3**	**1.2–1.4**	**1.8–2.2**	**8.8–11**	**5.1–6.2**
	by reach	0.090–0.38	0.38–**1.8**	0.44–**1.9**	0.63–**2.9**	**3.1–14**	**1.8–8.2**
Total TEQ - bird							
Spotted sandpiper	site wide	**1.1**	na	**5.4**	na	na	na
	by reach	0.089–**5.0**		0.44–**25**			
Great blue heron	site wide	0.13–0.37	**2.7–8.0**	0.64–**1.8**	**2.8–8.2**	**13–40**	**7.7–23**
	by reach	0.044–0.71	**1.0–15**	0.22–**3.5**	**1.0–15**	**4.9–74**	**2.8–42**
Belted kingfisher	site wide	0.43–0.50	**2.0–2.3**	**2.1–2.5**	**3.3–3.9**	**16–19**	**9.2–11**
	RM ≥ 6	0.50–0.53	**2.3–2.5**	**2.5–2.7**	**3.9–4.1**	**19–20**	**11–12**
	by reach	0.18–0.63	0.85–**2.9**	0.89–**3.1**	**1.4–4.8**	**6.9–23**	**3.9–13**
Total DDx							
Spotted sandpiper	site wide	0.069	na	0.64	na	na	na
	by reach	0.018–0.15		0.16–**1.4**			
Great blue heron	site wide	0.043–0.14	0.30–0.98	0.40–**1.3**	0.41–**1.3**	**2.9–9.4**	0.95–**3.1**
	by reach	0.020–0.26	0.14–**1.8**	0.19–**2.4**	0.19–**2.5**	**1.4–18**	0.44–**5.8**
Belted kingfisher	site wide	0.14–0.16	0.75–0.89	**1.3–1.5**	0.45–0.53	**3.2–3.7**	**1.0–1.2**
	RM ≥ 6	0.14–0.16	0.80–0.87	**1.3–1.4**	0.48–0.52	**3.4–3.6**	**1.1–1.2**
	by reach	0.066–0.20	0.37–**1.1**	0.61–**1.8**	0.22–0.65	**1.5–4.6**	0.50–**1.5**

Bold identifies HQs ≥ 1.0.

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

[b] Preliminary COCs are those COPECs with HQs ≥ 1.0 based on a LOAEL TRV.

[c] HQs are presented by exposure area for both dietary scenarios.

[d] TRVs were derived from the primary literature based on the process identified in Section 8.2.3.1.

[e] TRVs were based on USEPA's revised draft of the LPR restoration project FFS (Louis Berger et al. 2014) or first draft of the LPR restoration project FFS (Malcolm Pirnie 2007b).

[f] BMFs were derived based on the process identified in Section 8.2.2.2.

[g] BMFs were based on the process identified by USEPA (2015b), USEPA (2015c), and USEPA (2016g).

BERA – baseline ecological risk assessment
BMF – biomagnification factor
COC – chemical of concern
COPEC – chemical of potential ecological concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
FFS – focused feasibility study
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
LOE – line of evidence
LPR – Lower Passaic River
LPRSA – Lower Passaic River Study Area
na – not applicable (no TRV available)
NJDEP – New Jersey Department of Environmental Protection
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
PCDD – polychlorinated dibenzo–*p*–dioxin
PCDF – polychlorinated dibenzofuran
RM – river mile
TEQ – toxic equivalent
TRV – toxicity reference value
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT and 4,4′-DDT)

The results of this bird risk assessment will be used in the FS as a tool for risk managers to make potential remedial decisions for the LPRSA. Determining the potential for unacceptable ecological risk at the population level provides information pertaining to decisions to be made in the FS or other programmatic environmental management framework. The TRVs used to evaluate risk to birds in this BERA are organism-level effects, including those that affect particular attributes of organisms within a population, such as survival, growth, and reproduction. Survival, growth, and reproduction of individual organisms may, in turn, affect populations of those organisms depending upon the magnitude and severity of the effect. However, population-level effects—such as size or density of population, population growth, or population survival—are more direct measures of influences on the entire population as a whole. USEPA guidance states that assessment endpoints should be associated with sustaining the ecological structure and function of populations and communities rather than individual organisms, unless individuals warrant additional protection in specific cases (USEPA 1999). Since BERAs evaluate ***populations*** as assessment endpoints, not individuals, a number of other factors, including the potential magnitude and severity of the effect, should be assessed to determine if a risk driver (defined and identified in Section 13) should be used in developing PRGs and RALs.

9 Mammal Assessment

This section presents the risk assessment for the aquatic mammal species (i.e., mink and river otter) selected for evaluation in the LPRSA BERA. The risk assessment for potential aquatic mammal use of the LPRSA evaluated the following assessment endpoint, according to the PFD (Windward and AECOM 2009):

- **Assessment Endpoint No. 7 –** Protection and maintenance (i.e., survival, growth, and reproduction) of aquatic mammal population

Mink tracks were photographed near Dundee Dam in August 2010, therefore the upper part of the LPRSA above the dam, where more habitat is available, may be a more appealing habitat area. To evaluate the habitat suitability in and around the LPRSA, a spatial habitat analysis (Appendix I) was conducted and indicated insufficient riparian tree and shrub cover in the LPRSA to provide the habitat necessary for a breeding population. No current reports, either anecdotal or from surveys, were found of river otter in the LPRSA. Although the presence of these two mammalian species in the LPRSA is either undocumented (river otter) or limited (mink), this mammalian assessment is presented as an evaluation of potential present and possible future use of the LPRSA by aquatic mammals.

The potential for risk to mammalian species was characterized using a dietary LOE, whereby the estimated dietary doses were compared to dietary TRVs for COPECs identified in the SLERA. COPECs with calculated HQs ≥ 1.0 based on the LOAEL TRVs were identified as preliminary COCs.

The mammal risk assessment process is outlined in Table 9-1. Section 9.1 presents the dietary assessments; uncertainties associated with various components of the dietary assessment are discussed throughout Section 9.1 and summarized at the end of this section. Section 9.2 identifies the preliminary COCs, which are further evaluated in Section 13.

Table 9-1. Outline of the mammal risk assessment

Section No.	Section Title	Section Contents
9.1	Dietary Assessment	presents COPECs based on the SLERA, exposure and effects data, HQs, uncertainty discussion, and summary of risk characterization
9.2	Identification of preliminary COCs	identifies preliminary COCs

COC – chemical of concern
COPEC – chemical of potential ecological concern
FS – feasibility study
HQ – hazard quotient
SLERA – screening-level ecological risk assessment

9.1 DIETARY ASSESSMENT

This dietary assessment was conducted for the two mammal species selected for evaluation (river otter and mink), consistent with the USEPA-approved PFD (Windward and AECOM 2009). For mink and river otter, the assessment was conducted for the COPECs that were identified in the SLERA (see Section 5). This section summarizes the COPECs, describes how exposure and effects concentrations were derived, presents the HQs, and summarizes the uncertainties associated with the dietary assessment.

9.1.1 COPECs

The COPECs for mink and river otter were identified using a risk-based screening process conducted in the SLERA, wherein doses based on maximum concentrations were compared to dietary screening-level TRVs (Table 9-2; Appendix A); these comparisons are summarized in Section 5. The COPECs identified for mammals are presented in Table 9-2.

Table 9-2. Mammal dietary COPECs

COPEC	
Metals	
Arsenic	Nickel
Cadmium	Selenium
Copper	Vanadium
Lead	Zinc
Methylmercury/mercury	
PAHs	
Total HPAHs	
PCBs	
Total PCBs	PCB TEQ - mammal
PCDDs/PCDFs	
PCDD/PCDF TEQ -mammal	Total TEQ -mammal
Organochlorine Pesticides	
Dieldrin	

Note: COPECs are those COIs for which the maximum modeled dietary dose exceeded its TSV. If a TSV was exceeded based on either mink or river otter, it was retained as a COPEC for all mammals.

COI – chemical of interest
COPEC – chemical of potential ecological concern
HPAH – high-molecular weight polycyclic aromatic hydrocarbon
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
PCDD – polychlorinated dibenzo-*p*-dioxin
PCDF – polychlorinated dibenzofuran
TEQ – toxic equivalent
TSV – toxicity screening value

A number of COIs could not be screened as part of the SLERA (Appendix A) because no mammal dietary screening levels were available. These COIs are presented in

Section 5.2.2, along with a discussion of the implications of not being able to evaluate these COIs.

9.1.2 Exposure

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

9.1.2.1 Methods

Dietary doses for mammals were estimated based on ingestion of biota (i.e., prey), incidental ingestion of sediment, and ingestion of surface water. Dietary doses were estimated as milligrams of each COPEC ingested per kilogram of body weight per day (mg/kg bw/day) using Equation 8-1 and the methods presented in Section 8.1.2.1. The body weights, ingestion rates, and SUFs for mink and river otter are described in Section 9.1.2.2. The DFs assumed for mink and river otter and the assumptions used to derive them are described in Section 9.1.2.3. Both mink and river otter were conservatively assumed to use the LPRSA for the entire year (i.e., no adjustment was made for seasonal site use).

9.1.2.2 Body weights, ingestion rates, and SUFs

Average body weights and average ingestion rates for sediment, water, and food for use in the dietary dose calculations were obtained from the literature, as summarized in Table 9-3. The exposure parameters were selected as follows:

- Body weights for mammals were based on average male and female body weights reported in USEPA's *Wildlife Exposure Factors Handbook* (USEPA 1993). The effect on HQs of using these maximum and minimum male and female body weights is evaluated in the uncertainty section (Section 9.1.4.2).
- FIRs were based on the measured ingestion rate for mink and river otter (or similar species, when available) and were expressed on a wet weight basis. FIRs were not available for river otter or a similar species, so an allometric equation for non-herbivorous mammals was used to estimate ingestion rates (Nagy 1987). Marine mammals are known to have higher metabolic rates than related terrestrial mammals, leading to higher FIRs. Therefore, river otter HQs are also calculated using a range of FIRs in the risk characterization uncertainty section (Section 9.1.4.2). This uncertainty is also addressed in a sensitivity model in Appendix H.
- Best professional judgment was used to estimate incidental SIRs for mammals because species-specific or appropriate surrogate data were unavailable from the literature. Incidental SIRs were expressed on a dry weight basis, as a percentage of the dry weight FIR. The effect on HQs of varying the SIR is evaluated in the uncertainty section (Section 9.1.4.2).

- Dry weight FIRs (required for deriving incidental SIRs) were derived from wet weight ingestion rates assuming 80% moisture in prey (based on average moisture contents of 72, 79, and 88% for fish, invertebrates, and worms from the LPRSA, respectively) when dry weight FIRs were not available from the literature.
- WIRs for mammals were based on an allometric equation from Calder and Braun (1983), as cited in USEPA (1993).
- A SUF of 1 was used for both mink and river otter, based on the assumption that they obtain 100% of their diet from their preferential foraging (i.e., exposure) areas in the LPRSA. It is possible that mink or river otter forage outside of the LPRSA, and therefore use the LPRSA as their foraging (i.e., exposure) area less than 100% of the time. The exposure dose, and thus the HQ, is directly proportional to the SUF (Equation 8-1); if a species uses the LPRSA 50% of the time, the HQ will decrease by 50%. The use of an SUF of 1 provides conservative estimates of the potential risks in these cases. The effect on the HQs of varying the SUF is addressed in Section 9.1.4.2.

Table 9-3. Exposure parameter values for mink and river otter

		Food Ingestion		Incidental Sediment Ingestion			
Species	BW (kg)[a]	FIR (kg ww/day)	Source	SI (%)[b]	SIR (kg dw/day)	Source	WIR (L/day)[c]
Mink	1.0	0.14[d]	Bleavins and Aulerich (1981)	2	0.00056	no empirical data available; based on feeding habits and best professional judgment	0.099
River otter	8.0	1.3[e]	USEPA (1993); Nagy (1987)	2	0.0052	no empirical data available; based on feeding habits and best professional judgment	0.64

a Average of male and female adult body weights reported in USEPA (1993).

b Based on percentage of the dry diet that is incidentally ingested sediment. Dry weight FIRs estimated from wet weight FIRs assuming 80% moisture in the diet.

c WIR based on Calder and Braun (1983), in which mammal WIR = 0.099 x $BW^{0.90}$, and body weight is expressed in kilograms.

d FIR = 14% of body weight on average (range reported was 12 to 16%).

e The FIR was calculated based on the FMR for river otter using the following equation: FIR (kg ww/day) = FMR/ME * (0.001 g/kg) (USEPA 1993). The FMR (kcal/day) was calculated as 0.6167 x $BW^{0.862}$, wherein body weight is expressed in grams (based on the equation for non-herbivorous mammals from Nagy (1987). The ME (kcal/g ww) was calculated as GE (kcal/g ww) x AE (unitless) for the river otter prey. The ME for river otter of 1.1 kcal/g ww was calculated assuming a diet of 85% fish (GE =1.2 kcal/g ww and AE = 0.89) and 15% crab (GE = 1 kcal/g ww and AE = 0.86). This approach is analogous to the approach used for river otter as part of the Great Lakes Water Quality Initiative (USEPA 1995b).

AE – assimilation efficiency
BW or bw – body weight
dw – dry weight
FIR – food ingestion rate
FMR – field metabolic rate
GE – gross energy
ME – metabolizable energy
SI – sediment ingestion
SIR – sediment ingestion rate
USEPA – US Environmental Protection Agency
WIR – water ingestion rate
ww – wet weight

9.1.2.3 Prey composition and exposure areas

This section presents the prey composition and exposure areas that were selected for mink and river otter.

For the dietary dose equation (Equation 8-1), prey ingested by mink and river otter included only those prey types for which tissue chemistry data from the LPRSA were available. These tissue data include whole blue crab and whole fish.

The mink diet is composed of a relatively large percentage of terrestrial prey, so much of their diet is from outside the LPRSA. However, because there is some uncertainty in the amount of their diet that is terrestrial, dietary doses were conservatively calculated in two general ways: 1) assuming that terrestrial prey make up approximately half of the diet (with the terrestrial portion set equal to zero because terrestrial concentrations are not available), and 2) assuming that the diet is composed of all aquatic prey (Table 9-4).

Table 9-4. Prey composition used to estimate dietary dose for mink and river otter

Species	Percentage of Prey Type in Diet			
		Fish Size Range		
	Blue Crab	0–30 cm	> 30 cm	Terrestrial Prey
Mink				
Scenario 1 – aquatic/terrestrial prey	16.5	34	0	49.5[a]
Scenario 2 – aquatic/terrestrial prey	16.5	31	3	49.5[a]
Scenario 3 – aquatic prey only	16.5	83.5	0	0
Scenario 4 – aquatic prey only	16.5	80.5	3	0
Scenario 5 – aquatic prey only	33.5	63.5	3	0
River Otter				
Scenario 1	15	85	0	0
Scenario 2	15	80	5	0

[a] For the aquatic and terrestrial prey evaluation for mink, the portion of terrestrial prey concentrations of the diet was assumed to be negligible (i.e., COPEC concentration set equal to zero).

COPEC – chemical of potential ecological concern

Two exposure areas (i.e., RM ≥ 10 and site wide) were used for mink and river otter. For reasons presented in the sections below, it was assumed that mink and river otter could potentially use only areas of the LPRSA at and upstream of RM 10. Therefore, for mink and river otter, fish and sediment EPCs were derived using only data from areas at and upstream of RM 10 (Tables 9-5 and 9-6), with the exception of data for non-small forage fish (NFF). Because crab and NFF potentially move throughout the LPRSA, site-wide data were used for these prey. In addition, site-wide data were used to derive EPCs for sediment, crab, SFF, and NFF. For exposure areas from which surface water would be consumed, it was assumed that only freshwater (i.e., water at and upstream of RM 4) would be ingested by mink or river otter. The rationale for the prey composition and exposure areas is presented in more detail for mink and river otter in the remainder of this section.

Table 9-5. LPRSA exposure areas for mink and river otter

Species	Exposure Scenario	Exposure Area		
		Prey	Sediment	Surface Water
Mink/river otter	RM ≥10	RM ≥10 for SFF; site wide for NFF and blue crab	RM ≥10	RM ≥10
	site wide	site wide for all prey	site wide	RM ≥ 4

LPRSA – Lower Passaic River Study Area
NFF – non-small forage fish
RM – river mile
SFF – small forage fish

Table 9-6. Source of EPCs for mink and river otter

Species	Prey Type[a]						Surface Water	Sediment	Rationale
	Blue Crab		Fish ≤ 30 cm[b]		Fish > 30 cm[b]				
	% in Diet	Exposure Area	% in Diet	Exposure Area	% in Diet	Exposure Area			
Mink – RM ≥ 10									
Scenario 1 – aquatic/ terrestrial prey[c]	16.5	site wide[d]	34	NFF site wide, SFF ≥ RM 10[e]	0	site wide	RM ≥ 10	RM ≥ 10	Evaluate risk using mink diet based on fish size class expected to make up the majority of their diet; a diet that is approximately 50% terrestrial prey (concentrations in which are assumed to be negligible); and an exposure area RM ≥ 10 (which has more vegetation than < RM 10), which could provide mink habitat.
Scenario 2 – aquatic/ terrestrial prey[c]	16.5	site wide[d]	31	NFF site wide, SFF ≥ RM 10[e]	3[f]	site wide	RM ≥ 10	RM ≥ 10	Evaluate risk using mink diet with a range of fish size classes; range to include large fish, including common carp; a diet that is approximately 50% terrestrial prey (concentrations in which are assumed to be negligible);and an exposure area RM ≥ 10 (which has more vegetation than < RM 10), which could provide mink habitat.
Scenario 3 – aquatic prey only	16.5	site wide[d]	83.5	NFF site wide, SFF RM ≥10[e]	0	site wide	RM ≥ 10	RM ≥ 10	Evaluate risk using mink diet based on fish size class expected to make up the majority of their diet; a diet that includes all aquatic prey (a conservative assumption); and an exposure area RM ≥ 10 (which has more vegetation than RM <10), which could provide mink habitat.

Species	Prey Type[a]						Surface Water	Sediment	Rationale
	Blue Crab		Fish ≤ 30 cm[b]		Fish > 30 cm[b]				
	% in Diet	Exposure Area	% in Diet	Exposure Area	% in Diet	Exposure Area			
Scenario 4 – aquatic prey only	16.5	site wide[d]	80.5	NFF site wide, SFF RM ≥10[e]	3[f]	site wide	RM ≥ 10	RM ≥ 10	Evaluate risk using mink diet with a range of fish size classes (range to include large fish, including common carp); a diet that includes all aquatic prey (a conservative assumption); and an exposure area RM ≥10 (which has more vegetation than RM < 10), which could provide mink habitat.
Scenario 5 – aquatic prey only[g]	33.5	site wide[d]	63.5	NFF site wide, SFF RM ≥10[e]	3[f]	site wide	RM ≥10	RM ≥ 10	Evaluate risk using mink diet of 33.5% blue crab and 66.5% fish (all aquatic prey; a conservative assumption); including larger fish as part of the diet; and an exposure area RM ≥ 10 (which has more vegetation than RM < 10), which could provide mink habitat.
Mink – Site wide									
Scenario 1 – aquatic/ terrestrial prey	16.5	site wide[d]	34	site wide	0	site wide	RM ≥ 4[g]	site wide	Evaluate risk using mink diet based on fish size class expected to make up the majority of their diet, and a diet that is approximately 50% terrestrial prey (concentrations in which are assumed to be negligible and a site-wide exposure area.
Scenario 2 – aquatic/ terrestrial prey	16.5	site wide[d]	31	site wide	3[f]	site wide	RM ≥ 4[g]	site wide	Evaluate risk using mink diet with a range of fish size classes; range to include large fish, including common carp; a diet that is approximately 50% terrestrial prey (concentrations in which are assumed to be negligible); and a site-wide exposure area.

Species	Prey Type[a]						Surface Water	Sediment	Rationale
	Blue Crab		Fish ≤ 30 cm[b]		Fish > 30 cm[b]				
	% in Diet	Exposure Area	% in Diet	Exposure Area	% in Diet	Exposure Area			
Scenario 3 – aquatic prey only	16.5	site wide[d]	83.5	site wide	0	site wide	RM ≥ 4[g]	site wide	Evaluate risk using mink diet based on fish size class expected to make up the majority of their diet; a diet that includes all aquatic prey (a conservative assumption); and a site-wide exposure area.
Scenario 4 – aquatic prey only	16.5	site wide[d]	80.5	site wide	3[f]	site wide	RM ≥ 4[g]	site wide	Evaluate risk using mink diet with a range of fish size classes (range to include large fish, including common carp); a diet composed of all aquatic prey (a conservative assumption); and a site-wide exposure area.
Scenario 5 – aquatic prey only	33.5	site wide[d]	63.5	site wide	3[f]	site wide	RM ≥ 4[g]	site wide	Evaluate risk using mink diet of 33.5% blue crab and 66.5% fish (all aquatic prey; a conservative assumption); larger fish as part of the diet; and a site-wide exposure area.
River Otter – RM ≥ 10									
Scenario 1	15	site wide[d]	85	NFF site wide, SFF RM ≥10[e]	0	site wide	RM ≥ 10	RM ≥ 10	Evaluate river otter diet based on fish size class expected to make up the majority of their diet and an exposure area RM ≥ 10 (which has more vegetation than RM < 10), which could provide river otter habitat.
Scenario 2	15	site wide[d]	80	NFF site wide, SFF RM ≥10[e]	5[f]	site wide	RM ≥ 10	RM ≥ 10	Evaluate risk using river otter diet with a range of fish size classes (range to include large fish, including common carp) and an exposure area RM ≥ 10 (which has more vegetation than RM < 10), which could provide river otter habitat.

Species	Prey Type[a]						Surface Water	Sediment	Rationale
	Blue Crab		Fish ≤ 30 cm[b]		Fish > 30 cm[b]				
	% in Diet	Exposure Area	% in Diet	Exposure Area	% in Diet	Exposure Area			
River Otter – Site wide									
Scenario 1	15	site wide[d]	85	site wide	0	site wide	RM ≥ 4[g]	site wide	Evaluate river otter diet based on fish size class expected to make up the majority of their diet and a site-wide exposure area.
Scenario 2	15	site wide[d]	80	site wide	5[f]	site wide	RM ≥ 4[g]	site wide	Evaluate risk using river otter diet with a range of fish size classes (range to include large fish, including common carp) and a site wide exposure area.

Note: If fewer than six samples were available for calculating a UCL, the maximum concentration was used.

a Includes whole-body tissue.

b For composite samples, length is based on the maximum length of any fish in the sample.

c For the aquatic and terrestrial prey evaluation for mink, the portion of terrestrial prey (i.e., concentrations of the diet) was assumed to be negligible (i.e., COPEC concentration set equal to zero).

d Includes all available blue crab data (i.e., RM 1 to RM 10).

e SFF include mummichog, gizzard shad, mixed forage fish, pumpkinseed, spottail shiner, and silver shiner that were ≤ 20 cm in length. NFF are all other fish that are not considered SFF.

f A larger percentage of fish > 30 cm in the diet was evaluated in the uncertainty section (Section 9.1.4.2).

g Includes only freshwater.

COPEC – chemical of potential ecological concern
EPC – exposure point concentration
NFF – non-small forage fish
RM – river mile
SFF – small forage fish
UCL – upper confidence limit on the mean

Mink

Prey Composition

Mink are carnivores and their diet is influenced by location, habitat, season, and prey availability (Burgess 1978). Mink are opportunistic feeders, eating fish, crayfish, waterfowl, amphibians, aquatic invertebrates, and mammals. A number of mink diet studies from a variety of geographic locations indicate that the percentage of aquatic prey (i.e., fish, benthic invertebrates, and aquatic insects) in the diet generally ranges from about 25 to 40%, although it may be as low as 8% or as high as 89% (Wise et al. 1981; Ferreras and MacDonald 1999; Hamilton 1940; Korschgen 1958; Salo et al. 2010; Sealander 1943; Burgess 1978; Ward et al. 1986; Alexander 1977). For the Hudson River ERA, based on dietary studies[126] for New York State populations of mink, sources of prey for mink were estimated to be 34% fish, 16.5% aquatic invertebrates, and 49.5% terrestrial organisms (USEPA 2000; TAMS and Menzie-Cura 2000). Although a range in prey portions was found based on the scientific literature reviewed above, in general, prey portions were consistent with those used for the Hudson River ERA. These prey portions of 34% fish, 16.5% aquatic invertebrates, and 49.5% terrestrial organisms were also selected (mink diet Scenarios 1 and 2, Table 9-4). COPEC concentrations in terrestrial prey were not available; therefore, the terrestrial portion of the diet was set equal to zero (mink diet Scenarios 1 and 2, Table 9-4) in dietary dose calculations, assuming no exposure from terrestrial prey for approximately one-half of the mink diet. In addition, it was assumed that mink feed exclusively on LPRSA aquatic prey, a very conservative assumption. For dietary dose calculations, the aquatic invertebrate portion remained at 16.5%, while the fish portion of the diet was adjusted to 83.5% (mink diet Scenarios 3 and 4, Table 9-4). Prey portions of 33.5% for aquatic invertebrates and 66.5% for fish were also used (mink diet Scenario 5, Table 9-4).

Mink generally prefer fish that are ≤ 30 cm in length. The results of an analysis of scat from mink in Idaho showed that the mink diet consisted of fish ranging in length from 7 to 30 cm; neither largescale sucker nor northern squawfish, which range from 35 to 45 cm, were consumed (Melquist et al. 1981). Another study in Great Britain found that most fish consumed by mink were < 30 cm long, although some of the Northern pike consumed were up to 70 cm in length. However, Northern pike were only 1.5% of the mink's diet, so the overall percentage of large fish consumed was small. Therefore, the mink diet includes fish ≤ 30 cm (i.e., mink diet Scenarios 1 and 3, Table 9-4).

Fish ≤ 30 and > 30 cm in length were included in the mink diet (i.e., mink diet Scenarios 2, 4, and 5, Table 9-4 (USEPA 2015b, c, 2016g). The percentage of large fish is assumed to represent carrion consumed opportunistically, based on assumptions made by Wise et al. (1981), and typical body lengths of fish consumed by mink (Melquist et al. 1981; Sealander 1943; Wise et al. 1981). Guilday 1949, as cited by Pendleton (1982),

[126] With the exception of Hamilton (1940), the New York State studies cited in the Hudson River ERA were not found in the scientific literature and therefore were not reviewed.

provides a frequency of carrion in mink scats of 3%. This frequency has been applied as the dietary percentage. The percentage of fish > 30 cm in length in the mink diet is evaluated in the uncertainty evaluation (Section 9.1.4.2) and the sensitivity analysis (Appendix H).

Exposure Areas

Two exposure areas (i.e., at and upstream of RM 10 and site wide) were used for mink (Table 9-5). Mink are generally limited to natural shorelines with access to water (Allen 1986) and will dive for prey at depths of less than 10 ft (Harrington et al. 2012; Hays et al. 2007). Mink prefer areas with dense riparian or shrub-scrub vegetation with canopy and tend to avoid areas near human activity or limited vegetation, including areas of residential/recreational land use (Allen 1986; USEPA 2002b). This information suggests that mink are more likely to be restricted to the least disturbed/developed portions of the LPRSA (i.e., from about RM 10 and upstream). Mink habitat preference is also highly influenced by prey accessibility (Burgess 1978).

Between RM 8.5 and RM 9.5 there are some areas with vegetation, but not likely enough upland with sufficient cover away from the shoreline to serve as mink habitat. At and upstream of RM 10, there is more vegetation that could provide mink habitat. Therefore, it was assumed that mink could potentially use only areas of the LPRSA at RM 10 and upstream. As a result, SFF (which tend to be localized in their movements) used in the dietary dose calculations were limited to those found at RM 10 and upstream. Other fish are less localized in their movements than are SFF and, as such, prey tissue data for NFF include those caught throughout the LPRSA.

Site-wide tissue and sediment data were also used for the dietary dose calculations for mink (USEPA 2015b, c, 2016g). Using site-wide data assumes that mink could potentially use the entire LPRSA; however, there is a high degree of uncertainty using site-wide exposure area for mink because of the lack of mink-suitable habitat downstream of RM 10.

River Otter

Prey Composition

River otters occupy the upper trophic level of the food chain, and their diet consists primarily of fish, although they are opportunistic feeders known to eat crayfish, amphibians, aquatic invertebrates, birds, reptiles, and mammals (Knudsen and Hale 1968; Toweill 1974; Melquist and Hornocker 1983). The amount of fish and other aquatic animals that river otters consume depends on availability and abundance, size class of predator and prey, and swimming ability (Melquist and Hornocker 1983). Boyle (2006) reports that northern river otters do not appear to be selective when fishing and generally take the most available fish—usually slower species such as suckers, common carp, and catfish. Sheldon and Toll (1964) found that availability also was affected by the time of day the otters fed, fish spawning periods, fishing methods, and the effects of ice in winter.

Through scat analysis, which may underestimate the portion of shellfish soft tissue consumed by river otters, Melquist and Hornocker (1983) found that river otters living in Idaho fed primarily on fish, followed by invertebrates, birds, mammals, and reptiles. Sheldon and Toll (1964) analyzed scat in Massachusetts and identified the primary prey as centrarchids, yellow perch, white suckers, golden shiners, and crayfish. The chief prey items found in the stomach contents of river otters in eastern Arkansas were centrarchidae (primarily sunfishes), catostomidae (i.e., suckers), clupeidae (primarily gizzard shad), and crayfish (Tumlison et al. 1986). The same study found the following proportions of prey ingested by river otters: 71% fish, 18% crayfish, and 11% amphibians, reptiles, birds, insects, mammals, and mollusks. Larsen (1984) reported the following proportions of prey ingested by river otters in southeastern Alaska: 86% fish, 10% crabs, and 4% invertebrates other than crabs, birds, and mammals and plant material. In a study conducted in Great Britain, Wise et al. (1981) reported that the river otter's diet consisted of 93% fish and 1% aquatic invertebrates. Based on information from the literature, which indicates dietary ranges of 71 to 93% fish and 1 to 18% aquatic invertebrates, it was assumed that river otter in the LPRSA consume 85% fish and 15% aquatic invertebrates (Table 9-4).

River otters generally prey on fish that range from 2 to 50 cm in length; average prey length for an adult river otter is around 30 cm (USEPA 2003d; Melquist et al. 1981). For the dietary dose calculations, the diet of river otter was limited to fish ≤ 30 cm (otter diet Scenario 1, Table 9-4). However, river otter from some locations have been observed to consume fish up to 50 cm in length (Tumlison et al. 1986; Wise et al. 1981; Melquist et al. 1981) and may infrequently consume fish up to 70 cm in length. A percentage (i.e., 5%) of > 30-cm-long fish in the diet was also evaluated (otter diet Scenario 2, Table 9-4) (USEPA 2015b, c, 2016g).The percentage of > 30-cm-long fish in the river otter diet is evaluated in the uncertainty evaluation (Section 9.1.4.2) and the sensitivity analysis (Appendix H).

Exposure Areas

Two exposure areas (i.e., at and upstream of RM 10 and site wide, Table 9-5) were used for river otter. The literature suggests that river otter are generally limited to natural shorelines with access to water and will dive as deeply as approximately 60 ft for prey (USEPA 2003d). Feeding can occur in deeper water because larger prey (i.e., prey that are easily preyed upon by otter) selectively avoid shallow water (Cote et al. 2008). River otter have a high preference for shoreline areas with riparian vegetation and a low preference for areas with shrub-scrub or limited emergent vegetation (Boyle 2006; USEPA 2003d; Melquist and Hornocker 1983). The literature indicates that river otter generally will not use areas with human activity, including areas of residential and recreational land use (Boyle 2006; USEPA 2003d; Melquist and Hornocker 1983). This information suggests that river otter are more likely to be restricted to the least disturbed/developed portions of the LPRSA. Therefore, it was assumed that river otter could potentially use only areas at and upstream of RM 10 of the LPRSA. Accordingly, SFF (which tend to be localized in their movements) used in the dietary dose calculations were limited to those found at RM 10 and upstream. Prey tissue data for other fish less localized in their movements (i.e., NFF) include data for NFF caught throughout the LPRSA.

As a conservative evaluation, it was also assumed that river otter could potentially use the entire LPRSA, and therefore, site-wide tissue and sediment data were used in the dietary dose calculations. However, there is a high degree of uncertainty in using a site-wide exposure area for river otter because of the lack of river otter-suitable habitat present below RM 10. Further compounding the uncertainty is the fact that while these scenarios are based on the underlying assumption that river otter use the LPRSA, there is no available information documenting their presence in the LPR.

9.1.2.4 Exposure point concentrations

EPCs were calculated for each of the media types ingested (i.e., prey, sediment, and surface water) by mink and river otter for use in Equation 8-1 (Section 8.1.2.1) to calculate dietary doses. Fish and crab EPCs were calculated as presented in Table 9-6. For crab, the EPC was equal to the UCL calculated with all available blue crab data from throughout the LPRSA (RM 0 to RM 10), because of the large home range of crab. For fish, sediment, and surface water, the EPCs were equal to the UCLs calculated with all data from the relevant exposure area (Table 9-6). UCL concentrations were calculated using USEPA's ProUCL® statistical package (Version 5.0.00) (USEPA 2013d), as described in Section 4.3.7.[127] For each dataset with fewer than six samples, a UCL was not calculated; instead, the maximum concentration was used as the EPC.

[127] The UCL recommended by USEPA's ProUCL® software (typically the 95% UCL, but in some cases the 97.5% or even the 99% UCL) was used.

The UCLs used for calculating EPCs are presented in Table 9-7 for fish prey, Table 9-8 for blue crab prey, Table 9-9 for sediment, and Table 9-10 for surface water; all UCLs are also presented in Appendix C1. Uncertainties associated with the use of non-detects in calculations of total PCBs and TEQs - mammal are discussed in Section 9.1.4.2.

Table 9-7. COPEC summary statistics for LPRSA fish samples

COPEC	Unit (ww)	No. Detects/ No. Samples	% Detected	Concentration			
				Min. Detected	Max. Detected	Mean Detected	UCL
SFF RM ≥ 10 and ≤ 30 cm NFF Site Wide							
Metals							
Arsenic	mg/kg	38/42	90.5	0.031	0.28	0.17	0.18
Cadmium	mg/kg	42/42	100	0.003	0.19	0.024	0.045
Copper	mg/kg	42/42	100	0.4	50.9	5.5	7.5
Lead	mg/kg	42/42	100	0.052	3.2	0.76	1.3
Mercury	µg/kg	42/42	100	30	310	120	130
Methylmercury	µg/kg	42/42	100	14	330	110	130
Nickel	mg/kg	42/42	100	0.18	89.1	5	14
Selenium	mg/kg	42/42	100	0.22	3	0.86	1
Vanadium	mg/kg	40/42	95.2	0.016	1.2	0.25	0.45
Zinc	mg/kg	42/42	100	15	48	26	28
PAHs							
Benzo(a)pyrene	µg/kg	24/42	57.1	1.5	64	16	21
Total HPAHs	µg/kg	42/42	100	9.4	860	160	210
PCBs							
Total PCBs	µg/kg	42/42	100	170	5,100	1,400	2,100
PCB TEQ-mammal	ng/kg	42/42	100	2.1	41	15	19
PCDDs/PCDFs							
PCDD/PCDF TEQ-mammal	ng/kg	42/42	100	0.81	260	86	140
Total TEQ-mammal	ng/kg	42/42	100	5.5	300	100	160
Organochlorine Pesticides							
Dieldrin	µg/kg	42/42	100	7.8	110	27	32
Site Wide, ≤ 30 cm							
Metals							
Arsenic	mg/kg	58/62	93.5	0.031	0.52	0.24	0.26
Cadmium	mg/kg	62/62	100	0.003	0.19	0.026	0.033
Copper	mg/kg	62/62	100	0.4	50.9	4.7	6.1
Lead	mg/kg	62/62	100	0.052	4.9	0.99	1.6
Mercury	µg/kg	62/62	100	30	310	99	130
Methylmercury	µg/kg	62/62	100	14	330	90	110
Nickel	mg/kg	62/62	100	0.18	89.1	4.6	5.4
Selenium	mg/kg	62/62	100	0.22	3	0.8	0.9
Vanadium	mg/kg	60/62	96.8	0.016	1.3	0.35	0.54
Zinc	mg/kg	62/62	100	15	51.6	31	33

Table 9-7. COPEC summary statistics for LPRSA fish samples

COPEC	Unit (ww)	No. Detects/ No. Samples	% Detected	Concentration: Min. Detected	Max. Detected	Mean Detected	UCL
PAHs							
Benzo(a)pyrene	µg/kg	44/62	71	1.5	98	18	24
Total HPAHs	µg/kg	62/62	100	9.4	1000	170	250
PCBs							
Total PCBs	µg/kg	62/62	100	170	5,100	1,100	1,700
PCB TEQ-mammal	ng/kg	62/62	100	2.1	41	13	18
PCDDs/PCDFs							
PCDD/PCDF TEQ-mammal	ng/kg	62/62	100	0.81	260	72	91
Total TEQ-mammal	ng/kg	62/62	100	5.5	300	85	130
Organochlorine Pesticides							
Dieldrin	µg/kg	62/62	100	3.5	110	21	24
<u>Site Wide, > 30 cm</u>							
Metals							
Arsenic	mg/kg	66/69	95.7	0.017	0.61	0.1	0.15
Cadmium	mg/kg	69/69	100	0.0034	0.27	0.022	0.042
Copper	mg/kg	69/69	100	0.31	29	1.3	3.1
Lead	mg/kg	62/62	100	0.02	2.2	0.45	0.53
Mercury	µg/kg	69/69	100	32	680	180	210
Methylmercury	µg/kg	69/69	100	30	530	170	200
Nickel	mg/kg	69/69	100	0.11	4.2	0.52	0.85
Selenium	mg/kg	69/69	100	0.16	1.4	0.51	0.57
Vanadium	mg/kg	57/69	82.6	0.017	0.21	0.092	0.093
Zinc	mg/kg	69/69	100	12	104	29	39
PAHs							
Benzo(a)pyrene	µg/kg	26/69	37.7	0.55	24	2.7	1.9
Total HPAHs	µg/kg	69/69	100	3.1	310	45	53
PCBs							
Total PCBs	µg/kg	69/69	100	350	7900	2400	2800
PCB TEQ-mammal	ng/kg	69/69	100	2.7	240	30	36
PCDDs/PCDFs							
PCDD/PCDF TEQ-mammal	ng/kg	69/69	100	1.8	1400	150	190
Total TEQ-mammal	ng/kg	69/69	100	7	1500	180	230
Organochlorine Pesticides							
Dieldrin	µg/kg	69/69	100	7.2	88	34	38

Note: The UCL was selected as the EPC.

COPEC – chemical of potential ecological concern
EPC – exposure point concentration
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
LPRSA – Lower Passaic River Study Area
PCDD – polychlorinated dibenzo-*p*-dioxin
PCDF – polychlorinated dibenzofuran
RM – river mile
SFF – small forage fish

NFF – non-small forage fish
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
TEQ – toxic equivalent
UCL – upper confidence limit on the mean
ww – wet weight

Table 9-8. COPEC summary statistics for LPRSA blue crab samples

COPEC	Unit (ww)	No. Detects/ No. Samples	% Detected	Concentration			
				Min. Detected	Max. Detected	Mean Detected	UCL
Metals							
Arsenic	mg/kg	24/24	100	0.29	1.9	1	1.4
Cadmium	mg/kg	24/24	100	0.047	0.18	0.095	0.11
Copper	mg/kg	24/24	100	16.2	30.6	23	24.6
Lead	mg/kg	24/24	100	0.2	0.66	0.32	0.36
Mercury	µg/kg	24/24	100	79	190	130	140
Methylmercury	µg/kg	24/24	100	55	170	110	120
Nickel	mg/kg	24/24	100	0.52	1.9	0.89	1
Selenium	mg/kg	24/24	100	0.43	1.1	0.73	0.79
Vanadium	mg/kg	24/24	100	0.078	0.14	0.11	0.12
Zinc	mg/kg	24/24	100	28.3	41.1	35.3	36.4
PAHs							
Benzo(a)pyrene	µg/kg	24/24	100	1.9	40	7.4	10
Total HPAHs	µg/kg	24/24	100	14	350	82	110
PCBs							
Total PCBs	µg/kg	24/24	100	150	580	320	350
PCB TEQ-mammal	ng/kg	24/24	100	3	12	8	8.8
PCDDs/PCDFs							
PCDD/PCDF TEQ-mammal	ng/kg	24/24	100	26	93	56	61
Total TEQ-mammal	ng/kg	24/24	100	28	100	63	70
Organochlorine Pesticides							
Dieldrin	µg/kg	24/24	100	2.4	9.1	6.2	6.8

Note: The UCL was selected as the EPC. Data are for LPRSA blue crab whole-body tissue data from RM 0 to RM 10, an area assumed to be representative of site-wide exposure due to the large home range of crab.

COPEC – chemical of potential ecological concern
EPC – exposure point concentration
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
LPRSA – Lower Passaic River Study Area
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
PCDD – polychlorinated dibenzo-*p*-dioxin
PCDF – polychlorinated dibenzofuran
RM – river mile
TEQ – toxic equivalent
UCL – upper confidence limit on the mean
ww – wet weight

Table 9-9. COPEC summary statistics for LPRSA sediment samples

COPEC	Unit (dw)	No. Detects/ No. Samples	% Detected	Concentration			
				Min. Detected	Max. Detected	Mean Detected	UCL
RM ≥ 10							
Metals							
Arsenic	mg/kg	173/173	100	0.48	68.6	5.2	7.8
Cadmium	mg/kg	173/173	100	0.056	35.4	3.2	4.9
Copper	mg/kg	199/199	100	4.19	778	120	140
Lead	mg/kg	170/170	100	3.94	2050	200	280
Mercury	µg/kg	176/176	100	16.1	22,200	1,800	2,900
Methylmercury	µg/kg	47/48	97.9	0.035	4.38	1.1	1.8
Nickel	mg/kg	197/197	100	4.6	200	20	31
Selenium	mg/kg	131/173	75.7	0.038	3.3	0.7	0.71
Vanadium	mg/kg	173/173	100	3.99	91.6	20	22
Zinc	mg/kg	173/173	100	23.5	2,000	400	420
PAHs							
Benzo(a)pyrene	µg/kg	173/174	99.4	1.62	41,000	3,700	5,300
Total HPAHs	µg/kg	174/174	100	6.2	510,000	35,000	54,000
PCBs							
Total PCBs	µg/kg	176/176	100	1.34	23,800	1500	2,400
PCB TEQ-mammal	ng/kg	176/176	100	0.101	265	19	27
PCDDs/PCDFs							
PCDD/PCDF TEQ-mammal	ng/kg	176/176	100	0.553	51,400	1,100	2,200
Total TEQ-mammal	ng/kg	176/176	100	0.62	51,500	1,100	1,900
Organochlorine pesticides							
Dieldrin	µg/kg	169/175	96.6	0.107	88	4.8	7.5
Site Wide							
Metals							
Arsenic	mg/kg	426/426	100	0.48	118	7.6	9.6
Cadmium	mg/kg	426/426	100	0.053	46.6	3.7	4.9
Copper	mg/kg	503/503	100	4.19	930	100	170
Lead	mg/kg	422/422	100	3.94	2,050	200	270
Mercury	µg/kg	429/429	100	16.1	24,300	2,200	2,900
Methylmercury	µg/kg	136/137	99.3	0.035	23	2.9	3.9
Nickel	mg/kg	501/501	100	4.15	200	30	32
Selenium	mg/kg	341/406	84	0.038	5.2	1	0.93
Vanadium	mg/kg	426/426	100	3.99	110	24	27
Zinc	mg/kg	426/426	100	23.5	2,000	400	490
PAHs							
Benzo(a)pyrene	µg/kg	426/427	99.8	1.62	41,000	3800	4,700
Total HPAHs	µg/kg	427/427	100	6.2	510,000	36,000	46,000

Table 9-9. COPEC summary statistics for LPRSA sediment samples

COPEC	Unit (dw)	No. Detects/ No. Samples	% Detected	Concentration			
				Min. Detected	Max. Detected	Mean Detected	UCL
PCBs							
Total PCBs	µg/kg	429/429	100	1.34	28,600	1,800	2,600
PCB TEQ-mammal	ng/kg	429/429	100	0.000729	267	22	30
PCDDs/PCDFs							
PCDD/PCDF TEQ-mammal	ng/kg	428/428	100	0.0567	51,400	1,300	2,200
Total TEQ-mammal	ng/kg	428/428	100	0.62	51,500	1,300	2,200
Organochlorine pesticides							
Dieldrin	µg/kg	419/427	98.1	0.015	152	5.9	8.3

Note: The UCL was selected as the EPC.

COPEC – chemical of potential ecological concern
dw – dry weight
EPC – exposure point concentration
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
LPRSA – Lower Passaic River Study Area
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
PCDD – polychlorinated dibenzo-*p*-dioxin
PCDF – polychlorinated dibenzofuran
RM – river mile
TEQ – toxic equivalent
UCL – upper confidence limit on the mean

Table 9-10. COPEC summary statistics for LPRSA surface water samples

COPEC	Unit	No. Detects/ No. Samples	% Detected	Concentration			
				Min. Detected	Max. Detected	Mean Detected	UCL
RM ≥ 10							
Metals							
Arsenic	µg/l	38/40	95	0.6	3.02	1.0	1.4
Cadmium	µg/l	64/64	100	0.018	0.897	0.11	0.13
Copper	µg/l	64/64	100	2.02	41.7	7.05	7.93
Lead	µg/l	64/64	100	0.104	48.7	6.99	8.35
Mercury	ng/l	64/64	100	3.09	330	51	61
Methylmercury	ng/l	40/40	100	0.063	2.21	0.31	0.36
Nickel	µg/l	40/40	100	1.01	7.9	2.07	2.39
Selenium	µg/l	23/40	57.5	0.3	2	0.6	0.63
Vanadium	µg/l	40/40	100	0.73	7.54	2.2	3.2
Zinc	µg/l	40/40	100	7.1	106	20	31
PAHs							
Benzo(a)pyrene	ng/l	40/40	100	9.67	347	70.1	85.4
Total HPAHs	ng/l	40/40	100	117	2,930	733	871
PCBs							
Total PCBs	ng/l	64/64	100	1.96	183	18.8	22.3
PCB TEQ-mammal	ng/l	64/64	100	0.0000563	0.0085	0.000443	0.00102

Table 9-10. COPEC summary statistics for LPRSA surface water samples

COPEC	Unit	No. Detects/ No. Samples	% Detected	Concentration Min. Detected	Max. Detected	Mean Detected	UCL
PCDDs/PCDFs							
PCDD/PCDF TEQ-mammal	ng/l	64/64	100	0.0000172	0.0828	0.00875	0.0163
Total TEQ-mammal	ng/l	64/64	100	0.000938	0.0831	0.00911	0.0169
Organochlorine pesticides							
Dieldrin	ng/l	40/40	100	0.61	3.05	1.5	1.6
<u>RM ≥ 4</u>							
Metals							
Arsenic	µg/l	94/98	95.9	0.6	3.02	1.0	1.3
Cadmium	µg/l	153/154	99.4	0.018	0.897	0.15	0.17
Copper	µg/l	154/154	100	1.68	41.7	8.69	11
Lead	µg/l	154/154	100	0.104	48.7	9.28	12.2
Mercury	µg/l	154/154	100	3.09	407	90	120
Methylmercury	µg/l	98/98	100	0.063	2.21	0.32	0.44
Nickel	µg/l	98/98	100	1.01	7.9	2.21	2.39
Selenium	µg/l	44/98	44.9	0.2	3.5	0.7	0.67
Vanadium	µg/l	97/98	99	0.73	7.6	3.0	3.3
Zinc	µg/l	98/98	100	3.03	106	22	29
PAHs							
Benzo(a)pyrene	ng/l	96/98	98	9.67	560	82	95.3
Total HPAHs	ng/l	98/98	100	117	4,550	829	961
PCBs							
Total PCBs	ng/l	154/154	100	1.96	183	25.8	34
PCB TEQ-Mammal	ng/l	154/154	100	0.0000563	0.0085	0.00056	0.000917
PCDDs/PCDFs							
PCDD/PCDF TEQ-Mammal	ng/l	154/154	100	2.84E-06	1.88	0.0361	0.11
Total TEQ-Mammal	ng/l	154/154	100	0.000863	1.88	0.0365	0.11
Organochlorine pesticides							
Dieldrin	ng/l	98/98	100	0.412	3.18	1.4	1.5

Note: The UCL was selected as the EPC.

COPEC – chemical of potential ecological concern
EPC – exposure point concentration
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
LPRSA – Lower Passaic River Study Area
OC – organic carbon
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
PCDD – polychlorinated dibenzo-*p*-dioxin
PCDF – polychlorinated dibenzofuran
RM – river mile
TEQ – toxic equivalent
UCL – upper confidence limit on the mean

9.1.2.5 Estimated doses

Dietary doses were calculated for site-wide exposures using Equation 8-1, with the prey, sediment, and surface WIRs and body weights from Table 9-4; the prey composition from Table 9-5; and the EPCs (based on UCLs) from Tables 9-7 through 9-10. These dietary doses are presented in Table 9-11.

Table 9-11. Dietary doses calculated for mink and river otter

COPEC	Units	Area	Mink Diet Scenario					River Otter Diet Scenario	
			1[a]	2[b]	3[c]	4[d]	5[e]	1[f]	2[g]
Arsenic	mg/kg bw/day	RM ≥ 10[h]	0.050	0.045	0.058	0.058	0.087	0.064	0.064
		site wide[i]	0.050	0.050	0.068	0.068	0.095	0.076	0.075
Cadmium	mg/kg bw/day	RM ≥ 10[h]	0.0074	0.0074	0.011	0.011	0.012	0.012	0.012
		site wide[i]	0.0069	0.0069	0.0092	0.0092	0.011	0.010	0.011
Copper	mg/kg bw/day	RM ≥ 10[h]	1.0	0.99	1.5	1.5	1.9	1.7	1.7
		site wide[i]	0.95	0.94	1.4	1.4	1.8	1.6	1.5
Lead	mg/kg bw/day	RM ≥ 10[h]	0.23	0.22	0.32	0.31	0.29	0.37	0.36
		site wide[i]	0.24	0.23	0.34	0.34	0.31	0.41	0.40
Mercury	µg/kg bw/day	RM ≥ 10[h]	11	11	20	20	21	23	24
		site wide[i]	11	11	20	20	21	23	24
Methylmercury	µg/kg bw/day	RM ≥ 10[h]	9.0	9.3	18	18	18	21	21
		site wide[i]	8.0	8.4	16	16	16	18	19
Nickel	mg/kg bw/day	RM ≥ 10[h]	0.71	0.65	1.7	1.6	1.3	2.0	1.9
		site wide[i]	0.30	0.28	0.67	0.65	0.55	0.79	0.75
Selenium	mg/kg bw/day	RM ≥ 10[h]	0.066	0.065	0.14	0.13	0.13	0.16	0.15
		site wide[i]	0.062	0.060	0.12	0.12	0.12	0.14	0.14
Vanadium	mg/kg bw/day	RM ≥ 10[h]	0.037	0.035	0.068	0.067	0.059	0.080	0.077
		site wide[i]	0.044	0.042	0.081	0.079	0.069	0.095	0.092
Zinc	mg/kg bw/day	RM ≥ 10[h]	2.4	2.5	4.4	4.4	4.6	5.0	5.1
		site wide[i]	2.7	2.7	5.0	5.0	5.1	5.8	5.8
Benzo(a)pyrene	µg/kg bw/day	RM ≥ 10[h]	4.2	4.1	5.7	5.6	5.3	6.6	6.4
		site wide[i]	4.0	3.9	5.7	5.6	5.3	6.6	6.4
Total HPAHs	µg/kg bw/day	RM ≥ 10[h]	43	42	57	57	54	67	66
		site wide[i]	40	39	58	57	53	67	66
Total PCBs	µg/kg bw/day	RM ≥ 10[h]	109	112	255	258	216	300	306
		site wide[i]	90	95	208	213	181	245	254
PCB TEQ - mammal[c]	ng/kg bw/day	RM ≥ 10[h]	1.1	1.2	2.4	2.5	2.3	2.9	3.0
		site wide[i]	1.1	1.2	2.3	2.4	2.2	2.7	2.9
PCDD/PCDF TEQ - mammal	ng/kg bw/day	RM ≥ 10[h]	9.3	9.5	19	19	17	22	23
		site wide[i]	7.0	7.4	13	14	13	15	16

Table 9-11. Dietary doses calculated for mink and river otter

COPEC	Units	Area	Mink Diet Scenario					River Otter Diet Scenario	
			1[a]	2[b]	3[c]	4[d]	5[e]	1[f]	2[g]
Total TEQ - mammal[j]	ng/kg bw/day	RM ≥ 10[h]	10	11	21	22	20	25	26
		site wide[i]	9.0	9.5	18	18	17	21	22
Dieldrin	µg/kg bw/day	RM ≥ 10[h]	1.7	1.7	3.9	3.9	3.3	4.6	4.6
		site wide[i]	1.3	1.4	3.0	3.0	2.6	3.5	3.6

a Dietary doses were calculated using a diet with 16.5% blue crab and 34% ≤ 30-cm fish (49.5% of the diet was terrestrial, which was assumed to be zero).

b Dietary doses were calculated using a diet with 16.5% blue crab, 31% ≤ 30-cm fish, and 3% > 30-cm fish (49.5% of the diet was terrestrial, which was assumed to be zero).

c Dietary doses were calculated using a diet with 16.5% blue crab and 83.5% ≤ 30-cm fish.

d Dietary doses were calculated using a diet with 16.5% blue crab, 80.5% ≤ 30-cm fish, and 3% > 30-cm fish.

e Dietary doses were calculated using a diet with 33.5% blue crab, 63.5% ≤ 30-cm fish, and 3% > 30-cm fish.

f Dietary doses were calculated using a diet with 15% blue crab and 85% ≤ 30-cm fish.

g Dietary doses were calculated using a diet with 15% blue crab, 80% ≤ 30-cm fish, and 5% > 30-cm fish.

h The RM ≥ 10 exposure area scenario includes an exposure area at and upstream of RM 10 for surface water, sediment, and SFF; all other fish (i.e., NFF ≤ 30 cm and fish > 30 cm) and blue crab have a site-wide exposure area.

i The site-wide exposure area includes site-wide exposure for fish, blue crab, and sediment, and at and upstream of RM 4 for surface water (i.e., includes only freshwater).

j Total TEQ is equal to the sum of PCB TEQ and PCDD/PCDF TEQ on a sample-by-sample basis; however, this is not necessarily the case for the sum of dietary doses in which UCLs were used.

bw – body weight
COPEC – chemical of potential ecological concern
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
NFF – non-small forage fish
PCB – polychlorinated biphenyl
PCDD – polychlorinated dibenzo-*p*-dioxin
PCDF – polychlorinated dibenzofuran
RM – river mile
SFF – small forage fish
TEQ – toxic equivalent
UCL – upper confidence limit on the mean

9.1.3 Effects

This section presents the effects data (i.e., TRVs) selected from the toxicological literature for the COPECs that were screened into this BERA based on the SLERA. A range of TRVs was evaluated. The selection was based on a comprehensive review of the primary literature and an evaluation of acceptability. TRVs were also based on previous documents developed by USEPA Region 2 for the LPRSA. Selected TRVs are consistent with the comments, responses, and directives received from USEPA on June 30, 2017 (USEPA 2017b), September 18, 2017 (USEPA 2017e), July 10, 2018 (USEPA 2018), January 2, 2019 (CDM 2019), and March 5, 2019 (USEPA 2019); during face-to-face meetings or conference calls on July 24, September 27, October 3, November 6, 2017, July 24, 2018, and August 16, 2018; and via additional deliverables and communications between the Cooperating Parties Group (CPG) and USEPA from August through December 2017, July through September 2018, and January through June 2019.

9.1.3.1 Methods for selecting TRVs

Two sets of mammal dietary TRVs were used for the derivation of HQs in this BERA. One set of TRVs was based on previous documents developed by USEPA Region 2 for the LPRSA:

- USEPA's revised draft of the LPRSA FFS (Louis Berger et al. 2014)
- USEPA's first draft of the LPRSA FFS (Malcolm Pirnie 2007b)
- USEPA's LPR restoration project PAR (Battelle 2005)

The second set of TRVs was based on a comprehensive review of the primary literature and an assessment of acceptability. TRVs were selected by first conducting a literature search for relevant toxicological studies. These studies were then evaluated for acceptability of use. For those studies considered acceptable (described in Appendix E), NOAEL and LOAEL daily doses were derived. TRVs were then selected for each COPEC based on an evaluation of all the acceptable NOAEL and LOAEL TRVs. Details regarding the literature search and acceptability of the studies are presented in Appendix E. The TRV derivation and selection processes, along with general uncertainties in the use of TRVs to estimate risk, are the same for mammals as for birds, as described in Section 8.1.3.1. COPEC-specific uncertainties associated with mammal diet TRVs are discussed in the following section (Section 9.1.3.2).

9.1.3.2 Selected TRVs for mammals

The mammal dietary TRVs are presented in Table 9-12, and details on the derivation of these values are presented in the following subsections.

Table 9-12. Mammal dietary TRVs

COPEC	Units	Range of TRVs[a]								
		TRV-A[b]				TRV-B[c]				
		NOAEL	LOAEL	Endpoint	Source	NOAEL	LOAEL	Endpoint	Source	Document
Arsenic	mg/kg bw/day	2.6	5.4	growth (rat)	Hext et al. (1999)	0.32	4.7	growth (NOAEL); cellular (LOAEL) (rat)	Schroeder et al. (1968) (NOAEL); Brown et al. (1976) (LOAEL), both as cited in USEPA (2002f)	2005 PAR (Battelle 2005)
Cadmium	mg/kg bw/day	3.5	13	growth (rat)	Machemer and Lorke (1981)	0.060	2.64	reproduction (mouse)	Webster (1988) (NOAEL); Schroeder & Mitchener (1971) (LOAEL), both as cited in USEPA (2002f)	2005 PAR (Battelle 2005)
Copper	mg/kg bw/day	18	26	reproduction (mink)	Aulerich et al. (1982)	3.4	6.8	reproduction (mink)	Aulerich et al. (1982) as cited in USEPA (2007c)	revised FFS (Louis Berger et al. 2014)
Lead	mg/kg bw/day	11	90	growth (rat)	Azar et al. (1973)	0.71	7.0	reproduction (rat)	Grant et al. (1980) as cited in USEPA (2005b)	revised FFS (Louis Berger et al. 2014)
Methylmercury/ mercury	µg/kg bw/day	160	250	growth, survival (mink)	Wobeser et al. (1976b)	16	27	growth (mink)	Wobeser et al. (1976a, b) as derived in USEPA (1995a)	revised FFS (Louis Berger et al. 2014)
Nickel	mg/kg bw/day	40	80	reproduction (rat)	Ambrose et al. (1976)	0.133	31.6	reproduction (rat)	Smith et al. (1993) as cited in USEPA (2002f)	2005 PAR (Battelle 2005)
Selenium	mg/kg bw/day	0.016[d]	0.16	growth (rat)	Behne et al. (1992)	0.050	1.21	liver (NOAEL); reproduction (LOAEL) (mouse)	Harr et al. (1967) (NOAEL); Schroeder & Mitchener (1971) (LOAEL), both as cited in USEPA (2002f)	2005 PAR (Battelle 2005)
Vanadium	mg/kg bw/day	0.27[d]	2.7	growth (rat)	Adachi et al. (2000)	na	na	na	na	na
Zinc	mg/kg bw/day	160	320	reproduction (rat)	Schlicker and Cox (1968)	9.6	411	growth (mouse)	Culp et al. (1998) as cited in USEPA (2007d)	2005 PAR (Battelle 2005)

Table 9-12. Mammal dietary TRVs

COPEC	Units	Range of TRVs[a]								
		TRV-A[b]				TRV-B[c]				
		NOAEL	LOAEL	Endpoint	Source	NOAEL	LOAEL	Endpoint	Source	Document
Total HPAHs	µg/kg bw/day	na	na	na	na	620	3,100	growth (mouse)	Culp et al. (1998) as cited in USEPA (2007d)	revised FFS (Louis Berger et al. 2014)
Benzo(a)pyrene (HPAH)	µg/kg bw/day	1,000[d]	10,000	reproduction (mouse)	MacKenzie and Angevine (1981)	na	na	na	na	na
Total PCBs	µg/kg bw/day	80	96	reproduction (mink)	Chapman (2003)	69	82	reproduction (mink)	Chapman (2003)	revised FFS (Louis Berger et al. 2014)
PCB TEQ - mammal PCDD/PCDF TEQ - mammal Total TEQ - mammal	ng/kg bw/day	2.6	8.8	reproduction (mink)	Hochstein et al. (2001)	0.08	2.2	reproduction (mink)	Tillett et al. (1996)	revised FFS (Louis Berger et al. 2014)
Dieldrin	µg/kg bw/day	15	30	reproduction (rat)	Harr et al. (1970)	15	30	reproduction (rat)	Harr et al (1970) as cited in USEPA (2005b)	2014 FFS (Louis Berger et al. 2014)

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

[b] TRVs from the primary literature review were derived based on the process identified in Section 9.1.3.1.

[c] TRVs were based on USEPA's revised draft of the LPR restoration project FFS (Louis Berger et al. 2014), first draft of the LPR restoration project FFS (Malcolm Pirnie 2007b) or LPR restoration project PAR (Battelle 2005).

[d] NOAEL was extrapolated from LOAEL using an uncertainty factor of 10.

BERA – baseline ecological risk assessment
bw – body weight
LPR – Lower Passaic River
LPRSA – Lower Passaic River Study Area
PCB – polychlorinated biphenyl
PCDD – polychlorinated dibenzo-*p*-dioxin

COPEC – chemical of potential ecological concern
FFS – focused feasibility study
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
LOAEL – lowest-observed-adverse-effect level
na – not available
NJDEP – New Jersey Department of Environmental Protection
NOAEL – no-observed-adverse-effect level
PAR – pathways analysis report
PCDF – polychlorinated dibenzofuran
TEQ – toxic equivalent
TRV – toxicity reference value
USEPA – US Environmental Protection Agency

Arsenic

One toxicity study was considered acceptable for mammals exposed to arsenic (Byron et al. 1967). Data were insufficient for the development of an SSD curve (i.e., data were not available for at least five species). The NOAEL (2.6 mg/kg bw/day) and LOAEL (5.4 mg/kg bw/day) were based on a decrease in the body weights of rats exposed to dietary arsenic in the form of sodium arsenite. There is uncertainty due to the limited number of dietary studies (one study) identified from the literature review.

A NOAEL and LOAEL of 0.32 and 4.7 mg/kg bw/day, respectively, were also selected for arsenic (Battelle 2005), based on USEPA Region 9 BTAG TRVs (2002f). These TRVs, as cited by USEPA, were based on rat toxicity using data from Schroeder et al. (1968) and Brown et al. (1976). The LOAEL was based on a change in respiration rate, and both TRVs were based on drinking water exposure.

Cadmium

Three toxicity studies were considered acceptable for mammals exposed to cadmium (Pond and Walker 1975; Machemer and Lorke 1981; Dodds-Smith et al. 1992). These studies evaluated growth, mortality, and/or reproduction in rats and shrews. Data were insufficient for the development of an SSD (i.e., data were not available for at least five species). The lowest LOAEL of 13 mg/kg bw/day was based on the decreased maternal body weight of rats exposed to dietary cadmium chloride; the NOAEL from this study was 3.5 mg/kg/bw/day (Machemer and Lorke 1981). The LOAEL TRV of 13 mg/kg bw/day and NOEAL TRV of 3.5 mg/kg/bw/day were selected. There is uncertainty due to the limited number of dietary studies identified from the literature review.

A NOAEL and LOAEL of 0.06 and 2.64 mg/kg bw/day, respectively, were also selected (Battelle 2005), based on USEPA Region 9 BTAG TRVs (2002f). These TRVs, as cited, were based on mouse reproductive toxicity using data from Webster (1988) and Schroeder & Mitchener (1971). These TRVs were based on drinking water exposure.

Copper

Four acceptable toxicity studies were available from the literature in which mammals were exposed to dietary copper. Two of these studies were conducted with rats and mice (Hebert et al. 1993; NTP 1993), one study each was conducted with shrew (Dodds-Smith et al. 1992) and mink (Aulerich et al. 1982). These studies evaluated growth, mortality, and/or reproduction. Data were insufficient for the development of an SSD (i.e., data were not available for at least five species). TRVs were selected from the mink reproduction study by Aulerich et al. (1982), during which mink were fed 0, 25, 50, 100, or 200 mg/kg copper (as copper sulfate) for 153 or 357 days. In mink fed 100 mg/kg, the significant adverse effect of decreased kit survival was observed (38% mortality compared to 12% in the control), as was decreased litter mass (70 g/kit compared to 100 g/kit in the control) (Aulerich et al. 1982). The LOAEL and NOAEL from this study of 26 and 18 mg/kg bw/day, respectively, were selected as the TRVs for copper. There is uncertainty associated with the limited number of dietary studies identified from the literature review.

A NOAEL and LOAEL of 3.4 and 6.8 mg/kg bw/day, respectively, were also selected for copper from previous documents developed by USEPA Region 2 for the LPRSA (Louis Berger et al. 2014) based on USEPA's Eco-SSL document for copper (USEPA 2007c), which used the same mink reproductive toxicity data reported in Aulerich et al. (1982). The LOAEL of 6.8 mg/kg bw/day was derived from the observation of effects at 25 mg/kg in the diet. At this concentration, there was a reduction in the overall number of kits whelped because of the lower number of females whelping in the exposure group (6 out of 11 compared to 11 out of 12 in the control). At 25 mg/kg, there were no statistically identifiable effects on kit mortality or growth, nor any apparent effect on the average number of kits whelped per female, and the number of females whelping per exposure group was not dose responsive (12 out of 12 whelped in the mink fed 50 parts per million [ppm]). Thus, there is some uncertainty associated with this LOAEL.

Lead

Three acceptable toxicity studies were available from the literature in which mammals were exposed to dietary lead in the form of lead acetate. Two studies evaluated growth in rats (Azar et al. 1973) and mice (Wise 1981), and one study evaluated reproduction in mice (Iavicoli et al. 2006). Data were insufficient for the development of an SSD (i.e., data were not available for at least five species). The lowest LOAEL of 90 mg/kg bw/day resulted in decreased body weight of rat offspring (Azar et al. 1973). The NOAEL from this study was 11 mg/kg bw/day (Azar et al. 1973). The lowest LOAEL of 90 mg/kg bw/day and NOAEL of 11 mg/kg bw/day were selected as the LOAEL and NOAEL TRVs, respectively. There is uncertainty associated with the limited number of dietary studies identified from the literature review.

A NOAEL and LOAEL of 0.71 and 7.1 mg/kg bw/day, respectively, were selected for lead from previous documents developed by USEPA Region 2 for the LPRSA (Louis Berger et al. 2014) based on USEPA's Eco-SSL document for lead (USEPA 2005b). These TRVs used rat reproduction data from Grant et al. (1980), which were based on drinking water exposure, an uncertain method for evaluating the dietary exposure. There is uncertainty associated with using TRVs based on drinking water exposure to assess risks to LPRSA receptors via the dietary pathway.

Methylmercury/Mercury

Three toxicity studies were considered acceptable for the derivation of LOAELs for mammals exposed to mercury (Aulerich et al. 1974; Verschuuren et al. 1976; Wobeser et al. 1976b). These studies evaluated growth and mortality in mink. Data were insufficient for the development of an SSD (i.e., data were not available for at least five species). Both sets of TRVs were developed based on data reported by Wobeser et al.

(1976b). Using a body weight of 1.34 kg and a FIR of 0.18 kg/day based on Bleavins and Aulerich (1981), a LOAEL and NOAEL of 250 and 160 µg/kg bw/day, respectively, were derived. The LOAEL was based on the growth and survival of mink (Wobeser et al. 1976b) following exposure to methylmercury in their diet for 93 days. There is uncertainty associated with the limited number of dietary studies identified from the literature review.

A NOAEL and LOAEL of 16 and 27 µg/kg bw/day, respectively, were selected for mercury from previous documents developed by USEPA Region 2 for the LPRSA (Louis Berger et al. 2014) based on data from Wobeser et al. (1976a, 1976b), as reported by USEPA (1995a). These TRVs were based on mink growth and reproduction, and were derived using a female body weight of 1 kg and a FIR of 0.15 kg/day from Bleavins and Aulerich (1981) and Hornshaw et al. (1983), respectively. First, a NOAEL and LOAEL of 160 and 270 µg/kg bw/day, respectively, were derived; then a subchronic-to-chronic factor of 10 was applied to the NOAEL and LOAEL to derive the selected TRVs of 16 and 27 µg/kg bw/day. There is uncertainty associated with the use of extrapolation factors to derive TRVs.

At both the selected dietary dose LOAELs (250 and 270 µg/kg bw/day), mink were fed 1.8 mg/kg mercury in their diet over 93 days. However, there is uncertainty associated with the reduction of this LOAEL by a factor of 10, because sufficient information is not available to conclude that if the dietary concentrations had been reduced by a factor of 10 (to 0.18 mg/kg), effects would have been observed over a longer exposure period. In fact, Wobeser et al. (1976a) found that mink fed diets of up to 75% fish containing 0.44 mg/kg mercury over a 145-day period suffered no effects.

Nickel

Three studies were considered acceptable for the derivation of LOAELs and NOAELs for mammals exposed to nickel (Weber and Reid 1969; Ambrose et al. 1976; Nation et al. 1985). These studies evaluated the growth, reproduction, and/or mortality effects of a dietary dose of nickel in rats, mice, and dogs. Data were insufficient for the development of an SSD (i.e., data were not available for at least five species). The lowest LOAEL of 80 mg/kg bw/day was based on reduced body weight of rat offspring over three generations (Ambrose et al. 1976). The NOAEL from this study was 40 mg/kg bw/day (Ambrose et al. 1976). The LOAEL of 80 mg/kg bw/day and NOAEL of 40 mg/kg bw/day were selected as TRVs. There is uncertainty associated with the limited number of dietary studies identified from the literature review.

A NOAEL and LOAEL of 0.133 and 31.6 mg/kg bw/day, respectively, were also selected for nickel based on USEPA Region 9 BTAG TRVs (2002f). These TRVs were based on rat reproductive toxicity using drinking water exposure data from Smith et al. (1993).

Selenium

There were four studies considered acceptable for the derivation of LOAELs for mammals exposed to selenium (Halverson et al. 1966; Behne et al. 1992; Jia et al. 2005; Julius et al. 1983); these studies evaluated growth or mortality in rats or hamsters. Data were insufficient for the development of an SSD (i.e., data were not available for at least five species). The lowest LOAEL of 0.16 mg/kg bw/day was based on a decrease in the body weights of rats exposed to dietary selenium in the form of selenomethionine (Behne et al. 1992). No NOAEL was available from this study using selenomethionine, although selenium in the form of selenite did not result in a body weight decrease at 0.16 mg/kg bw/day. The lowest LOAEL of 0.16 mg/kg bw/day was selected as the LOAEL TRV. A NOAEL of 0.016 mg/kg bw/day was derived by dividing the LOAEL by 10. There is uncertainty associated with the limited number of dietary studies identified from the literature review. There is also uncertainty associated with the use of an extrapolation factor to derive a NOAEL.

A NOAEL and LOAEL of 0.05 and 1.21 mg/kg bw/day, respectively, were also selected for selenium (Battelle 2005) based on USEPA Region 9 BTAG TRVs (2002f). These TRVs were based on mouse toxicity using data from Harr et al. (1967) and Schroeder & Mitchener (1971).

Vanadium

Two studies were considered acceptable for the derivation of LOAELs for mammals exposed to vanadium (Elfant and Keen 1987; Adachi et al. 2000); these studies evaluated reproduction and/or growth in rats. Data were insufficient for the development of an SSD (i.e., data were not available for at least five species). The lowest LOAEL of 2.7 mg/kg bw/day was based on a decrease in the body weights of rats exposed to dietary vanadium in the form of sodium meta-vanadate (Adachi et al. 2000). No NOAEL was available from this study (Adachi et al. 2000). The lowest LOAEL of 2.7 mg/kg bw/day was selected as the LOAEL TRV, and a NOAEL of 0.27 mg/kg bw/day was derived by dividing the LOEAL by 10. There is uncertainty associated with the limited number of dietary studies identified from the literature review. There is also uncertainty associated with the use of an extrapolation factor to derive a NOAEL.

No TRVs for vanadium were available from the draft FFS (Malcolm Pirnie 2007b) or revised FFS (Louis Berger et al. 2014).

Zinc

Three acceptable toxicity studies were available from the literature in which mammals were exposed to dietary zinc (Sutton and Nelson 1937; Schlicker and Cox 1968; Straube et al. 1980). These studies evaluated growth, reproduction, or mortality in rats or ferrets. Data were insufficient for the development of an SSD (i.e., data were not available for at least five species). The lowest LOAEL of 320 mg/kg bw/day was based on a reduction in fetal growth and an increase in fetal resorptions in rats when exposed to dietary zinc oxide (Schlicker and Cox 1968). A NOAEL of 160 mg/kg bw/day from this study (Schlicker and Cox 1968) was the highest NOAEL below the lowest LOAEL. The LOAEL of 320 mg/kg bw/day and NOAEL of 160 mg/kg bw/day were selected as TRVs. There is uncertainty associated with the limited number of dietary studies identified from the literature review.

A NOAEL and LOAEL of 9.6 and 411 mg/kg bw/day, respectively, were also selected for zinc (Battelle 2005) based on USEPA Region 9 BTAG TRVs (2002f). These TRVs were based on rat and mouse toxicity using data from Aughey et al. (1977) and Schlicker & Cox (1968).

Total HPAHs and Benzo(a)pyrene

No acceptable studies were found that exposed mammals to HPAH mixtures in the diet; instead, the lowest LOAEL for a single HPAH (i.e., benzo(a)pyrene) was selected and used as a surrogate for HPAHs. There was one acceptable study in which a LOAEL for benzo(a)pyrene could be derived (Appendix E); in this study, reproduction was adversely affected in mice exposed to benzo(a)pyrene via gavage over 10 days during gestation (MacKenzie and Angevine 1981). Two additional studies evaluated mammals and benzo(a)pyrene; however, no adverse effects were observed in these studies. A LOAEL of 10,000 µg/kg bw/day was selected, and a NOAEL of 1,000 µg/kg bw/day was derived by dividing the LOAEL by 10. There is uncertainty associated with the limited number of studies identified from the literature review, and with the selection of TRVs based on gavage exposure. There is also uncertainty associated with the use of an extrapolation factor to derive a NOAEL.

A NOAEL and LOAEL of 0.62 and 3.1 µg/kg bw/day, respectively, were also selected for HPAHs from previous documents developed by USEPA Region 2 for the LPRSA (Louis Berger et al. 2014) based on USEPA's Eco-SSL document for PAHs (USEPA 2007d). These TRVs were derived based on mouse growth toxicity data as reported by Culp et al. (1998); mice were exposed to a PAH mixture, but only the value for benzo(a)pyrene was used to calculate the TRV. Using the total PAH concentration from the study resulted in a NOAEL and LOAEL of 30 and 61 µg/kg bw/day, respectively. Not all HPAHs are known to be as toxic as benzo(a)pyrene, so the comparison of a dose of total HPAHs to a benchmark dose based on benzo(a)pyrene is considered conservative and uncertain.

Total PCBs

For total PCBs, 14 toxicity studies were considered for the derivation of TRVs: 12 of these studies were conducted with mink (Aulerich and Ringer 1977; Aulerich et al. 1985; Bleavins et al. 1980; Brunström et al. 2001; Bursian et al. 2006; Bursian et al. 2013; Heaton et al. 1995; Hornshaw et al. 1983; Jensen et al. 1977; Kihlstrom et al. 1992; Restum et al. 1998; Wren et al. 1987) and 2 were conducted with mice (Linzey 1987; Simmons and McKee 1992). There were not enough species to derive an SSD curve. Because of the numerous studies on mink and the reported sensitivity of mink to

PCBs, the mink data were evaluated in greater detail to determine whether a dose-response relationship could be developed. In five of the toxicity studies for mink, diets consisted of field-collected fish (Bursian et al. 2006; Bursian et al. 2013; Heaton et al. 1995; Hornshaw et al. 1983; Restum et al. 1998). These studies are not recommended for TRV derivation using dose response data because of potential co-contaminants in the fish collected from the field. However, the effects levels from these studies were included in this evaluation to determine how toxicity data based on field-collected diets compare, in general, to data based on laboratory-controlled diets.

Toxicity data for mink include a number of variables that can influence effects levels, such as:

- **Type of PCB in diet –** laboratory PCB mixture (e.g., Aroclor 1254 or 1242) or field-collected fish
- **Specific reproductive endpoint observed –** number of kits born alive, kit growth, kit survival after birth, adult growth
- **Exposure period –** number of breeding periods to which maternal generation was exposed, or exposure of second generation

Therefore, the NOAELs and LOAELs from the mink studies were plotted for two separate endpoints (number of live kits whelped per female [Figure 9-1] and kit body weight at four to six weeks [Figure 9-2]), incorporating information about exposure periods and using data for both technical PCB mixtures and field-collected fish. Detailed information from the toxicity studies used to create these figures are presented in Appendix E.

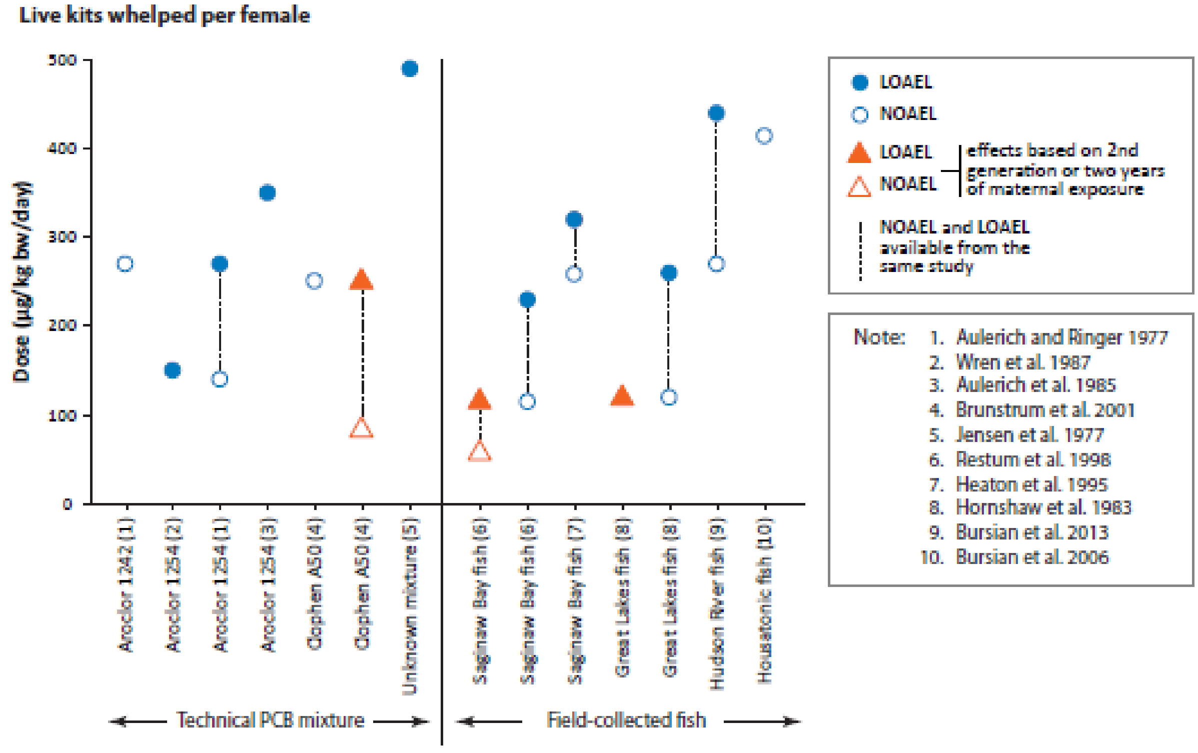


Figure 9-1. Mammal dietary PCB NOAELs and LOAELs for number of live mink kits whelped per female

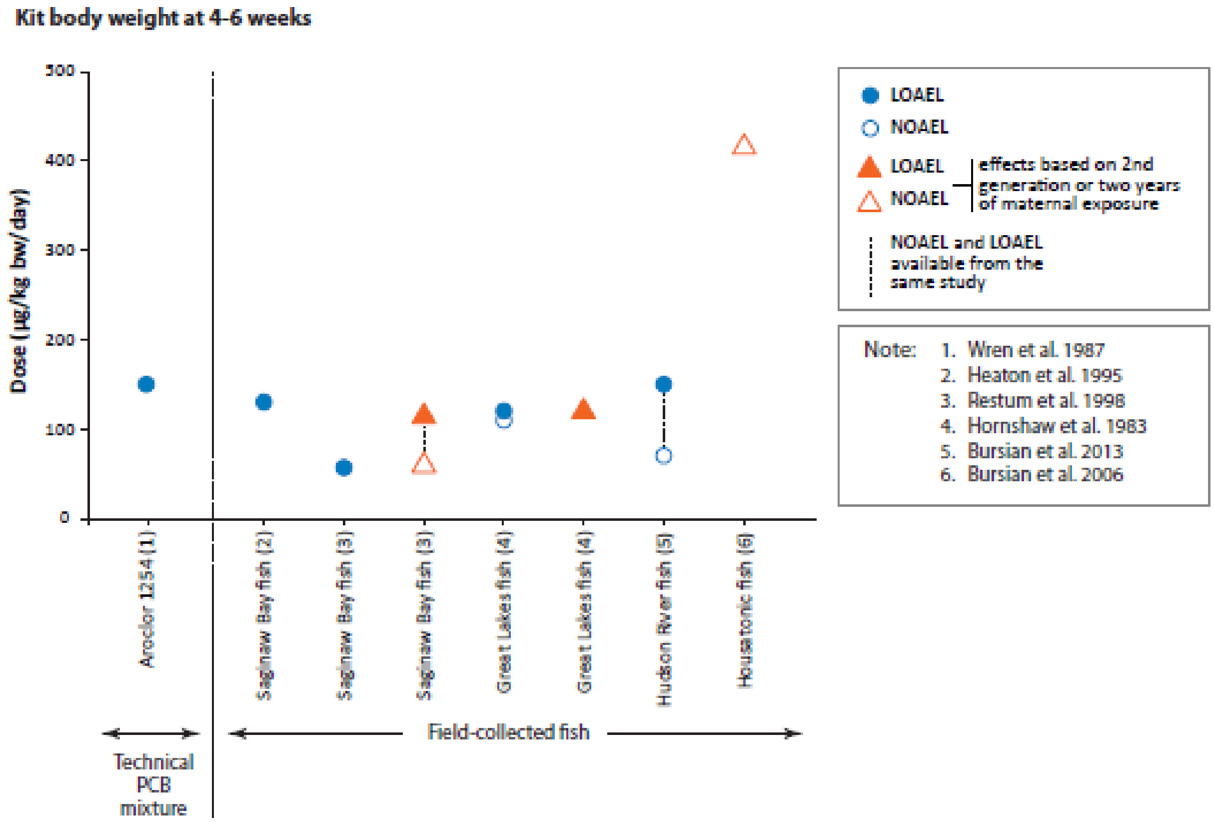


Figure 9-2.Mammal dietary PCB NOAELs and LOAELs for mink kit body weight at 4 to 6 weeks

Figures 9-1 and 9-2 present both NOAELs and LOAELs, if available, for a particular study. If both a NOAEL and a LOAEL are both available, the level at which effects might occur is somewhere between the two values (represented as a dashed line). NOAELs with no LOAELs are unbounded; it is uncertain at what higher level, if tested, effects could occur. Similarly, LOAELs with no NOAELs are unbounded, and no-effects levels below the LOAEL are unknown.

LOAELs were generally lower for the kit body weight endpoint, ranging from 57 to 150 µg/kg bw/day, compared to the endpoint for live kits whelped per female, which ranged from 120 to 490 µg/kg bw/day. The results show that in general, there is better agreement among studies regarding effects on kit body weight after four to six weeks than on live kits whelped per female.

To evaluate the dose-response relationship for studies conducted with technical PCB mixtures (i.e., excluding studies conducted with field-collected fish), data for percent reduction in kit survival compared to control kit survival were plotted against the dose (Figure 9-3). Data for the kit body weight endpoint could not be evaluated using a dose-response relationship, because there was only one study conducted with a technical PCB mixture for this endpoint. For the technical PCB mixture dose-response relationship, there was either 0% reduction in kit survival at a dose

≤ 270 µg/kg bw/day, or 100% reduction in kit survival at a dose ≥ 270 µg/kg bw/day in most studies. The dose of 270 µg/kg bw/day was both a NOAEL (at 100% survival) using Aroclor 1242 and a LOAEL (at 0% survival) using Aroclor 1254 in the same study (Aulerich and Ringer 1977), indicating that mink are more sensitive to Aroclor 1254 than to Aroclor 1242. These data were insufficient to develop a dose-response curve because of the lack of data with responses between the 0 and 100% levels.

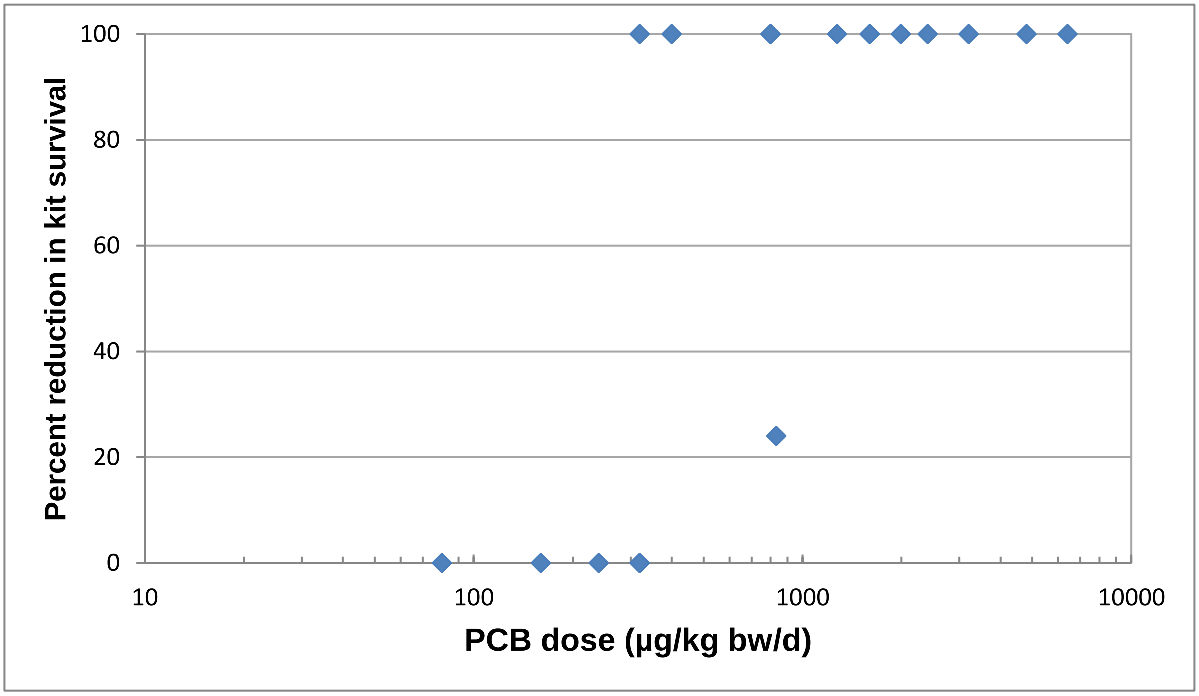


Figure 9-3. Dose-response results for mink fed laboratory technical PCB mixtures

Chapman (2003) evaluated mink PCB toxicity data to derive TRVs for USEPA Region 5 based on interpolation of laboratory toxicity data from Aulerich and Ringer (1977), Wren et al. (1987), and Kakela (2002). Effects levels were calculated for both Aroclor 1242 and Aroclor 1254 data, and were lower for Aroclor 1254. The interpolated dietary concentration resulting in a 25% decrease in endpoint response (1,000 µg/kg ww) was determined to be the low-effect level for Aroclor 1254, and the interpolated dietary concentration associated with a 10% decrease in endpoint response (1,100 µg/kg ww) was determined to be the no-effect level (Chapman 2003) (Table 9-13). A factor of 0.52 was applied to the no-effect and low-effect levels to account for the lower effects levels observed in several studies that were conducted over 2 years or into the second generation (Brunström et al. 2001; Restum et al. 1998), resulting in adjusted interpolated dietary concentrations of 500 and 600 µg/kg ww, respectively. These dietary concentrations were converted to NOAEL and LOAEL doses (80 and 96 µg/kg bw/day, respectively) assuming a female FIR of 0.16 kg/kg bw/day from Bleavins and Aulerich (1981); a female FIR was used because the endpoint measured was reproduction. USEPA recommends the lower of the TRVs (i.e., Aroclor 1254 rather than Aroclor 1242) because of the uncertain toxicity of PCBs in the field compared to that of Aroclors under controlled conditions. The USEPA Region 5 LOAEL TRV for Aroclor 1254 of 96 mg/kg bw/day is similar to the lowest LOAEL from laboratory studies (150 mg/kg bw/day for kit body weight after four to six weeks), but also accounts for increased toxicity after 2 years of exposure or in the second generation; therefore, this value was selected as the TRV (Table 9-13).

Table 9-13. Interpolated dietary PCB effect levels for mammals

Aroclor	Interpolated Effect Level (µg/kg bw/day)	
	10% (NOAEL)	25% (LOAEL)
1242	210	220
1254	**80**	**96**

Source: Chapman (2003)

Note: Effects levels in Chapman (2003) are presented as concentrations in diet (µg/kg food); these values were converted to doses (µg/kg bw/day), assuming a female FIR of 0.14 kg/kg bw/day (Bleavins and Aulerich 1981).

Bold identifies the selected TRV.

BERA – baseline ecological risk assessment
bw – body weight
FIR – food ingestion rate
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level

A NOAEL and LOAEL of 69 and 82 µg/kg bw/day, respectively, were also selected for total PCBs from previous documents developed by USEPA Region 2 for the LPRSA (Louis Berger et al. 2014), based on the same interpolated no-effect and low-effect values, using mink reproductive data as described in Chapman (2003), and assuming a FIR of 0.137 kg/day. The two sets of TRVs differ slightly due to the different FIRs used.

TEQ - Mammal

Six acceptable toxicity studies were available from the literature in which mammals in the laboratory were exposed to 2,3,7,8-TCDD incorporated into the diet. Two of these studies were conducted with mink (Hochstein et al. 1998; Hochstein et al. 2001), one with guinea pigs (DeCaprio et al. 1986), and three with rats (Murray et al. 1979; Kociba et al. 1978; Van Birgelen et al. 1994). Data were not sufficient to develop an SSD or a dose-response relationship (i.e., data were not available for at least five species). The lowest LOAEL was a dietary dose of 4.9 ng/kg bw/day, resulting in decreased growth in guinea pigs (DeCaprio et al. 1986); animals exposed to this dose had 12 to 15% decreased body weights compared to those in the control groups. The second-lowest LOAEL was 8.8 ng/kg bw/day, resulting in decreased mink kit survival at three and six weeks after birth compared to the control (Hochstein et al. 2001). The LOAEL of 8.8 ng/kg bw/day was selected as the LOAEL TRV, because the studies with mink were more directly applicable to the selected LPRSA mammal species (i.e., mink and the closely related river otter). The NOAEL of 2.6 ng/kg bw/day from the same study was selected as the LOAEL TRV.

Three additional toxicity studies were considered in which mink were fed field-collected fish from sites contaminated with PCBs, PCDDs/PCDFs, and other potential contaminants. The three locations from which fish were collected were Saginaw Bay (Tillitt et al. 1996), the Housatonic River (Bursian et al. 2006), and the Hudson River (Bursian et al. 2013). The following LOAELs from these studies were based on decreased kit or juvenile survival: 2.24 ng/kg bw/day for the Saginaw Bay study (Tillitt et al. 1996), 7.7 ng/kg bw/day for the Housatonic River study (Bursian et al. 2006), and 0.97 ng/kg/day for the Hudson River study (Bursian et al. 2013).

A NOAEL and LOAEL of 0.08 and 2.2 ng/kg bw/day, respectively, were selected for TEQ-mammal from previous documents developed by USEPA Region 2 for the LPRSA (Louis Berger et al. 2014) based on Tillett et al. (1996). These TRVs were based on mink exposure to field-contaminated carp. Field-collected fish may also have contained other contaminants; therefore, it is impossible to determine if impacts on the mink are solely due to PCDD exposure in their diet, and there is some uncertainty associated with these selected TRVs.

Dieldrin

Eight acceptable toxicity studies were available from the literature in which mammals were exposed to dietary dieldrin. Data were insufficient for the development of an SSD (i.e., data were not available for at least five species). The lowest LOAEL of 30 µg/kg bw/day resulted in adverse reproductive effects in rats (Harr et al. 1970). The NOAEL from this study was 15 µg/kg bw/day (Harr et al. 1970). The lowest LOAEL and NOAEL of 30 and 15 µg/kg bw/day, respectively, were selected. There is uncertainty associated with the limited number of dietary studies identified from the literature review.

These same NOAEL and LOAEL values (i.e., 15 and 30 µg/kg bw/day, respectively) were also selected for dieldrin (Louis Berger et al. 2014) based on USEPA's Eco-SSL document for dieldrin (USEPA 2005b) using data from Harr et al (1970).

9.1.4 Risk characterization

This section presents the HQs for mammals (Section 9.1.4.1), as well as uncertainties associated with the HQ calculations (Section 9.1.4.2). In addition to the original HQ calculations, this section presents alternate HQs calculated based on the identified uncertainties. These alternate HQs were calculated to determine if any of the uncertainties could result in risk conclusions that were different from those determined by the original HQs.

For COPECs with HQs ≥ 1.0 when compared with LOAEL TRVs, a comparison of background data to site data is also presented, consistent with USEPA guidance (USEPA 2002c).

9.1.4.1 Dietary HQs

Dietary HQs were calculated for the COPECs identified in Table 9-2 using the EPCs described in Table 9-6 (based on UCLs) and the TRVs identified in Table 9-12. Appendix G detail the dietary doses, TRVs, and calculated HQs for the mammal dietary COPECs (Tables G16 through G22).

Mink

HQs for mink were calculated using the five diet scenarios identified in Table 9-4 and the two exposure areas identified in Table 9-5, and are presented in Table 9-14. Mink dietary LOAEL HQs were ≥ 1.0 for total PCBs, PCB TEQ - mammal, PCDD/PCDF TEQ - mammal, and total TEQ - mammal. Aquatic/terrestrial diets (i.e., Scenarios 1 and 2) resulted in lower HQs than more conservative diets that assumed that mink feed exclusively on aquatic prey (i.e., Scenarios 3 through 5). The RM ≥ 10 exposure area resulted in higher HQs for PCBs and TEQ - mammal than the site-wide exposure area because of higher EPCs for ≤ 30-cm fish. The EPCs for ≤ 30-cm fish for both exposure areas were influenced by higher PCB and TEQ - mammal concentrations in perch and the percentage of SFF (which had lower concentrations). The ≤ 30-cm fish for the site-wide exposure area were 45% SFF, whereas ≤ 30-cm fish for the RM ≥ 10 exposure area (i.e., SFF from RM ≥ 10 and NFF site wide) were 19% SFF. A higher percentage of SFF in the EPC dataset resulted in lower total PCB and TEQ - mammal EPCs and HQs, because the SFF effectively reduced the influence of the perch on the EPC.

Table 9-14. Mink dietary HQs

Diet Scenario[b]	Area	Dose		Range of TRVs[a]				Range of HQs[a]			
				TRV-A[c]		TRV-B[d]		HQ Based on TRV-A[c]		HQ Based on TRV-B[d]	
		Value	Unit	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL
Arsenic											
1–5	RM ≥ 10[e]	0.045–0.087	mg/kg bw/day	2.6	5.4	0.32	4.7	0.017–0.033	0.0084–0.016	0.14–0.27	0.0096–0.018
	site wide[f]	0.050–0.095	mg/kg bw/day					0.019–0.037	0.0092–0.018	0.16–0.30	0.011–0.020
Cadmium											
1–5	RM ≥ 10[e]	0.0074–0.012	mg/kg bw/day	3.5	13	0.060	2.54	0.0021–0.0035	0.00057–0.00093	0.12–0.20	0.0028–0.0046
	site wide[f]	0.0069–0.011	mg/kg bw/day					0.0020–0.0032	0.00053–0.00085	0.11–0.18	0.0026–0.0042
Copper											
1–5	RM ≥ 10[e]	0.99–1.9	mg/kg bw/day	18	26	3.4	6.8	0.055–0.11	0.038–0.074	0.29–0.56	0.14–0.28
	site wide[f]	0.94–1.8	mg/kg bw/day					0.052–0.10	0.036–0.069	0.28–0.53	0.14–0.27
Lead											
1–5	RM ≥ 10[e]	0.22–0.31	mg/kg bw/day	11	90	0.71	7	0.020–0.029	0.0025–0.0035	0.32–0.45	0.032–0.045
	site wide[f]	0.23–0.35	mg/kg bw/day					0.021–0.032	0.0026–0.0039	0.33–0.49	0.033–0.050

Table 9-14. Mink dietary HQs

		Dose		Range of TRVs[a]				Range of HQs[a]			
				TRV-A[c]		TRV-B[d]		HQ Based on TRV-A[c]		HQ Based on TRV-B[d]	
Diet Scenario[b]	Area	Value	Unit	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL
Mercury											
1–5	RM ≥ 10[e]	11–21	µg/kg bw/day	160	250	16	27	0.069–0.13	0.044–0.083	0.69–**1.3**	0.41–0.76
	site wide[f]	11–21	µg/kg bw/day					0.069–0.13	0.044–0.083	0.69–**1.3**	0.41–0.76
Methylmercury											
1–5	RM ≥ 10[e]	9–18	µg/kg bw/day	160	250	16	27	0.056–0.11	0.036–0.073	0.56–**1.1**	0.33–0.68
	site wide[f]	8–16	µg/kg bw/day					0.050–0.10	0.032–0.065	0.50–**1.0**	0.30–0.60
Nickel											
1–5	RM ≥ 10[e]	0.65–1.7	mg/kg bw/day	40	80	0.133	31.6	0.056–0.11	0.036–0.073	**4.9–13**	0.021–0.053
	site wide[f]	0.28–0.67	mg/kg bw/day					0.050–0.10	0.032–0.065	**2.1–5.1**	0.0088–0.21
Selenium											
1–5	RM ≥ 10[e]	0.065–0.14	mg/kg bw/day	0.016	0.16	0.050	121	**4.0–8.5**	0.40–0.85	**1.3–2.7**	0.053–0.11
	site wide[f]	0.060–0.12	mg/kg bw/day					**3.8–7.8**	0.38–0.78	**1.2–2.5**	0.050–0.10

Table 9-14. Mink dietary HQs

Diet Scenario[b]	Area	Dose		Range of TRVs[a]				Range of HQs[a]			
				TRV-A[c]		TRV-B[d]		HQ Based on TRV-A[c]		HQ Based on TRV-B[d]	
		Value	Unit	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL
Vanadium											
1–5	RM ≥ 10[e]	0.035–0.068	mg/kg bw/day	0.27	2.7	na	na	0.13–0.25	0.013–0.025	na	na
	site wide[f]	0.042–0.081	mg/kg bw/day					0.16–0.29	0.016–0.029		
Zinc											
1–5	RM ≥ 10[e]	2.4–4.6	mg/kg bw/day	160	320	9.6	411	0.015–0.029	0.0075–0.014	0.25–0.48	0.0059–0.11
	site wide[f]	2.7–5.1	mg/kg bw/day					0.017–0.032	0.0084–0.016	0.28–0.53	0.0065–0.012
Benzo(a)pyrene											
1–5	RM ≥ 10[e]	4.1–5.7	µg/kg bw/day	1,000	10,000	na	na	0.0041–0.0057	0.00041–0.00057	na	na
	site wide[f]	3.9–5.7	µg/kg bw/day					0.0039–0.0057	0.00039–0.00057		
Total HPAHs											
1–5	RM ≥ 10[e]	42–57	µg/kg bw/day	na	na	620	3,100	na	na	0.069–0.093	0.010–0.019
	site wide[f]	39–58	µg/kg bw/day					na	na	0.064–0.093	0.013–0.021

Table 9-14. Mink dietary HQs

Diet Scenario[b]	Area	Dose		Range of TRVs[a]				Range of HQs[a]			
				TRV-A[c]		TRV-B[d]		HQ Based on TRV-A[c]		HQ Based on TRV-B[d]	
		Value	Unit	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL
Total PCBs											
1	RM ≥ 10[e]	109	µg/kg bw/day	80	96	69	82	**1.4**	**1.1**	**1.6**	**1.3**
	site wide[f]	90	µg/kg bw/day					**1.1**	0.94	**1.3**	**1.1**
2	RM ≥ 10[e]	112	µg/kg bw/day					**1.4**	**1.2**	**1.6**	**1.4**
	site wide[f]	95	µg/kg bw/day					**1.2**	0.99	**1.4**	**1.2**
3	RM ≥ 10[e]	255	µg/kg bw/day					**3.2**	**2.7**	**3.7**	**3.1**
	site wide[f]	208	µg/kg bw/day					**2.6**	**2.2**	**3.0**	**2.5**
4	RM ≥ 10[e]	258	µg/kg bw/day					**3.2**	**2.7**	**3.7**	**3.1**
	site wide[f]	213	µg/kg bw/day					**2.7**	**2.2**	**3.1**	**2.6**
5	RM ≥ 10[e]	216	µg/kg bw/day					**2.7**	**2.3**	**3.1**	**2.6**
	site wide[f]	181	µg/kg bw/day					**2.3**	**1.9**	**2.6**	**2.2**

Table 9-14. Mink dietary HQs

Diet Scenario[b]	Area	Dose		Range of TRVs[a]				Range of HQs[a]			
				TRV-A[c]		TRV-B[d]		HQ Based on TRV-A[c]		HQ Based on TRV-B[d]	
		Value	Unit	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL
PCB TEQ - mammal											
1	RM ≥ 10[e]	109	µg/kg bw/day	2.6	8.8	0.08	2.2	0.43	0.13	**14**	0.51
	site wide[f]	90	µg/kg bw/day					0.41	0.12	**13**	0.49
2	RM ≥ 10[e]	112	µg/kg bw/day					0.46	0.14	**15**	0.54
	site wide[f]	95	µg/kg bw/day					0.44	0.13	**14**	0.52
3	RM ≥ 10[e]	255	µg/kg bw/day					0.94	0.28	**30**	**1.1**
	site wide[f]	208	µg/kg bw/day					0.89	0.26	**29**	**1.1**
4	RM ≥ 10[e]	258	µg/kg bw/day					0.97	0.29	**31**	**1.1**
	site wide[f]	213	µg/kg bw/day					0.92	0.27	**30**	**1.1**
5	RM ≥ 10[e]	216	µg/kg bw/day					0.87	0.26	**28**	**1.0**
	site wide[f]	181	µg/kg bw/day					0.84	0.25	**27**	0.99

Table 9-14. Mink dietary HQs

Diet Scenario[b]	Area	Dose		Range of TRVs[a]				Range of HQs[a]			
				TRV-A[c]		TRV-B[d]		HQ Based on TRV-A[c]		HQ Based on TRV-B[d]	
		Value	Unit	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL
PCDD/PCDF TEQ - mammal											
1	RM ≥ 10[e]	9.3	ng/kg bw/day	2.6	8.8	0.08	2.2	**3.6**	**1.1**	**116**	**4.2**
	site wide[f]	7	ng/kg bw/day					**2.7**	0.79	**87**	**3.2**
2	RM ≥ 10[e]	9.5	ng/kg bw/day					**3.7**	**1.1**	**119**	**4.3**
	site wide[f]	7.4	ng/kg bw/day					**2.8**	0.84	**92**	**3.4**
3	RM ≥ 10[e]	19	ng/kg bw/day					**7.3**	**2.2**	**238**	**8.6**
	site wide[f]	13	ng/kg bw/day					**5.1**	**1.5**	**166**	**6.0**
4	RM ≥ 10[e]	19	ng/kg bw/day					**7.4**	**2.2**	**240**	**8.7**
	site wide[f]	14	ng/kg bw/day					**5.3**	**1.6**	**171**	**6.2**
5	RM ≥ 10[e]	17	ng/kg bw/day					**6.7**	**2.0**	**217**	**7.9**
	site wide[f]	13	ng/kg bw/day					**5.0**	**1.5**	**162**	**5.9**

Table 9-14. Mink dietary HQs

Diet Scenario[b]	Area	Dose Value	Dose Unit	Range of TRVs[a] TRV-A[c] NOAEL	TRV-A[c] LOAEL	TRV-B[d] NOAEL	TRV-B[d] LOAEL	Range of HQs[a] HQ Based on TRV-A[c] NOAEL	HQ Based on TRV-A[c] LOAEL	HQ Based on TRV-B[d] NOAEL	HQ Based on TRV-B[d] LOAEL
Total TEQ - mammal[g]											
1	RM ≥ 10[e]	10	ng/kg bw/day	2.6	8.8	0.08	2.2	**4.0**	**1.2**	**129**	**4.7**
	site wide[f]	9	ng/kg bw/day					**3.5**	**1.0**	**113**	**4.1**
2	RM ≥ 10[e]	11	ng/kg bw/day					**4.1**	**1.2**	**132**	**4.8**
	site wide[f]	9.5	ng/kg bw/day					**3.6**	**1.1**	**118**	**4.3**
3	RM ≥ 10[e]	21	ng/kg bw/day					**8.2**	**2.4**	**267**	**9.7**
	site wide[f]	18	ng/kg bw/day					**6.9**	**2.1**	**226**	**8.2**
4	RM ≥ 10[e]	22	ng/kg bw/day					**8.3**	**2.5**	**271**	**9.9**
	site wide[f]	18	ng/kg bw/day					**7.1**	**2.1**	**231**	**8.4**
5	RM ≥ 10[e]	20	ng/kg bw/day					**7.5**	**2.2**	**244**	**8.9**
	site wide[f]	17	ng/kg bw/day					**6.6**	**1.9**	**213**	**7.7**
Dieldrin											
1–5	RM ≥ 10[e]	1.7–3.9	µg/kg bw/day	15	30	15	30	0.11–0.26	0.056–0.13	0.11–0.26	0.056–0.13
	site wide[f]	1.3–3.0	µg/kg bw/day					0.087–0.20	0.043–0.10	0.087–0.20	0.043–0.10

Bold identifies HQs ≥ 1.0.

Shaded cells identify LOAEL HQs ≥ 1.0.

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

[b] Dietary doses were calculated using five diet scenarios. Scenario 1 included 16.5% blue crab and 34% ≤ 30-cm fish (49.5% of the diet was terrestrial, which was assumed to be zero). Scenario 2 included 16.5% blue crab, 31% ≤ 30-cm fish, and 3% > 30-cm fish (49.5% of the diet was terrestrial, which was assumed to be zero). Scenario 3 included 16.5% blue crab and 83.5% ≤ 30-cm fish. Scenario 4 included 16.5% blue crab, 80.5% ≤ 30-cm fish, and 3% > 30-cm fish. Scenario 5 included 33.5% blue crab, 63.5% ≤ 30-cm fish, and 3% > 30-cm fish.

[c] TRVs were derived from the primary literature based on the process identified in Section 9.1.3.1.

[d] TRVs were based on USEPA's revised draft of the LPR restoration project FFS (Louis Berger et al. 2014), first draft of the LPR restoration project FFS (Malcolm Pirnie 2007b), or LPR restoration project PAR (Battelle 2005).

[e] The RM ≥ 10 exposure area scenario includes an exposure area at and upstream of RM 10 for surface water, sediment, and SFF; all other fish (i.e., NFF ≤ 30 cm and fish > 30 cm) and blue crab have a site-wide exposure area.

[f] The site-wide exposure area includes site-wide exposure for fish, blue crab, and sediment, as well as RM ≥ 4 for surface water (i.e., includes only freshwater).

[g] Total TEQ - mammal is equal to the sum of PCB TEQ - mammal and PCDD/PCDF TEQ - mammal on a sample-by-sample basis; however, this is not necessarily the case for the sum of dietary doses in which UCLs were used.

BERA – baseline ecological risk assessment
bw – body weight
COPEC – chemical of potential ecological concern
FFS – focused feasibility study
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
LPR – Lower Passaic River
LPRSA – Lower Passaic River study Area
na – not applicable (no TRV available)
NFF – non-small forage fish
NJDEP – New Jersey Department of Environmental Protection
NOAEL – no-observed-adverse-effect level
PAR – pathways analysis report
PCB – polychlorinated biphenyl
PCDD - polychlorinated dibenzo-*p*-dioxin
PCDF – polychlorinated dibenzofuran
RM – river mile
SFF – small forage fish
TEQ – toxic equivalent
TRV – toxicity reference value
UCL – upper confidence limit on the mean
USEPA – US Environmental Protection Agency

Total PCBs

Mink dietary LOAEL HQs for total PCBs ranged from 0.94 to 2.6 on a site-wide basis, and ranged from 1.1 to 3.1 for RM ≥ 10. The highest HQs were associated with diet Scenarios 3 (16.5% blue crab and 83.5% ≤ 30 cm fish) and 4 (16.5% blue crab, 80.5% ≤ 30 cm fish, and 3% > 30 cm fish).

PCB TEQ - mammal, PCDD/PCDF TEQ - mammal, and Total TEQ - mammal

Mink dietary LOAEL HQs for PCB TEQ - mammal ranged from 0.12 to 1.1 on a site-wide basis, and ranged from 0.13 to 1.1 for RM ≥ 10. LOAEL HQs for PCDD/PCDF TEQ - mammal ranged from 0.79 to 6.2 on a site-wide basis, and ranged from 1.1 to 8.7 for RM ≥ 10. LOAEL HQs for total TEQ - mammal ranged from 1.0 to 8.4 for RM ≥ 10. The highest HQs were generally associated with diet Scenarios 3 (16.5% blue crab and 83.5% ≤ 30 cm fish) and 4 (16.5% blue crab, 80.5% ≤ 30 cm fish, and 3% > 30 cm fish).

River Otter

River otter dietary LOAEL HQs were ≥ 1.0 for total PCBs, PCB TEQ - mammal, PCDD/PCDF TEQ - mammal, and total TEQ - mammal (Table 9-15). Like the mink HQs, the RM ≥ 10 exposure scenario resulted in higher HQs for PCBs and TEQ - mammal than the site-wide exposure area scenario, which was driven by the EPCs for ≤ 30-cm fish. The EPCs for ≤ 30-cm fish were influenced by higher PCB and TEQ - mammal concentrations in perch and the percentage of SFF (with lower concentrations). The ≤ 30-cm fish for the site-wide exposure scenario were composed of 45% SFF, whereas the RM ≥ 10 exposure area scenario (i.e., SFF from RM ≥ 10 and NFF site wide) had 19% SFF. A higher percentage of SFF in the EPC dataset resulted in lower total PCB and TEQ- mammal EPCs and HQs, because the SFF effectively reduced the influence of the perch in the EPC.

Table 9-15. River otter dietary HQs

Diet Scenario[b]	Area	Dose Value	Dose Units	Range of TRVs[a] TRV-A[c] NOAEL	TRV-A[c] LOAEL	TRV-B[d] NOAEL	TRV-B[d] LOAEL	Range of HQs[a] HQ Based on TRV-A[c] NOAEL	HQ Based on TRV-A[c] LOAEL	HQ Based on TRV-B[d] NOAEL	HQ Based on TRV-B[d] LOAEL
Arsenic											
1–2	RM ≥ 10[e]	0.064	mg/kg bw/day	2.6	5.4	0.32	4.7	0.025	0.012	0.20	0.014
	site wide[f]	0.076	mg/kg bw/day					0.029	0.014	0.24	0.016
Cadmium											
1–2	RM ≥ 10[e]	0.012	mg/kg bw/day	3.5	13	0.06	2.64	0.0034–0.0035	0.00093	0.020	0.0046
	site wide[f]	0.01	mg/kg bw/day					0.003	0.00080–0.00081	0.17–0.18	0.0040
Copper											
1–2	RM ≥ 10[e]	1.7	mg/kg bw/day	18	26	3.4	6.8	0.094–0.096	0.065–0.066	0.50–0.51	0.25
	site wide[f]	1.5-1.6	mg/kg bw/day					0.085–0.086	0.059–0.060	0.45–0.46	0.22–0.23
Lead											
1–2	RM ≥ 10[e]	0.36-0.37	mg/kg bw/day	11	90	0.71	7.0	0.033–0.034	0.0041	0.51–0.52	0.052–0.053
	site wide[f]	0.40-0.41	mg/kg bw/day					0.036–0.037	0.0044–0.0045	0.56–0.57	0.057–0.058
Mercury											
1–2	RM ≥ 10[e]	23-24	µg/kg bw/day	160	250	16	27	0.15	0.093–0.096	**1.5**	0.86–0.89
	site wide[f]	23-24	µg/kg bw/day					0.15	0.093–0.096	**1.5**	0.86–0.89
Methylmercury											
1–2	RM ≥ 10[e]	21	µg/kg bw/day	160	250	16	27	0.13	0.084–0.086	**1.3**	0.77–0.79
	site wide[f]	18-19	µg/kg bw/day					0.11–0.12	0.072–0.075	**1.1–1.2**	0.67–0.70
Nickel											
1–2	RM ≥ 10[e]	1.9-2.0	mg/kg bw/day	40	80	0.133	31.6	0.047–0.049	0.023–0.025	**14–15**	0.059–0.063
	site wide[f]	0.75-0.79	mg/kg bw/day					0.019–0.020	0.0094–0.0099	**5.7–5.9**	0.024–0.025

Table 9-15. River otter dietary HQs

Diet Scenario[b]	Area	Dose		Range of TRVs[a]				Range of HQs[a]			
				TRV-A[c]		TRV-B[d]		HQ Based on TRV-A[c]		HQ Based on TRV-B[d]	
		Value	Units	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL
Selenium											
1–2	RM ≥ 10[e]	0.15-0.16	mg/kg bw/day	0.016	0.16	0.05	1.21	**9.6–9.9**	0.96–0.99	**3.1–3.2**	0.13
	site wide[f]	0.14	mg/kg bw/day					**8.8–9.0**	0.88–0.90	**2.8–2.9**	0.12
Vanadium											
1–2	RM ≥ 10[e]	0.077-0.080	mg/kg bw/day	0.27	2.7	na	na	0.28–0.29	0.028–0.029	na	na
	site wide[f]	0.092-0.095	mg/kg bw/day					0.34–0.35	0.034–0.035		
Zinc											
1–2	RM ≥ 10[e]	5.0-5.1	mg/kg bw/day	160	320	9.6	411	0.031–0.032	0.016	0.52–0.53	0.012
	site wide[f]	5.8	mg/kg bw/day					0.036	0.018	0.60–0.61	0.014
Benzo(a)pyrene											
1–2	RM ≥ 10[e]	6.4-6.6	µg/kg bw/day	1,000	10,000	na	na	0.0064–0.0066	0.00064–0.00066	na	na
	site wide[f]	6.4-6.6	µg/kg bw/day					0.0064–0.0066	0.00064–0.00066		
Total HPAHs											
1–2	RM ≥ 10[e]	66-67	µg/kg bw/day	na	na	620	3,100	na	na	0.11	0.021–0.022
	site wide[f]	66-67	µg/kg bw/day					na	na	0.11	0.021–0.022
Total PCBs											
1	RM ≥ 10[e]	300	µg/kg bw/day	80	96	69	82	**3.8**	**3.1**	**4.4**	**3.7**
	site wide[f]	245	µg/kg bw/day					**3.1**	**2.6**	**3.6**	**3.0**
2	RM ≥ 10[e]	306	µg/kg bw/day					**3.8**	**3.2**	**4.4**	**3.7**
	site wide[f]	254	µg/kg bw/day					**3.2**	**2.6**	**3.7**	**3.1**

Table 9-15. River otter dietary HQs

Diet Scenario[b]	Area	Dose		Range of TRVs[a]				Range of HQs[a]			
				TRV-A[c]		TRV-B[d]		HQ Based on TRV-A[c]		HQ Based on TRV-B[d]	
		Value	Units	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL
PCB TEQ - mammal											
1	RM ≥ 10[e]	2.9	ng/kg bw/day	2.6	8.8	0.08	2.2	**1.1**	0.32	**36**	**1.3**
	site wide[f]	2.7	ng/kg bw/day					**1.0**	0.31	**34**	**1.2**
2	RM ≥ 10[e]	3	ng/kg bw/day					**1.2**	0.34	**37**	**1.4**
	site wide[f]	2.9	ng/kg bw/day					**1.1**	0.33	**36**	**1.3**
PCDD/PCDF TEQ - mammal											
1	RM ≥ 10[e]	22	ng/kg bw/day	2.6	8.8	0.08	2.2	**8.6**	**2.5**	**278**	**10**
	site wide[f]	15	ng/kg bw/day					**6.0**	**1.8**	**194**	**7.0**
2	RM ≥ 10[e]	23	ng/kg bw/day					**8.7**	**2.6**	**283**	**10**
	site wide[f]	16	ng/kg bw/day					**6.3**	**1.9**	**204**	**7.4**
Total TEQ - mammal[g]											
1	RM ≥ 10[e]	25	ng/kg bw/day	2.6	8.8	0.08	2.2	**9.6**	**2.8**	**313**	**11**
	site wide[f]	21	ng/kg bw/day					**8.1**	**2.4**	**264**	**9.6**
2	RM ≥ 10[e]	26	ng/kg bw/day					**9.9**	**2.9**	**320**	**12**
	site wide[f]	22	ng/kg bw/day					**8.4**	**2.5**	**274**	**10**
Dieldrin											
1–2	RM ≥ 10[e]	4.6	µg/kg bw/day	15	30	15	30	0.31	0.15	0.31	0.15
	site wide[f]	3.5-3.6	µg/kg bw/day					0.23–0.24	0.12	0.23–0.24	0.12

Bold identifies HQs ≥ 1.0.

Shaded cells identify LOAEL HQs ≥ 1.0.

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

[b] Dietary doses were calculated using two diet scenarios. Scenario 1 dietary doses were calculated using a diet with 15% blue crab and 85% ≤ 30-cm fish. Scenario 2 included 15% blue crab, 80% ≤ 30-cm fish, and 5% > 30-cm fish.

[c] TRVs were derived from the primary literature based on the process identified in Section 9.1.3.1.

[d] TRVs were derived based on USEPA's revised draft of the LPR restoration project FFS (Louis Berger et al. 2014), first draft of the LPR restoration project FFS (Malcolm Pirnie 2007b), or LPR restoration project PAR (Battelle 2005).

[e] The RM ≥ 10 exposure area scenario includes an exposure area at and upstream of RM 10 for surface water, sediment, and SFF; all other fish (i.e., NFF ≤ 30 cm and fish > 30 cm) and blue crab have a site-wide exposure area.

[f] The site-wide exposure area includes site-wide exposure for fish, blue crab, and sediment, as well as RM ≥ 4 for surface water (i.e., includes only freshwater).

[g] Total TEQ - mammal is equal to the sum of PCB TEQ - mammal and PCDD/PCDF TEQ - mammal on a sample-by-sample basis; however, this is not necessarily the case for the sum of dietary doses in which UCLs were used.

BERA – baseline ecological risk assessment
bw – body weight
COPEC – chemical of potential ecological concern
FFS – focused feasibility study
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
LPR – Lower Passaic River
LPRSA – Lowr Passaic River study Area
na – not applicable (no TRV available)
NFF – non-small forage fish
NJDEP – New Jersey Department of Environmental Protection
NOAEL – no-observed-adverse-effect level
PAR – pathways analysis report
PCB – polychlorinated biphenyl
PCDD - polychlorinated dibenzo-*p*-dioxin
PCDF – polychlorinated dibenzofuran
RM – river mile
SFF – small forage fish
TEQ – toxic equivalent
TRV – toxicity reference value
UCL – upper confidence limit on the mean
USEPA – US Environmental Protection Agency

Total PCBs

River otter dietary LOAEL HQs for total PCBs ranged from 2.6 to 3.1 on a site-wide basis, and ranged from 3.1 to 3.7 for RM ≥ 10. HQs were slightly higher for diet Scenario 2 (includes fish > 30 cm) than for diet Scenario 1 (does not include fish > 30 cm).

PCB TEQ - mammal, PCDD/PCDF TEQ - mammal, and total TEQ - mammal

River otter dietary LOAEL HQs for PCB TEQ - mammal ranged from 0.31 to 1.3 on a site-wide basis, and ranged from 0.32 to 1.4 for RM ≥ 10. LOAEL HQs for PCDD/PCDF TEQ - mammal ranged from 1.8 to 7.4 on a site-wide basis, and ranged from 2.5 to 10 for RM ≥ 10. LOAEL HQs for total TEQ - mammal ranged from 2.4 to 10 on a site-wide basis, and ranged from 2.8 to 12 for RM ≥ 10. HQs were slightly higher for diet Scenario 2 (includes fish > 30 cm) than for diet Scenario 1 (does not include fish > 30 cm).

9.1.4.2 Uncertainties in risk characterization

This section discusses uncertainties that could affect HQ calculations for mammals. It discusses uncertainties in the diet composition and exposure area assumptions. Uncertainties related to TRVs are discussed in Section 9.1.3. An analysis was conducted to evaluate uncertainties associated with exposure assumptions and EPC calculations. In addition, to address a combination of exposure assumption uncertainties simultaneously, a sensitivity analysis conducted for river otter and mink is summarized in this section and presented in detail in Appendix H. This section also discusses the results of a habitat analysis conducted for mink to provide a more detailed evaluation of mink exposure in the LPRSA based on the availability of habitat; details on this habitat analysis are presented in detail in Appendix I.

Dietary Composition Uncertainties

For mink, two general diets were evaluated: Scenario 1, which assumed that terrestrial prey make up approximately one-half of the diet (with the terrestrial portion set equal to zero because terrestrial concentrations are not available), and Scenario 2, which assumed that the diet is composed of all aquatic prey (Table 9-4). Two aquatic/terrestrial diets were evaluated in the HQ calculations (i.e., mink diet Scenarios 1 and 2); Scenario 2 included large fish while Scenario 1 did not. Three aquatic prey-only diets were evaluated in HQ calculations (i.e., mink diet Scenarios 3 through 5). Mink diet Scenarios 3 and 4 had the same proportions of crab (i.e., 16.5%) and fish (83.5%); Scenario 4 included large fish while Scenario 3 did not. Mink diet Scenario 5 included a greater percentage of crab (i.e., 33.5%) than the other aquatic prey-only diets and included large fish. Aquatic/terrestrial diets (i.e., Scenarios 1 and 2) resulted in lower HQs than more conservative diets that assumed that mink feed exclusively on aquatic prey (i.e., Scenarios 3 through 5) (Table 9-14).

For river otter, two dietary exposure scenarios were evaluated in HQ calculations. Diet Scenario 2 included large fish (> 30 cm), while diet Scenario 1 did not (Table 9-4). Diet Scenario 2 resulted in slightly higher HQs for river otter than did diet Scenario 2 (Table 9-15).

Exposure Area Uncertainties

Two exposure areas (i.e., at and upstream of RM 10 and site wide; Table 9-5) were used for mink and river otter. It was assumed that mink and river otter habitat is limited to areas with more vegetation and that are less disturbed/developed than other areas in the vicinity (i.e., RM ≥ 10). However, the conservative assumption that the entire LPRSA offers habitat suitable for mink and river otter was also evaluated (i.e., site wide). The exposure areas for prey for both scenarios were assumed to be site wide with the exception of SFF, which were limited to at and upstream of RM 10 for the RM ≥ 10 scenario because of the their localized movements. The RM ≥ 10 scenario resulted in higher HQs for PCBs and TEQ - mammal than the site-wide scenario for both mink and river otter. These results were driven by the EPCs for small fish (≤ 30-cm fish), which were influenced by higher PCB and TEQ - mammal concentrations in perch and the percentage of SFF (with lower concentrations). The site-wide exposure area scenario had 45% SFF, whereas the RM ≥ 10 exposure area scenario (i.e., SFF from RM ≥ 10 and NFF site wide) had 19% SFF. A greater percentage of SFF in the EPC dataset for the site-wide exposure scenario resulted in lower total PCB and TEQ - mammal EPCs and HQs, because the SFF effectively reduced the influence of the perch on the EPC.

Exposure Assumptions and EPC Uncertainties

A quantitative evaluation was conducted by varying certain exposure parameter assumptions and EPC calculations to determine the effect on HQs. The exposure assumptions and EPC uncertainties that were evaluated are as follows:

- **Body weight –** The average of the male and female body weights was used in the HQ calculations. The effect on HQs of using the maximum and minimum male and female body weights reported in USEPA's *Wildlife Exposure Factors Handbook* (USEPA 1993) was evaluated.
- **Sediment ingestion rate –** SIRs were based on an estimate of 2% of the FIR, based on best professional judgment. The effect on HQs of using alternative SIRs within a reasonable range to bracket the original estimate (1 and 4%) was evaluated.
- **Food ingestion rate –** FIRs used for mink and river otter were approximately 14 and 16%, respectively, of body weight. The effect on HQs of altering the FIRs to 12 and 22%, respectively, of body weight (the range provided by USEPA (1993) for mink), and to 14 and 18%, respectively, of body weight for river otter (± 2%) was evaluated.

- **Dietary proportions** – In mink diet Scenarios 2, 4, and 5 and river otter diet Scenario 2, the portion of the diet that consisted of > 30-cm fish was 3%. The effect on HQs of using a portion consisting of 10 to 20% > 30-cm fish in the mink and river otter diets was evaluated.
- **Fish EPCs –** Fish EPCs were calculated by size class for mink and river otter diet scenarios for both site-wide and RM ≥ 10 exposure areas (only SFF were limited to RM ≥ 10; all other fish had a site-wide exposure). Four variations in fish EPCs were explored:
  - **All sizes –** The effect on HQs of using a single EPC calculated from all fish (i.e., not divided by size class) was evaluated.
  - **Weighted by site-wide abundance by size class –** The effect on HQs of using fish EPCs calculated using a weighted approach based on the site-wide abundance of various fish groups by size class was evaluated (see Appendix H for more details on abundance calculations).
  - **Weighted by RM ≥ 10 abundance by size class –** The effect on HQs of using fish EPCs calculated using a weighted approach based on the abundance of various fish groups at and upstream of RM 10 by size class was evaluated (see Appendix H for more details on abundance calculations).
  - **RM ≥ 10 for all fish –** The effect on HQs of using fish EPCs calculated from RM ≥ 10 for all fish was evaluated.
- **Site use factor –** An SUF of 1 was used for river otter and mammal exposure. The effect on HQs of using an alternative SUF of 0.5 was evaluated.
- **Exposure area** – HQs were based on exposure areas of RM ≥ 10 (i.e., SFF and sediment and water EPCs restricted to RM ≥ 10) and the entire LPRSA for both mink and river otter. The effect on HQs of restricting all fish EPCs to RM ≥ 10, in addition to restricting sediment and water EPCs to RM ≥ 10, was evaluated.
- **Treatment of non-detects for EPCs –** The concentrations of congeners that were not detected were assumed to be zero when calculating total PCBs, and TEQs - mammal were calculated using USEPA's TEQ calculator (USEPA 2014) using the Kaplan-Meier method. The effect on HQs of using one-half the DL or the full DL was evaluated for total PCBs. The effect on TEQ- mammal HQs of using zero, one-half the DL, or the full DL was evaluated for total TEQ - mammal.

The effects of these uncertainties on HQ calculations are presented in Table 9-16 for one diet scenario for mink and river otter.[128]

[128] Mink diet scenario 4 consisting of aquatic prey only (16.5% blue crab, 80.5% ≤ 30 cm fish, 3% fish > 30 cm) for the site-wide exposure area and river otter diet scenario 2 (15% blue crab, 80% ≤ 30 cm fish, 5% fish > 30 cm) for the site-wide exposure area was evaluated.

Table 9-16. Mammal dietary HQs based on uncertainties in exposure parameters and EPCs

Uncertainty	Parameter Values/Assumptions		Range of LOAEL HQs[a]															
			Total PCBs				PCB TEQ - Mammal				PCDD/PCDF TEQ - Mammal				Total TEQ - Mammal			
			HQ Based on TRV-A[b]		HQ Based on TRV-B[c]		HQ Based on TRV-A[b]		HQ Based on TRV-B[c]		HQ Based on TRV-A[b]		HQ Based on TRV-B[c]		HQ Based on TRV-A[b]		HQ Based on TRV-B[c]	
Uncertainty	Original	Adjusted	Orig.	Adj.	Orig.	Adj.	Orig.	Adj.	Orig.	Adj.	Orig.	Adj.	Orig.	Adj.	Orig.	Adj.	Orig.	Adj.
Mink[c]																		
Body weight	1.0 kg	1.7 kg		**2.2**		**2.6**		0.27		**1.1**		**1.6**		**6.3**		**2.1**		**8.5**
		0.55 kg		**2.3**		**2.7**		0.28		**1.1**		**1.6**		**6.5**		**2.2**		**8.7**
SIR	2% of FIR	1% of FIR		**2.2**		**2.6**		0.27		**1.1**		**1.5**		**5.9**		**2.0**		**8.1**
		4% of FIR		**2.2**		**2.6**		0.27		**1.1**		**1.7**		**6.8**		**2.2**		**9.0**
FIR	14% of body weight	12% of body weight		**1.9**		**2.2**		0.23		0.94		**1.3**		**5.3**		**1.8**		**7.2**
		22% of body weight		**3.5**		**4.1**		0.43		**1.7**		**2.4**		**9.8**		**3.3**		**13**
Diet proportions	3% fish > 30 cm	10% fish > 30 cm		**2.3**		**2.7**		0.29		**1.2**		**1.7**		**6.7**		**2.2**		**8.8**
		20% fish > 30 cm		**2.5**		**2.9**		0.32		**1.3**		**1.8**		**7.3**		**2.4**		**9.5**
Fish EPCs	fish EPCs calculated by size class (site-wide exposure area)	fish EPCs calculated by including all data (i.e., not by size class)		**3.0**		**3.5**		0.36		**1.4**		**2.7**		**11**		**3.0**		**12**
		fish EPCs calculated according to site-wide abundance by size class[d]	**2.2**	**1.1**	**2.6**	**1.3**	0.27	0.16	**1.1**	0.65	**1.6**	**1.1**	**6.2**	**4.5**	**2.1**	**1.3**	**8.4**	**5.2**
		fish EPCs calculated according abundance for RM ≥ 10 by size class[e]		**1.2**		**1.3**		0.17		0.68		0.94		**4.3**		**1.1**		**4.3**
		fish EPCs calculated by size class for RM ≥ 10 for all fish		**1.2**		**1.4**		0.19		0.74		**1.0**		**4.0**		**1.1**		**4.4**
SUF	1	0.5		**1.1**		**1.3**		0.14		0.55		0.78		**3.1**		**1.0**		**4.2**
Treatment of non-detects	DL = 0 for non-detects for PCB congeners and use of Kaplan-Meier method in USEPA's TEQ calculator (USEPA 2014) for TEQs - mammal	use of DL = 0 (for TEQ - mammal),one-half the DL, or the full DL for non-detects[f]		**2.2**		**2.6**		0.27		**1.1**		**1.6**		**6.2**		**2.1**		**8.4**

Table 9-16. Mammal dietary HQs based on uncertainties in exposure parameters and EPCs

Uncertainty	Parameter Values/Assumptions		Range of LOAEL HQs[a]															
			Total PCBs				PCB TEQ - Mammal				PCDD/PCDF TEQ - Mammal				Total TEQ - Mammal			
			HQ Based on TRV-A[b]		HQ Based on TRV-B[c]		HQ Based on TRV-A[b]		HQ Based on TRV-B[c]		HQ Based on TRV-A[b]		HQ Based on TRV-B[c]		HQ Based on TRV-A[b]		HQ Based on TRV-B[c]	
	Original	Adjusted	Orig.	Adj.	Orig.	Adj.	Orig.	Adj.	Orig.	Adj.	Orig.	Adj.	Orig.	Adj.	Orig.	Adj.	Orig.	Adj.
River otter[g]																		
Body weight	8.0 kg	10.4 kg	2.6	**2.5**	3.1	**2.9**	0.33	0.31	1.3	**1.2**	1.9	**1.8**	7.4	**7.0**	2.5	**2.4**	10	**9.4**
		4.74 kg		**2.8**		**3.2**		0.34		**1.4**		**1.9**		**7.8**		**2.6**		**10**
SIR	2% of FIR	1% of FIR		**2.6**		**3.1**		0.32		**1.3**		**1.8**		**7.1**		**2.4**		**9.6**
		4% of FIR		**2.7**		**3.1**		0.33		**1.3**		**2.0**		**8.1**		**2.7**		**11**
FIR	16% of body weight	14% of body weight		**2.3**		**2.7**		0.28		**1.1**		**1.6**		**6.4**		**2.1**		**8.6**
		18% of body weight		**2.9**		**3.4**		0.36		**1.4**		**2.1**		**8.2**		**2.8**		**11**
Diet proportions	3% fish > 30 cm	10% fish > 30 cm		**2.7**		**3.2**		0.34		**1.4**		**1.9**		**7.8**		**2.6**		**10**
		20% fish > 30 cm		**2.9**		**3.4**		0.38		**1.5**		**2.1**		**8.5**		**2.8**		**11**
Fish EPCs	fish EPCs calculated by size class (site-wide exposure area)	fish EPCs calculated by including all data (i.e., not by size class)		**3.6**		**4.2**		0.42		**1.7**		**3.2**		**13**		**3.5**		**14**
		fish EPCs calculated by weighting according to site-wide abundance by size class[e]		**1.4**		**1.7**		0.20		0.79		**1.4**		**5.6**		**1.6**		**5.6**
		fish EPCs calculated by weighting according abundance RM ≥ 10 by size class[d]		**1.5**		**1.7**		0.21		0.85		**1.2**		**5.0**		**1.4**		**5.7**
		fish EPCs calculated by size class for RM ≥ 10 for all fish		**1.4**		**1.7**		0.23		0.91		**1.2**		**4.8**		**1.3**		**5.3**
SUF	1	0.5		**1.3**		**1.5**		0.16		0.65		0.93		**3.7**		**1.2**		**5.0**
Treatment of non-detects	DL = 0 for non-detects for PCB congeners and use of Kaplan-Meier method in USEPA's TEQ calculator (USEPA 2014) for TEQs - mammal	use of DL = 0 (for only TEQ - mammal),one-half the DL, or the full DL for non-detects[f]		**2.6**		**3.1**		0.27		**1.3**		**1.9**		**7.4**		**2.5**		**10**

Bold identifies HQs ≥ 1.0.

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

[b] TRVs were derived from the primary literature based on the process identified in Section 9.1.3.1.

[c] TRVs were derived based on USEPA's revised draft of the LPR restoration project FFS (Louis Berger et al. 2014) or first draft of the LPR restoration project FFS (Malcolm Pirnie 2007b).

[d] Both original and adjusted HQs presented in this table are based on mink diet Scenario 4 (16.5% blue crab, 80.5% ≤ 30-cm fish, and 3% fish > 30 cm) for the site-wide exposure area.

[e] See Appendix H for abundance calculations.

[f] LOAEL HQs are the same regardless of treatment of non-detected values as one-half the DL or as the full DL (and DL = 0 for TEQ - mammal).

[g] Both original and adjusted HQs presented in this table are based on river otter diet Scenario 2 (15% blue crab, 80% ≤ 30-cm fish, and 5% fish > 30 cm) for the site-wide exposure area.

BERA – baseline ecological risk assessment
DL – detection limit
EPC – exposure point concentration
FFS – focused feasibility study
FIR – food ingestion rate
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
LPR – Lower Passaic River
LPRSA – Lower Passaic River study Area
NJDEP – New Jersey Department of Environmental Protection
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
PCDD – polychlorinated dibenzo-*p*-dioxin
PCDF – polychlorinated dibenzofuran
RM – river mile
SFF – small forage fish
SIR – sediment ingestion rate
SUF – site use factor
TEQ – toxic equivalent
USEPA – US Environmental Protection Agency

Table 9-17 provides a summary of the percent differences in HQs based on the results presented in Table 9-16. The changes in LOAEL HQs are as follows:

- **Body weight –** Changes in the body weight for mink and river otter resulted in relatively small changes to LOAEL HQs (i.e., maximum of 0.4 HQ units).
- **Sediment ingestion rate –** SIRs resulted in relatively small changes to LOAEL HQs (i.e., maximum of 0.7 HQ units).
- **Food ingestion rate –** The FIR adjustments resulted in a maximum of 1.0 HQ unit change for river otter. The FIRs for mink resulted in larger changes to LOAEL HQs (maximum of 4.6 HQ units), particularly when the FIR was changed from 14 to 22% of the body weight (resulting in increases in HQs). It is unlikely that mink would consume food at maximum food ingestion; therefore, the average FIR was selected, as it is more likely to represent actual food ingestion than the maximum.
- **Dietary proportions** – Changing the percentage of > 30-cm fish in the diet to 10 or 20% increased HQs (maximum of 0.5 and 1.1 HQ units, respectively). The EPCs for > 30-cm fish are greater than those for ≤ 30-cm fish.
- **Fish EPCs –**Four variations in fish EPCs were evaluated:
  - **All sizes –**When the selected size classes (i.e., ≤ 30 cm and > 30 cm) were eliminated and all fish were grouped together, LOAEL HQs increased (maximum of 5.6 HQ units). The percentage of > 30-cm fish was 53% when all fish were grouped together. High percentages of > 30-cm fish in the mink and river otter diets are not supported by the literature (see Section 9.1.2.3). The proportions of large fish in the mink and river otter diets were further evaluated in the sensitivity analysis (Appendix H).
  - **Weighted by site-wide abundance by size class –** Weighting fish EPCs for each size class (i.e., ≤ 30 cm and > 30 cm) by site-wide abundance and by abundance for RM ≥ 10 decreased LOAEL HQs (maximum of 4.4 HQ units). For ≤ 30-cm fish, which make up the majority of the mink and river otter diets, the most abundant species group was SFF (83% for the site-wide exposure area and 71% for the RM ≥ 10 exposure area; Appendix H). PCB TEQ - mammal concentrations in SFF are less than in < 30 cm fish, and as a result weighting by abundance reduces the HQs. The HQs for PCB TEQ - mammal that were ≥ 1.0 were adjusted to < 1.0 for mink and river otter using fish EPCs calculated by abundance weighting by size class (using both site-wide abundance and abundance for RM ≥ 10).
  - **RM ≥ 10 for all fish –** Using an exposure area limited to RM ≥ 10 for all fish resulted in decreased LOAEL HQs (maximum of 4.7 HQ units). The HQs for PCB TEQ - mammal that were ≥ 1.0 were adjusted to < 1.0 for mink and river otter using fish EPCs calculated by limiting the exposure area to RM ≥ 10 for

all fish. A site-wide exposure area was assumed for all fish except for SFF in the RM ≥ 10 exposure area scenario.

- **Site use factor –** Assuming that mink and river otter only use the LPRSA seasonally (i.e., SUF = 0.5) resulted in reduction in HQs by half (maximum of 5.0 HQ unit decrease). The HQs for PCB TEQ - mammal that were ≥ 1.0 were adjusted to < 1.0 for mink and river otter when these species were assumed to use the LPRSA seasonally. Neither river otter nor mink have been observed in the LPRSA. This BERA has used a conservative assumption that both mink and river otter use the LPRSA year-round.
- **Treatment of non-detects for EPCs –** The treatment of non-detects and adjustments resulted in relatively small changes to LOAEL HQs (maximum of 0.6 HQ unit decrease).

Table 9-17. Summary of uncertainties evaluated for mammal species

Species	General Uncertainty Evaluated	Specific Evaluation Conducted	Rationale	Difference in HQ[a]
Mink	average body weight	Include the minimum and maximum male and female body weights reported in USEPA (1993) .	Evaluate effect on risk estimates based on minimum and maximum body weights.	≤ 0.3 (±)
River otter				≤ 0.4 (±)
Mink/river otter	SIR of 2% based on best professional judgement	Include SIRs of 1 and 4%.	Evaluate effect on risk estimates based on reasonable range to bracket the original estimate.	≤ 0.7 (±)
Mink	FIR of 14% of the body weight	Include range of FIRs provided by USEPA (1993) for mink (12 and 22%).	Evaluate effect on risk estimates based on the minimum and maximum FIRs.	≤ 4.6 (±)
River otter	FIR of 16% of the body weight	Include FIRs of 14 and 18% of the body weight.	Evaluate effect on risk estimates based on reasonable range to bracket the original FIR.	≤ 1.0 (±)
Mink/river otter	selected percentage of > 30-cm fish (i.e., 3% for mink and 5% for river otter)	Include 10% > 30-cm fish in diet.	Evaluate effect on risk estimates based on a diet with a high percentage of fish > 30 cm.	≤ 0.5 (+)
		Include 20% > 30-cm fish in diet.	Evaluate effect on risk estimates based a diet with a very high percentage of fish > 30 cm.	≤ 1.1 (+)
Mink/river otter	selected portions of fish prey size classes (i.e., ≤ 30 cm and > 30 cm)	Group all fish prey as a single size class rather than dividing by size class.	Evaluate difference in risk estimates based on grouping all fish prey together to derive EPCs.	≤ 5.6 (+)

Table 9-17. Summary of uncertainties evaluated for mammal species

Species	General Uncertainty Evaluated	Specific Evaluation Conducted	Rationale	Difference in HQ[a]
Mink/river otter	fish EPCs based on available data	Include diet based on site-wide fish abundance by fish prey size class.[b]	Evaluate difference in risk estimates based on weighting fish EPCs by site-wide abundance of fish prey size class.	≤ 4.4 (-)
		Include diet based on fish abundance from RM ≥ 10 by fish prey size class.[b]	Evaluate difference in risk estimates based on weighting fish EPCs by fish prey size class abundance from RM ≥ 10.	≤ 4.7 (-)
Mink/river otter	selected exposure areas for fish prey	Include exposure area limited to RM ≥ 10 for all fish prey.	Evaluate difference in risk estimates based on limiting exposure area for all fish prey to RM ≥ 10.	≤ 4.7 (-)
Mink/river otter	assumption of 100% site use	Include SUF of 0.5 (rather than 1).	Evaluate the effect on risk estimates when assuming use of the LPRSA only seasonally.	≤ 5 (-)
Mink/river otter	treatment of non-detects (DL = 0 for PCB congeners and Kaplan-Meier method for TEQ)	Include DL = 0 (for TEQ - mammal only), one-half the DL, or the full DL for non-detects.	Evaluate the effect on risk estimates based on treatment of non-detects.	≤ 0.06 (-)

[a] Differences in LOAEL HQs were calculated from the PCB and TEQ - mammal data presented in Table 9-16 and are based on mink diet Scenario 4 (16.5% blue crab, 80.5% ≤ 30-cm fish, and 3% fish > 30 cm) and otter diet Scenario 2 (15% blue crab, 80% ≤ 30-cm fish, and 5% fish > 30 cm) for the site-wide exposure area, unless otherwise noted. Direction of the HQ change is provided in parentheses.

[b] See Appendix H for details on abundance calculations.

DL – detection limit
EPC – exposure point concentration
FIR – food ingestion rate
HQ – hazard quotient
LPRSA – Lower Passaic River Study Area
LOAEL – lowest-observed-adverse-effect level
PCB – polychlorinated biphenyl
RM – river mile
SIR – sediment ingestion rate
SUF – site use factor
TEQ – toxic equivalent
TRV – toxicity reference value
USEPA – US Environmental Protection Agency

Sensitivity Analysis

A sensitivity analysis of risk estimates was conducted using probabilistic methods for river otter and mink for total PCBs, PCB TEQ - mammal, PCDD/PCDF TEQ - mammal, and total TEQ - mammal, because the LOAEL HQs for these COPEC-mammal species pairs were ≥ 1.0 for one or both of these species. The use of probabilistic methods to conduct this sensitivity analysis allowed for the incorporation of variability and uncertainty associated with input parameter values into the resulting risk estimates. These risk estimates, in turn, allowed for a better understanding of the potential range of risk estimates associated with these COPECs, and a better understanding of which parameters have the greatest impact on the resulting HQs. This sensitivity analysis evaluated a variety of assumptions:

- **Toxicity reference values** – a range of TRVs was used.
- **Exposure area** – both site-wide and at and upstream of RM 10 exposure areas were evaluated.
- **Dietary scenarios** – two diet scenarios were evaluated. The first scenario, henceforth referred to as the primary diet scenario, is a diet based on size classes wherein all fish of a given size class were grouped together. The second scenario, henceforth referred to as the abundance-weighted diet scenario, is a diet wherein the fish species with in a given size class are weighted by abundance.

Thus, for each chemical-species combination, a total of eight analyses were conducted to cover the range of variables evaluated. The details of this analysis are presented in Appendix H.

Three of the key exposure parameters or parameter groups that were used to calculate the point estimate HQs presented in Section 9 (Tables 9-14 and 9-15) were evaluated in this sensitivity analysis. Exposure distributions were defined for each of these parameters/parameter groups. It should also be noted that as part of the sensitivity evaluation, there were some differences in how these key parameters were considered, which are described as follows:

- **Dietary fractions** – The diets of river otter and mink (both the components of the diets and the DFs themselves) were adjusted to more accurately reflect the opportunistic feeding habits of mink and river otter. Diets were assumed to be composed of prey items from five broad categories: small fish (≤ 30 cm), large fish (> 30 cm; consumed as carrion), invertebrates, birds, and mammals (only mink was assumed to consume birds and mammals). For the primary diet scenario, fish data based on size classes (wherein all fish of a given size class were grouped together) were used to develop dietary distribution ranges. For the abundance-weighted diet scenario, subcategories were developed, for both the small and large fish categories, using the available site-specific fish abundance data to develop these dietary distribution ranges (i.e., the available fish community data from the LPRSA were used to calculate abundance for the fish species included in mink and river otter diets; see Appendix H for details).
- **Food ingestion rate** – To evaluate the impact of the FIR on risk estimates, this sensitivity evaluation was conducted using the exposure model presented by Moore et al. (1999), in which the FIR was calculated using the assimilation efficiency (AE) of the various prey items, the gross energy (GE) of the prey items, and the metabolic rates of mink and river otter. Thus, the FIR was represented as a distribution of values in the sensitivity analysis.

- **Prey concentrations** – Rather than using a single value (e.g., a UCL) to represent the prey concentration, distributions were developed using the LPRSA data for each prey category for use in the sensitivity analysis. As a health-protective assumption, the low end of the distributions was truncated at the minimum detected concentration. No truncation occurred at the high end of the distribution to acknowledge that individuals with higher concentrations than those detected in the available samples could be present in the LPRSA.

The Moore et al. (1999) exposure model and the various distributions needed to parameterize the model (e.g., distributions for the parameters needed to calculate the consumption rate, the DF distributions, and the concentration distributions) are presented in detail in Appendix H.

The sensitivity analysis for mink and river otter was conducted using a Monte Carlo simulation (see Appendix H for details). A value from the distribution for each input parameter was selected at random in each of the 5,000 model iterations and used to calculate the dietary dose and HQs for each chemical. Using the output from this simulation, the approximate percentage of the population for which the LOAEL HQ was less than the threshold of 1.0 was determined, as summarized in Table 3-1. HQs for river otter are greater than those for mink, largely because a portion of the mink diet is comprised of birds and mammals. Birds and mammals are assumed to have prey concentrations equal to 0 (see Appendix H), which means that a greater percentage of the population for mink has an LOAEL HQ below the threshold of 1.0. LOAEL HQs for PCB TEQ - mammal for both mink and river otter were generally < 1 for more than 80% of the population (Table 9-18).

Table 9-18. HQs for mink and river otter based on the sensitivity analysis

Chemical	Approximate Percentage of the Population for which the LOAEL HQ is Below the Threshold of 1.0 using a Range of TRVs[a]							
	Mink				River Otter			
	Primary Diet Scenario		Abundance-weighted Diet Scenario		Primary Diet Scenario		Abundance-weighted Diet Scenario	
	RM ≥ 10	Site Wide	RM ≥ 10	Site Wide	RM ≥ 10	Site Wide	RM ≥ 10	Site Wide
TRV								
Total PCBs	74%	79%	93%	92%	24%	34%	22%	16%
PCDF/PCDD TEQ - mammal	70%	77%	88%	88%	42%	47%	61%	53%
PCB TEQ - mammal	100%	100%	100%	100%	98%	100%	100%	100%
Total TEQ - mammal	63%	68%	80%	80%	34%	37%	39%	35%

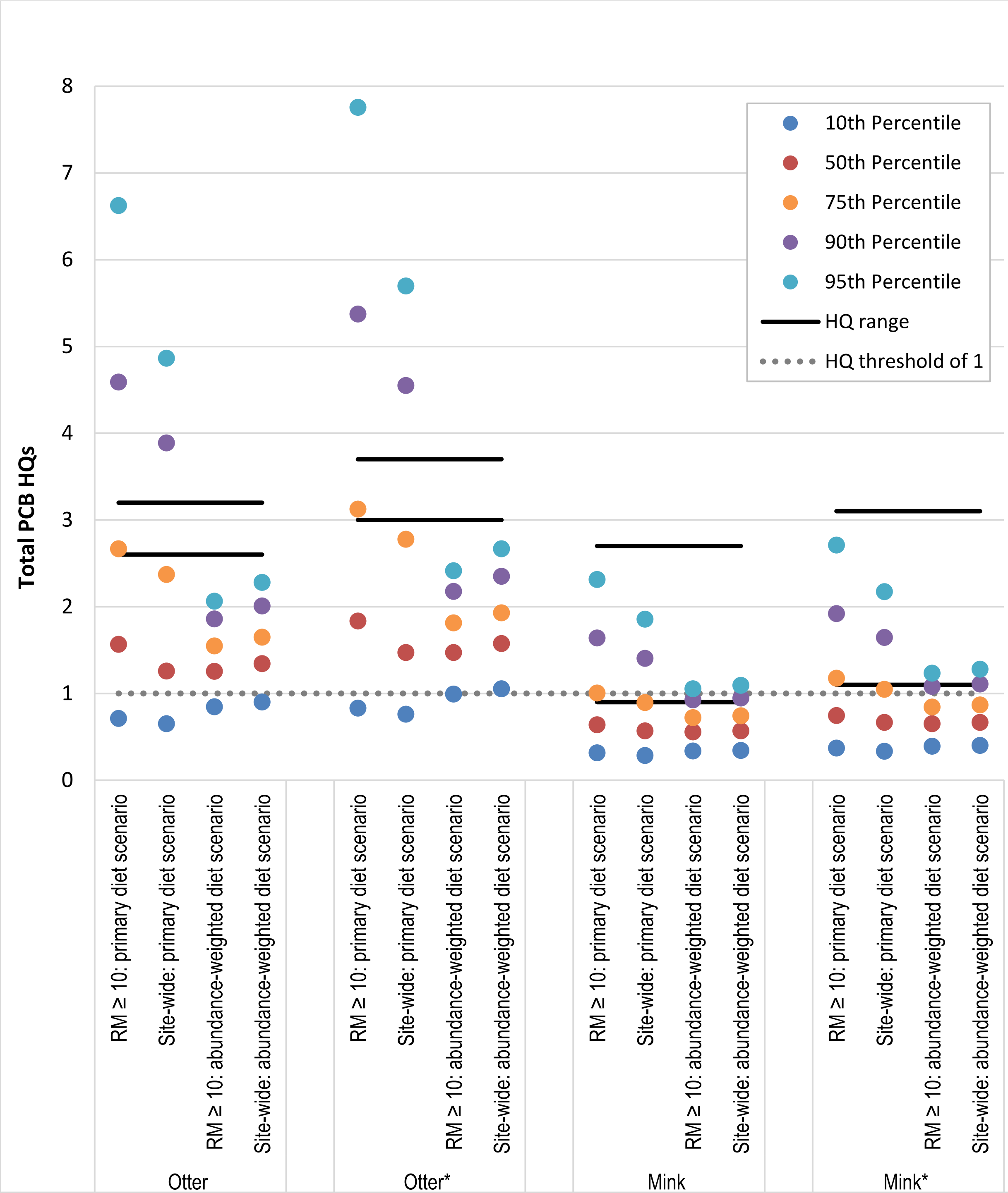
Chemical	Approximate Percentage of the Population for which the LOAEL HQ is Below the Threshold of 1.0 using a Range of TRVs[a]							
	Mink				River Otter			
	Primary Diet Scenario		Abundance-weighted Diet Scenario		Primary Diet Scenario		Abundance-weighted Diet Scenario	
	RM ≥ 10	Site Wide	RM ≥ 10	Site Wide	RM ≥ 10	Site Wide	RM ≥ 10	Site Wide
TRV[b]								
Total PCBs	67%	72%	86%	84%	17%	24%	10%	6%
PCDF/PCDD TEQ - mammal	8%	7%	4%	4%	3%	1%	0%	0%
PCB TEQ - mammal	91%	94%	98%	99%	52%	62%	84%	81%
Total TEQ - mammal	5%	4%	2%	2%	0%	0%	0%	0%

Shaded cells indicate that the HQ is less than 1.0 for 80% or more of the population.

[a] TRVs were derived from the primary literature based on the process identified in Section 9.1.3.1, or were based on USEPA's revised draft of the LPR restoration project FFS (Louis Berger et al. 2014) or first draft of the LPR restoration project FFS (Malcolm Pirnie 2007b).

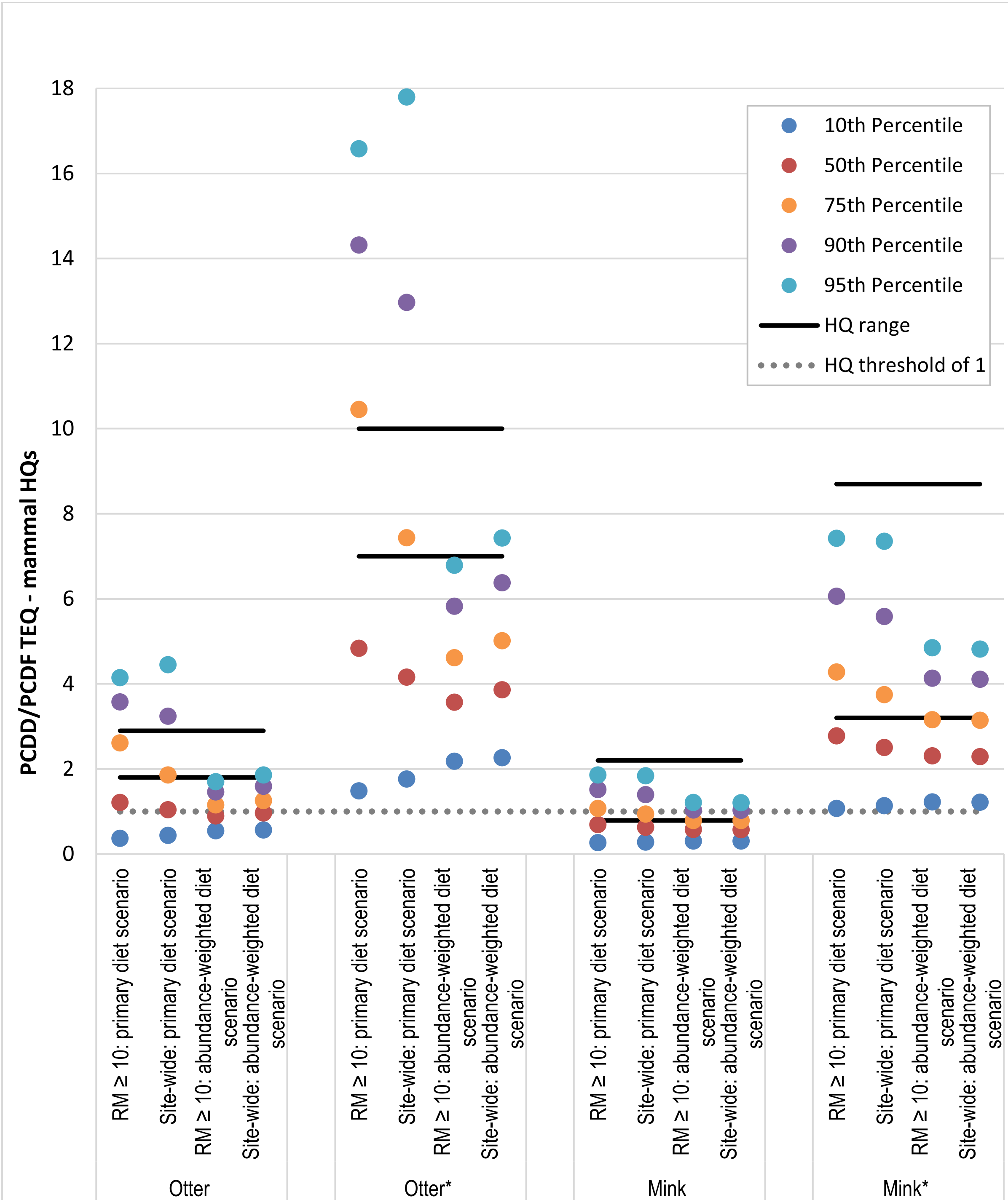
FFS – focused feasibility study
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
LPR – Lower Passaic River
PCB – polychlorinated biphenyl
PCDD – polychlorinated dibenzo-*p*-dioxin
PCDF – polychlorinated dibenzofuran
RM – river mile
TEQ – toxic equivalent
TRV – toxicity reference value
USEPA – US Environmental Protection Agency

In addition to the results shown in Table 9-18, the LOAEL HQ results are presented graphically in Figures 9-4 through 9-7. These figures also show the range of point estimate HQs as horizontal black lines as compared with the HQ distribution from the sensitivity analysis. As in Table 9-18, these figures show that the HQs for river otter are greater than those for mink. In addition, these figures show that the deterministically calculated HQs are conservative (i.e., health protective), since they are generally towards the upper end of the distributions presented in Figures 9-4 through 9-7.



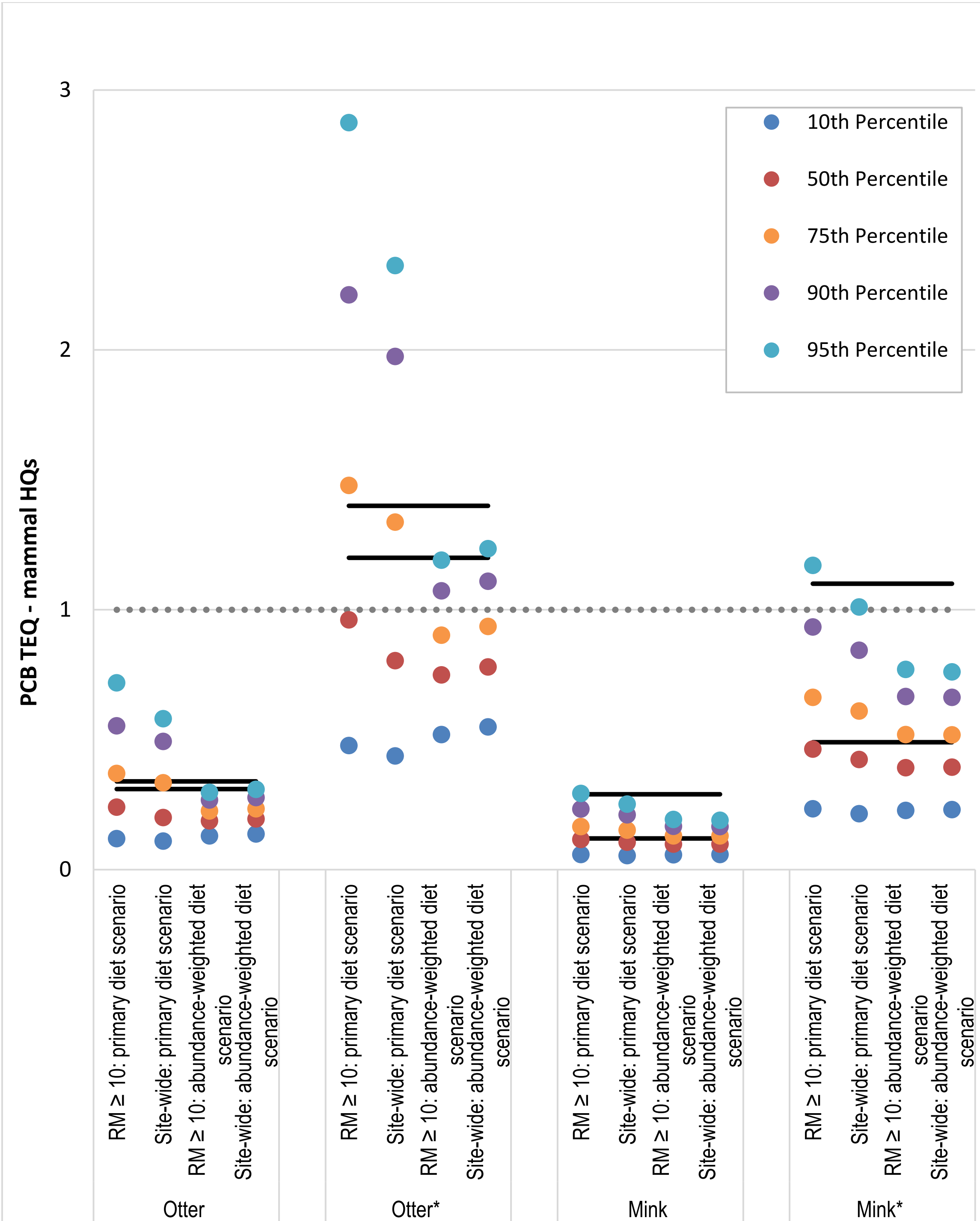
Note: A range of TRVs was used. TRVs were derived based on the process identified in Section 9.1.3.1, or based on USEPA's revised draft of the LPR restoration project FFS (Louis Berger et al. 2014) or first draft of the LPR restoration project FFS (Malcolm Pirnie 2007b).

Figure 9-4. Distribution of LOAEL HQs for total PCBs



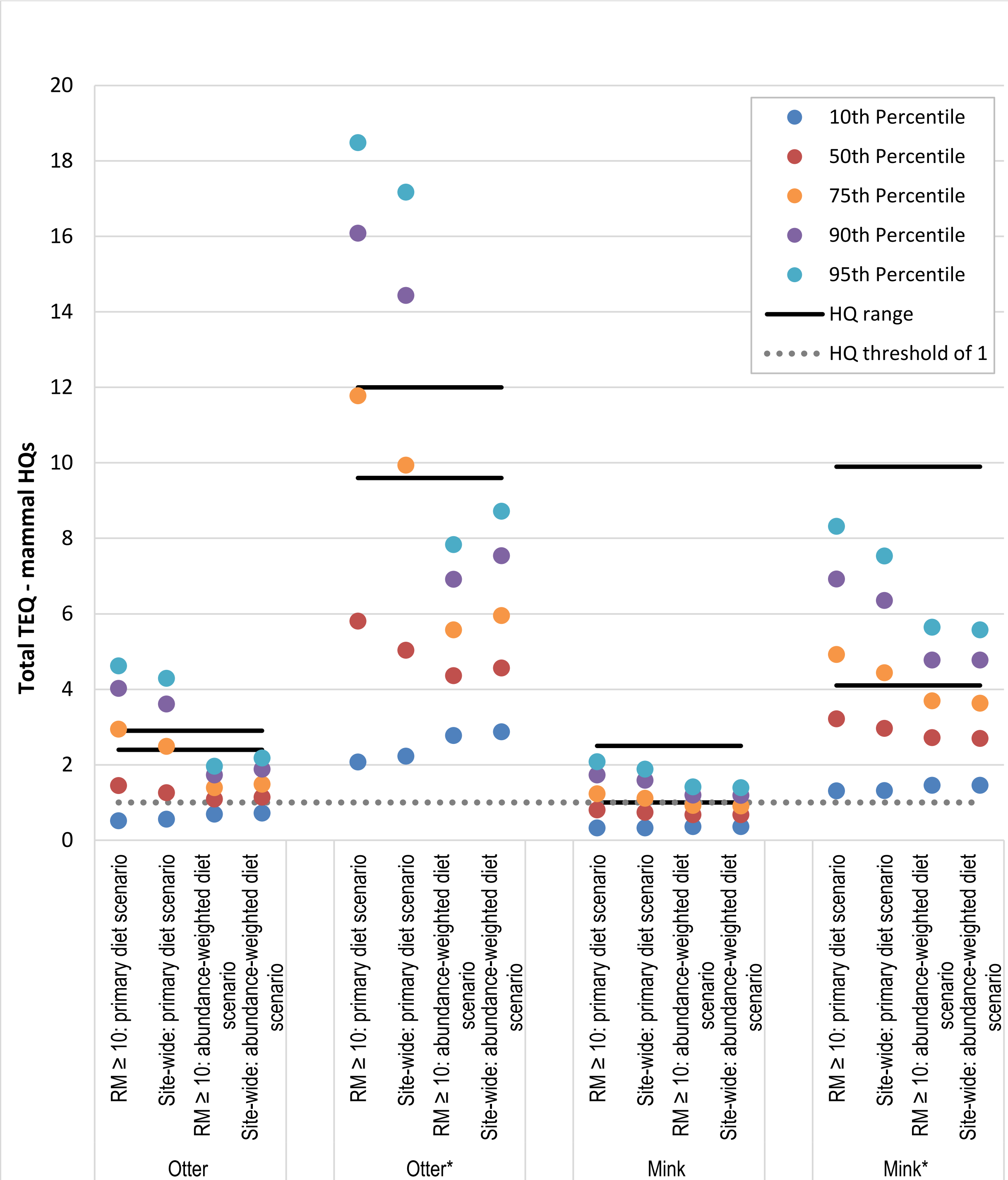
Note: A range of TRVs was used. TRVs were derived based on the process identified in Section 9.1.3.1, or based on USEPA's revised draft of the LPR restoration project FFS (Louis Berger et al. 2014) or first draft of the LPR restoration project FFS (Malcolm Pirnie 2007b).

Figure 9-5. Distribution of LOAEL HQs for PCDD/PCDF TEQ - mammal



Note: A range of TRVs was used. TRVs were derived based on the process identified in Section 9.1.3.1, or based on USEPA's revised draft of the LPR restoration project FFS (Louis Berger et al. 2014) or first draft of the LPR restoration project FFS (Malcolm Pirnie 2007b).

Figure 9-6. Distribution of LOAEL HQs for PCB TEQ - mammal



Note: A range of TRVs was used. TRVs were derived based on the process identified in Section 9.1.3.1, or based on USEPA's revised draft of the LPR restoration project FFS (Louis Berger et al. 2014) or first draft of the LPR restoration project FFS (Malcolm Pirnie 2007b).

Figure 9-7. Distribution of LOAEL HQs for total TEQ - mammal

In addition to using this sensitivity analysis to generate HQs, an evaluation of the sensitivity of the model to the various input parameters was conducted to allow for a better understanding of the key parameters that drive the risk estimates. For this analysis, one exposure area (i.e., site wide) for river otter was selected for further evaluation. Both dietary scenarios were evaluated, meaning that the sensitivity of HQs to parameters used to calculate both the primary diet scenario and the abundance-weighted diet scenario were considered. When evaluating the results of this exercise, it is important to recognize that there were fewer parameters for the primary diet scenario (n = 13) as compared with the abundance-weighted diet scenario (n = 29).

Table 9-19 presents a summary of the results of this evaluation, showing only those parameters with correlation coefficients greater than 0.2 or less than -0.2 (i.e., those that have the greatest impact on the HQs). Parameters for which an increase in the parameter value results in an increase in the dietary dose (e.g., the metabolic rate) have positive correlation coefficients, while parameters for which an increase in the parameter value results in a decrease in the dietary dose (e.g., the DF of mammals for mink) have negative correlation coefficients. The following is a brief discussion of the key parameters shown in Table 9-19:

- **Prey exposure concentrations** – Prey concentration distributions included in Table 9-19 are those that make up a large part of the river otter diet and/or for which the concentrations detected in samples from the LPRSA are quite variable (i.e., there is a wide range of values detected in samples collected from the LPRSA, and thus the concentration selected from the distribution has a large impact on the HQ). The importance of these parameters is the result of the natural variability of concentrations in prey tissue.
- **Dietary fractions** – DFs included in Table 9-19 are those with either a wide range of values and/or for which the associated prey concentration is much higher or much less than the average prey concentration. DFs were based on a combination of literature information and (for the abundance-weighted diet scenario) empirical data for determining fish abundance, and thus these parameters represent both a source of variability (the diet of mink and river otter may vary across different portions of the LPRSA and among seasons) and a source of uncertainty (the literature studies were not site specific and may not accurately represent what mink and river otter would be eating in the LPRSA).
- **Field metabolic rate, gross energy, and absorption efficiency** – These parameters are used to calculate the rate of prey consumption (i.e., like the FIR is used to calculate the point estimate HQs), and thus are parameters to which the model can be highly sensitive. These parameter values were based on the literature (i.e., no site-specific values were available), and thus there is some uncertainty associated with these parameters. However, efforts were made to reduce this uncertainty by using multiple LOEs to determine the field metabolic rate (FMR) for both mink and river otter.

A more detailed presentation of this evaluation of key parameters is presented in Appendix H, including graphs showing the correlation coefficients for the evaluation of sensitivity.

Table 9-19. Evaluation of key parameters impacting the calculated LOAEL HQs for river otter

Parameter Category	Parameter	Correlation Coefficient[a]			
		Total PCBs	PCDD/PCDF TEQ – Mammal	PCB TEQ – Mammal	Total TEQ – Mammal
Primary diet scenario					
Prey exposure concentrations	C_{prey}: crab	0.05	0.09	0.10	0.10
	C_{prey}: fish ≤ 30cm	**0.89**	**0.87**	**0.85**	**0.88**
	C_{prey}: fish > 30cm	0.19	**0.22**	**0.21**	0.20
DFs	DF: crab	-0.04	0.07	-	0.04
	DF: fish ≤ 30cm	-	-0.04	-0.03	-
	DF: fish > 30cm	0.11	0.06	0.08	0.07
Parameters affecting FIR	AE: fish ≤ 30cm	-0.07	-	-0.07	-0.03
	FMR: otter	0.18	0.16	**0.21**	0.19
	GE: crab	-0.04	-0.07	-0.06	-0.07
	GE: fish ≤ 30cm	**-0.22**	-0.16	**-0.24**	-0.20
	GE: fish > 30cm	-0.09	-0.05	-0.07	-0.06
Abundance-weighted diet scenario					
Prey exposure concentrations	C_{prey}: carp > 30cm	0.15	**0.30**	-	**0.28**
	C_{prey}: crab	-	0.18	**0.21**	0.16
	C_{prey}: eel ≤ 30cm	0.20	-	**0.21**	-
	C_{prey}: perch < 30cm	**0.30**	**0.25**	**0.23**	**0.23**
	C_{prey}: SFF	**0.35**	**0.50**	**0.36**	**0.48**
DFs	DF: carp		0.17	-	0.18
	DF: eel ≤ 30cm		-0.19	-	-0.17
	DF: fish > 30cm	**0.22**	**0.22**	0.18	**0.22**
	DF: perch < 30cm	**0.25**	0.19	0.20	0.20
	DF: SFF	-0.12	-	-	-
Parameters affecting FIR	FMR: otter	**0.40**	**0.28**	**0.44**	**0.33**
	GE: crab	-	-	-0.14	-
	GE: fish ≤ 30cm	**-0.47**	**-0.24**	**-0.48**	**-0.27**
	GE: fish > 30cm	-0.13	-	-0.13	-

Bold text indicates parameters with the greatest impact on risk; these are the parameters with correlation coefficients greater than 0.2 or less than -0.2.

[a] For cells that contain a "-", the parameter was not one of the top 10 parameters to which the risk estimates were the most sensitive.

AE – assimilation efficiency
C_{prey} – prey concentration
LOAEL – lowest-observed-adverse-effect level
PCB – polychlorinated biphenyl

DF – dietary fraction
FIR – food ingestion rate
FMR – field metabolic rate
GE – gross energy
HQ – hazard quotient
PCDD – polychlorinated dibenzo-*p*-dioxin
PCDF – polychlorinated dibenzofuran
SFF – small forage fish
TEQ – toxic equivalent

The results from this sensitivity analysis indicate that the deterministically calculated HQs are conservative (i.e., health protective) since they generally fall toward the upper end of the probabilistically calculated distributions, both for the primary diet scenario and the abundance-weighted diet scenario (Figures 9-4 to 9-7). This is particularly true since the conservative assumptions that were used in the sensitivity analysis are especially likely to influence the upper end of the distribution of HQs. Thus, the 90th and 95th percentiles of the HQ distributions likely overestimate risks to mink and river otter, because these percentiles are the result of compounded conservative assumptions. Additionally, the differences between the deterministically calculated HQs and HQs calculated in the sensitivity analysis are generally not large enough to affect the overall risk conclusions for mink and river otter (i.e., whether or not the LOAEL HQs are ≥ 1.0). Overall, this sensitivity evaluation indicates that the calculated HQs for mink and river otter are conservative estimates of the risk associated with total PCBs, PCB TEQ - mammal, PCDD/PCDF TEQ - mammal, and total TEQ - mammal.

Habitat Evaluation

Due to the limited habitat surrounding the LPRSA and lack of direct observations of mink (although mink tracks were observed along the bank near Dundee Dam during the summer avian survey in August 2010), a habitat analysis was conducted to determine if the area surrounding the LPRSA could support a breeding population of mink. This habitat analysis is presented in Appendix I and summarized below.

The first step of the analysis was to combine land cover and land use data, aerial photos, and field observations in a geographic information system (GIS) to evaluate the quantity and quality of potential mink habitat along the LPRSA. Two different assumptions were used: 1) that mink use areas at a distance of 33 m (100 ft) from the shoreline, which includes most of the riparian vegetation where mink are most likely to reside, and 2) that mink use areas at a distance of 100 m (328 ft) from the shoreline, consistent with the mink HSI model (Allen 1986); the second assumption is the more conservative. The second step of the analysis was to use literature data on mink habitat use and population density to estimate the number of mink that might occupy the available habitat. In the final step, land cover GIS data were used to estimate the approximate area needed to support a minimum viable population.

The results from the habitat analysis, as presented in detail in Appendix I, are as follows:

- Potential mink habitat included 49.2 ha within 33 m of the shoreline and 79.7 ha within 100 m of the shoreline (Appendix I). This habitat is generally considered poor and is patchily dispersed throughout the LPRSA.

- There were no areas within the LPRSA with the minimum amount of habitat (12 ha) within the larger maximum home range estimate (3 km) to support a reproducing female mink.
- At least 50 mink are necessary for a viable population (Pertoldi et al. 2013). The analysis conducted to determine the area needed to support a minimum viable population indicated that more a "habitat" buffer of more than 7 mi would be needed around the LPRSA. The contribution of LPRSA habitat to the total amount of habitat in this area is negligible.

Therefore, although it is possible that the available habitat in the LPRSA might support, at most, one reproducing adult female mink, it is more likely to support none. In addition, it is unlikely there is any population risk from exposure to the LPRSA because there is insufficient area to support a viable mink population.

Future Use and Restoration Activities

Neither river otter nor mink have been observed in the LPRSA. The habitat analysis for mink concluded that the available habitat in the LPRSA is not likely to support any reproducing adult female mink. Uncertainty exists as to whether the LPRSA will be restored to support mink and river otter in the future.

9.1.4.3 Comparison to background

Consistent with USEPA guidance (USEPA 2002c), this section presents background concentrations for prey for mammal dietary COPECs with LOAEL HQs ≥ 1.0 (total PCBs, PCB TEQ - mammal, PCDD/PCDF TEQ - mammal, and total TEQ - mammal). Three background datasets were developed for use in this BERA using available data from the following areas: 1) upstream of Dundee Dam, to represent freshwater urban habitat, 2) Jamaica Bay/Lower Harbor, to represent estuarine urban habitat, and 3) Mullica River/Great Bay, to represent estuarine/freshwater rural habitat. These datasets are summarized in Section 4.2, and details on how background values were determined from these datasets are presented in Appendix J. Data were limited to mummichog and other killifish in the Jamaica Bay/Lower Harbor and Mullica River/Great Bay background areas, as no whole-body data were available for LPRSA fish species. Table 9-20 presents the comparison of LPRSA fish tissue concentrations to concentrations in background areas, where data are available, for fish COPECs with LOAEL HQs ≥ 1.0.

This comparison is summarized as follows:

- For total PCBs, the LPRSA whole-body fish tissue EPCs were generally greater than maximum concentrations and UCLs upstream of Dundee Dam. The mummichog UCL from Jamaica Bay/Lower Harbor (1,900 µg/kg) was approximately 3 times greater than the EPC from the LPRSA (600 µg/kg). Similarly, the maximum total PCB concentration in mummichog from Jamaica Bay/Lower Harbor (3,200 µg/kg) was approximately 3 times greater than the

LPRSA mummichog maximum concentration (930 µg/kg). The mean mummichog lipid content was higher in Jamaica Bay/Lower Harbor mummichog (3.1%) than in LPRSA mummichog (2.0%). The lipid-normalized maximum PCB concentration was approximately 2.6 times greater in mummichog from Jamaica Bay/Lower Harbor (94 mg/kg lipid) than in mummichog from the LPRSA (36 mg/kg lipid). Although the greater mean mummichog lipid content in Jamaica Bay/Lower Harbor could indicate better fish condition, there are other factors that may affect lipid content in fish, such as size, age, sex, reproductive status, genetic background, diet, water temperature, and seasonality (Mraz 2012; Iverson et al. 2002).

- For PCB TEQ - mammal, the LPRSA whole-body fish tissue EPCs were generally greater than UCLs and maximum concentrations upstream of Dundee Dam. The PCB TEQ - mammal EPC for LPRSA mummichog was also greater than the UCL and maximum concentration in Mullica River/Great Bay mummichog. However, the Jamaica Bay/Lower Harbor UCL for mummichog (240 ng/kg) was greater than LPRSA UCL for mummichog (8.0 ng/kg). Similarly, the maximum PCB TEQ - mammal for mummichog from Jamaica Bay/Lower Harbor (70 ng/kg) was approximately 6 times greater than for mummichog from the LPRSA (12 ng/kg). The lipid-normalized maximum PCB TEQ - mammal was approximately five times greater in mummichog from Jamaica Bay/Lower Harbor (0.0025 mg/kg lipid) than in mummichog from the LPRSA (0.00048 mg/kg lipid).
- For PCDD/PCDF TEQ - mammal and total TEQ - mammal, the LPRSA whole-body fish tissue EPCs were greater than UCLs and maximum concentrations upstream of Dundee Dam. The PCDD/PCDF TEQ - mammal EPC for LPRSA mummichog was also greater than UCLs and maximum concentrations in Mullica River/Great Bay and Jamaica Bay/Lower Harbor mummichog. Maximum total TEQ - mammal concentrations were higher in mummichog from the LPRSA than in mummichog from Mullica River/Great Bay and Jamaica Bay/Lower Harbor (Table 9-20).

Table 9-20. LPRSA tissue compared to background tissue for mammal dietary COPECs with LOAEL HQs ≥ 1.0

Species	LPRSA				Above Dundee Dam				Jamaica Bay/Lower Harbor				Mullica River/Great Bay			
	N	EPC	Min. Detect	Max. Detect	N	UCL	Min. Detect	Max. Detect	N	UCL	Min. Detect	Max. Detect	N	UCL	Min. Detect	Max. Detect
Total PCB congeners (µg/kg ww)																
American eel	21	2,000	420	5,700	16	1,080	206	1,880	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]
Brown bullhead	6	1,400	260	1,700	6	519	183	614	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]
Common carp	12	5,200	1,500	7,900	10	2,100	755	2,560	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]
Channel catfish	11	1,700	350	2,700	4	na	948	2,130	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]
Mummichog/killifish[a]	18	600	240	930	1	na	219	219	7	1,900	55	3,200	na[c]	na[c]	na[c]	na[c]
Northern pike	1	2,000	2,000	2,000	1	na	1,880	1,880	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]
Other forage fish	10	550	170	870	2	na	107	853	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]
Smallmouth bass	3	1,400	630	1,400	3	na	1,000	1,310	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]
White perch	22	2,500	290	5,100	8	834	408	1,130	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]
White sucker	5	2,900	540	2,900	5	na	327	872	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]
PCB TEQ - mammal (ng/kg ww)																
American eel	21	11	2.8	17	16	11.1	0.867	15.5	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]
Brown bullhead	6	18	6.1	23	6	7.91	3.74	9.27	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]
Common carp	12	58	16	86	10	38.7	7.49	81.1	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]
Channel catfish	11	25	2.7	38	4	na	17.5	45.5	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]
Mummichog/killifish[a]	18	8.0	3.6	12	1	na	4.05	4.05	7	240	0.047	70	10	4.9	3	3.4
Northern pike	1	31	31	31	1	na	35.1	35.1	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]
Other forage fish	10	7.4	2.1	11	2	na	1.74	10.8	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]
Smallmouth bass	3	19	9.4	19	3	na	14.6	18.0	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]
White perch	22	26	2.9	41	8	11.9	6.34	13.7	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]

Table 9-20. LPRSA tissue compared to background tissue for mammal dietary COPECs with LOAEL HQs ≥ 1.0

Species	LPRSA				Above Dundee Dam				Jamaica Bay/Lower Harbor				Mullica River/Great Bay			
	N	EPC	Min. Detect	Max. Detect	N	UCL	Min. Detect	Max. Detect	N	UCL	Min. Detect	Max. Detect	N	UCL	Min. Detect	Max. Detect
White sucker	5	45	11	45	5	na	3.82	14.3	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]
PCDD/PCDF TEQ - mammal (ng/kg)																
American eel	21	24	0.81	48	16	1.44	0.168	2.50	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]
Brown bullhead	6	160	8.5	200	6	2.10	1.03	2.44	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]
Common carp	12	610	8.2	1,400	10	5.43	2.89	6.60	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]
Channel catfish	11	100	22	170	4	na	2.97	8.43	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]
Mummichog/killifish[a]	18	50	11	100	1	na	0.368	0.368	7	17	7	12	12	0.33	0.036	0.48
Northern pike	1	100	100	100	1	na	4.89	4.89	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]
Other forage fish	10	48	3.8	96	2	na	0.138	2.73	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]
Smallmouth bass	3	76	8.6	76	3	na	1.64	1.86	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]
White perch	22	200	19	260	8	2.44	1.38	3.02	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]
White sucker	5	130	4.1	130	5	na	0.599	2.55	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]
Total TEQ - mammal (ng/kg)																
American eel	21	34	5.5	56	16	12.5	0.902	16.9	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]
Brown bullhead	6	180	15	220	6	9.87	4.76	11.7	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]
Common carp	12	680	24	1,500	10	43.6	11.2	85.2	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]
Channel catfish	11	130	25	210	4	na	20.4	53.8	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]
Mummichog/killifish[a]	18	59	15	110	1	na	4.42	4.42	7	200	27	74	10	5.2	3.3	9.5
Northern pike	1	130	130	130	1	na	40	40.0	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]
Other forage fish	10	56	10	110	2	na	1.88	13.5	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]
Smallmouth bass	3	96	22	96	3	na	16.4	19.8	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]

Table 9-20. LPRSA tissue compared to background tissue for mammal dietary COPECs with LOAEL HQs ≥ 1.0

Species	LPRSA				Above Dundee Dam				Jamaica Bay/Lower Harbor				Mullica River/Great Bay			
	N	EPC	Min. Detect	Max. Detect	N	UCL	Min. Detect	Max. Detect	N	UCL	Min. Detect	Max. Detect	N	UCL	Min. Detect	Max. Detect
White perch	22	230	25	300	8	14.3	7.68	16.7	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]
White sucker	5	170	15	170	5	na	4.42	16.8	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]	na[c]

Note: The maximum detected concentration for background areas exclude outlier concentrations as described in Appendix J.

[a] The mummichog/killifish group consists of mummichog from the LPRSA, Jamaica Bay/Lower Harbor, and Mullica/Great Bay, and banded killifish from above Dundee Dam.

[b] Total PCB congener data were not available; value was based on total PCB Aroclor data. Background value was based on DL; all 10 total PCB Aroclor values in dataset were reported as non-detected concentrations.

[c] Data not available.

DL – detection limit
EPC – exposure point concentration
LPRSA – Lower Passaic River Study Area
na – not available
PCB – polychlorinated biphenyl
PCDD – polychlorinated dibenzo-*p*-dioxin
PCDF – polychlorinated dibenzofuran
TEQ – toxic equivalent
UCL – upper confidence limit on the mean
ww – wet weight

9.1.5 Summary of key uncertainties

The primary uncertainty in the mammal risk assessment is whether mink or river otter are exposed to COPECs based on their use of the LPRSA. The habitat analysis for mink concluded that the available habitat in the LPRSA is not likely to support any reproducing adult female mink. In addition, neither river otter nor mink have been observed in the LPRSA. Another key uncertainty is the range of TRVs used in the risk calculations. Uncertainties associated with TRVs are discussed in Section 9.1.3.

When the variability in exposure parameters and EPCs was evaluated in combination in a probabilistic manner, 75th percentile HQs were similar to the deterministic HQ range for river otter, whereas the 90th and 95th percentile HQs were slightly greater than the deterministic HQ range for river otter. For mink, the deterministic HQ range was greater than the 90th and 95th percentile HQs.

- For other uncertainties in the risk assessment, such as the TEQ methodology and the use of laboratory toxicity data to predict effects, it is possible that effects could be either over- or underestimated. The HQs likely represent an overestimation of risk because of the conservative assumptions used in the risk evaluation, such as the use of the lowest LOAEL among all species or endpoints as the TRV, the use of an upper exposure value (i.e., UCL) to calculate dietary exposure concentrations, and the assumption that a species feeds exclusively from the LPRSA (i.e., SUF = 1).

9.1.6 Summary

Sixteen dietary COPECs were evaluated for mammals. LOAEL HQs were ≥ 1.0 for total PCBs, PCB TEQ - mammal, PCDD/PCDF TEQ - mammal, and total TEQ - mammal. Table 9-21 provides the range in LOAEL HQs for all dietary and exposure area scenarios, using a range of TRVs for the COPECs with LOAEL HQs ≥ 1. The primary uncertainty associated with the mammal risk assessment is the use of the LPRSA by river otter and mink. The habitat analysis for mink concluded that the available habitat in the LPRSA is not likely to support any reproducing adult female mink. Neither river otter nor mink have been observed in the LPRSA.

Table 9-21. Summary of mammal dietary LOAEL HQs

Preliminary COC[b]	Range of LOAEL HQs[a]				Key Uncertainties
	Mink		River Otter		
	HQ Based on TRV-A[c]	HQ Based on TRV-B[d]	HQ Based on TRV-A[c]	HQ Based on TRV-B[d]	
Total PCBs	0.94–**2.7**	**1.1–3.1**	**2.6–3.2**	**3.0–3.7**	• TRV-A and TRV-B based on mink exposure to dietary PCBs; TRVs based on the same literature source with slightly different ingestion rate and body weight assumptions • HQ based on range of dietary scenarios and two exposure areas; HQs slightly lower based on site-wide exposure area (vs. > RM 10); mink HQs lower based on consumption of aquatic prey and assuming zero exposure from terrestrial prey
PCB TEQ - mammal	0.12–0.29	0.49–**1.1**	0.31–0.34	**1.2–1.4**	• TRV-A based on mink fed laboratory-prepared diet; TRV-B based on mink fed field-collected carp • HQ based on range of dietary scenarios and two exposure areas; HQs slightly lower based on site-wide exposure area (vs. > RM10); mink HQs lower based on consumption of aquatic prey and assuming zero exposure from terrestrial prey
PCDD/ PCDF TEQ - mammal	0.79–**2.2**	**3.2–8.7**	**1.8–2.6**	**7.0–10**	
Total TEQ - mammal	**1.0–2.5**	**4.1–9.9**	**2.4–2.9**	**9.6–12**	

Bold identifies HQs ≥ 1.0.

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

[b] Only COPECs with HQs ≥ 1.0 based on LOAEL TRVs are included in the table.

[c] TRVs were derived from the primary literature based on the process identified in Section 9.1.3.1.

[d] TRVs were derived based on USEPA's revised draft of the LPR restoration project FFS (Louis Berger et al. 2014) or first draft of the LPR restoration project FFS (Malcolm Pirnie 2007b).

BERA – baseline ecological risk assessment
COC – chemical of concern
COPEC – chemical of potential ecological concern
FFS – focused feasibility study
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
LPR – Lower Passaic River
LPRSA – Lower Passaic River study Area
NJDEP – New Jersey Department of Environmental Protection
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
PCDD – polychlorinated dibenzo-*p*-dioxins
PCDF – polychlorinated dibenzofurans
RM – river mile
TEQ – toxic equivalent
TRV – toxicity reference value
USEPA – US Environmental Protection Agency

9.2 Summary of Preliminary COCs for Mammals

The potential for unacceptable risk from COPECs to aquatic mammals in the LPRSA was evaluated based on the CSM presented in Section 3. Specifically, the risk assessment for mammals evaluated Assessment Endpoint No. 7:

- Protection and maintenance (i.e., survival, growth, and reproduction) of aquatic mammal populations

The potential for risk to mammals was characterized using LPRSA tissue, sediment, and water chemistry to estimate dietary doses of COPECs to two mammal species (i.e., river otter and mink). Dietary doses were compared to a range of TRVs to derive risk estimates (HQs) in the risk characterization.

COPECs with HQs ≥ 1.0 based on LOAEL TRVs were identified as preliminary COCs. For mink and river otter, four preliminary COCs (i.e., PCDD/PCDF TEQ - mammal, total TEQ - mammal, total PCBs, and PCB TEQ - mammal) were identified with HQs ≥ 1.0 (Table 9-22).

Table 9-22. Summary of preliminary COCs

Preliminary COC[b] and Exposure Area	Range of LOAEL HQ[a]			
	Mink		River Otter	
	HQ Based on TRV-A[c]	HQ Based on TRV-B[d]	HQ Based on TRV-A[c]	HQ Based on TRV-B[d]
Total PCBs				
RM ≥ 10	**1.1–2.7**	**1.3–3.1**	**3.1–3.2**	**3.7**
site wide	0.94–**2.2**	**1.1–2.6**	**2.6**	**3.0–3.1**
PCB TEQ - mammal				
RM ≥ 10	0.13–0.29	0.51–**1.1**	0.32–0.34	**1.3–1.4**
site wide	0.12–0.27	0.49–**1.1**	0.31–0.33	**1.2–1.3**
PCDD/PCDF TEQ - mammal				
RM ≥ 10	**1.1–2.2**	**4.2–8.7**	**2.5–2.6**	**10**
site wide	0.79–**1.6**	**3.2–6.2**	**1.8–1.9**	**7.0–7.4**
Total TEQ - mammal				
RM ≥ 10	**1.2–2.5**	**4.7–9.9**	**2.8–2.9**	**11–12**
site wide	**1.0–2.1**	**4.1–8.4**	**2.4–2.5**	**9.6–10**

Bold identifies HQs ≥ 1.0.

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

b Only COPECs with HQs ≥ 1.0 based on LOAEL TRVs are included in the table.

[c] TRVs were derived from the primary literature based on the process identified in Section 9.1.3.1.

[d] TRVs were derived based on USEPA's revised draft of the LPR restoration project FFS (Louis Berger et al. 2014) or first draft of the LPR restoration project FFS (Malcolm Pirnie 2007b).

BERA – baseline ecological risk assessment
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LOAEL – lowest-observed-adverse-effect level
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NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
PCDD – polychlorinated dibenzo-*p*-dioxins
PCDF – polychlorinated dibenzofurans
RM – river mile
TEQ – toxic equivalent
TRV – toxicity reference value
USEPA – US Environmental Protection Agency

The results of this mammal risk assessment will be used in the FS as a tool for risk managers to make potential remedial decisions for the LPRSA. Determining the potential for unacceptable ecological risk at the population level provides information pertaining to decisions to be made in the FS or other programmatic environmental management changes. The TRVs used to evaluate risk to mammals in this BERA are organism-level effects, including those that affect particular attributes of organisms within a population, such as survival, growth, and reproduction. Survival, growth, and reproduction of individual organisms may, in turn, affect populations of those organisms depending upon the magnitude and severity of the effect. However, population-level effects that influence the entire population—such as size or density of population, population growth, or population survival—are more direct measures of influences on the population as a whole. USEPA guidance states that assessment endpoints should be associated with sustaining the ecological structure and function of populations and communities rather than individual organisms, unless individuals warrant additional protection in specific cases (USEPA 1999). Since BERAs evaluate ***populations***, not individuals, as assessment endpoints, other factors, including the magnitude and severity of the effect, should be assessed to determine if a risk driver (defined and identified in Section 13) should be used in developing PRGs and RALs.

10 Zooplankton Assessment

The risk assessment for zooplankton in the LPRSA evaluated the following assessment endpoint:

- **Assessment Endpoint No. 1 –** Maintenance of the zooplankton community that serves as a food base for juvenile fish

The evaluation of risks to the zooplankton community in the LPRSA was based on a comparison of LPRSA surface water concentrations to TRVs intended to be protective of a variety of aquatic organisms. The assessment of zooplankton exposed to surface water was the same as that of fish presented in Section 7.3. EPCs were based on all mean and UCL COPEC concentrations in LPRSA surface water from two areas: between RM 0 and RM 13 for comparison to estuarine thresholds, and between RM 4 and RM 17.4 for comparison to freshwater thresholds. TRVs for surface water were selected based on available invertebrate and fish toxicity data (Appendix D). SSDs were used to derive TRVs when sufficient data were available (i.e., toxicity data were available for a minimum of five species). Details on this assessment are provided in Section 7.3. As for the surface water assessment for fish, Appendix G compiles EPCs, TRVs, and calculated HQs for the surface water COPECs applicable to zooplankton into a single table (Table G5).

A total of 25[129] COPECs were evaluated for this receptor group (Table 7-27). COPECs with HQs ≥ 1.0 were identified as preliminary COCs. Two surface water COPECs had a range of effect-level HQs, some of which were ≥ 1.0 and were identified as preliminary COCs (Table 7-30): copper (HQs ranged from 0.14 to 2.7) and estuarine cyanide (HQs ranged from 1.6 to 5.3).

[129] TEQ COPECs (i.e., PCDD/PCDF TEQ, PCB TEQ, and total TEQ) were evaluated for fish, but not for zooplankton.

11 Amphibian and Reptile Assessment

The risk assessment for amphibians and reptiles evaluated the following assessment endpoint:

- **Assessment Endpoint No. 8 –** Protection and maintenance (i.e., survival, growth, and reproduction) of healthy amphibian and reptile populations

The evaluation of risks to amphibians and reptiles in the LPRSA was based on a comparison of LPRSA surface water concentrations to amphibian-specific TRVs. Limited amphibian- and reptile-specific water toxicity data are available, so the evaluation of risks to amphibians and reptiles is limited and uncertain. Because of the uncertainty associated with the evaluation of reptiles and amphibians, the evaluation is presented in Appendix N and summarized in this section. Appendix G (Table G23) includes EPCs, TRVs, and HQs for amphibian/reptile surface water COPECs.

The calculated HQs for all seven amphibian and reptile COPECs evaluated (chromium, copper, lead, mercury, nickel, silver, and zinc) were < 1.0. No preliminary COCs were identified for amphibians and reptiles because all COPECs had effect-level HQs < 1.0. Unacceptable population-level risks to amphibians and reptiles from exposure to surface water are not expected. Due to a lack of TRVs for herptiles, the potential risk to and impact on herptile populations is unknown.

12 Aquatic Plant Assessment

This risk assessment for aquatic plants in the LPRSA evaluated the following assessment endpoint:

- **Assessment Endpoint No. 9 –** Maintenance of healthy aquatic plant populations as a food resource and habitat for fish and wildlife populations

The evaluation of risks to aquatic plants in the LPRSA was based on a comparison of LPRSA surface water and sediment data to media-specific effects thresholds expected to be protective of aquatic plants. The paucity and questionable applicability of both exposure and effects data, especially for the sediment evaluation, reduce the level of certainty for the quantitative estimates of risk to the aquatic plant community. Because plants are important components of the ecosystem, an assessment was conducted to provide a summary of the information available to evaluate potential impacts on aquatic plants from surface water- and sediment-associated chemicals. However, risk estimates from this assessment are highly uncertain and should be considered only qualitatively for the purposes of risk management conclusions and decisions. The aquatic plant evaluation is presented in Appendix O, and the results are summarized below. Appendix G (Table G24) includes EPCs, TRVs, and HQs for aquatic plant surface water and sediment COPECs.

COPECs with effect-level HQs ≥ 1.0 based on either the surface water or sediment LOE were identified as preliminary COCs for aquatic plants. The following seven preliminary COCs were identified for aquatic plants based on the sediment LOE:

- Chromium (HQ = 160)
- Copper (HQ = 2.4)
- Lead (HQ = 2.3)
- Mercury (HQ = 9.7)
- Selenium (HQ = 1.8)
- Vanadium (HQ = 14)
- Zinc (HQ = 3.1)

The following four preliminary COCs were identified for aquatic plants based on the surface water LOE:

- Copper (HQs = 0.64–1.8)
- Zinc (estuarine; HQ = 21)
- TBT (HQs = 1.1–50)
- Cyanide (estuarine; HQ = 2.0)

13 Summary of Preliminary COCs and Risk Drivers

This final BERA for the 17.4 mi of the LPRSA was conducted and prepared in accordance with Section IX.37.d of the May 2007 Administrative Settlement Agreement and Order on Consent (AOC) (USEPA 2007a). This final BERA has been amended to address comments, responses, and directives received from USEPA on January 2, 2019 (CDM 2019), March 5, 2019 (USEPA 2019), and via additional communications between the CPG and USEPA from January through June 2019.

This BERA evaluated nine assessment endpoints that addressed the protection and maintenance of communities or healthy populations of the ecological species or groups that were evaluated (Table 13-1). These assessments endpoints were evaluated within a site-specific framework that represented site-related chemicals, a developed understanding of the site conceptual model, the implications of an estuarine system for ecological impact, and that incorporated the urban characteristics of the LPR. Developing a site-specific BERA is particularly important in an urban setting such as the LPRSA, which is a large, complex site within a highly developed region. Adjacent land use is predominantly industrial in the lower river and becomes more commercial, residential, and recreational in the upper reaches of the study area. Like many other urban systems, the LPRSA has been subjected to a broad range of contaminant loadings from multiple sources, including untreated industrial and municipal wastewater, CSOs/SWOs, and direct runoff.

The potential for unacceptable risk was assessed using empirical and modeled data collected from a variety of chemical and biological sampling events and surveys conducted as part of the LPRSA RI. A step-by-step process included an initial screening-level evaluation (presented in the SLERA; Appendix A), which identified media-specific COPECs, followed by a more detailed evaluation of potential site-specific exposures and effects to derive risk estimates (expressed as HQs) to identify the potential for unacceptable ecological risk under baseline conditions. COPECs with effect-level HQs ≥ 1.0[130] were identified as preliminary COCs. The ERA of benthic invertebrates followed an approach similar to that of the surface water and tissue LOEs; however, the assessment of risk to community structure and function was based on an SQT analysis of sediment chemical concentration, sediment toxicity test, and benthic invertebrate community data. Preliminary COCs were not derived using the SQT analysis.

The preliminary COCs for each assessment endpoint are presented in Table 13-1. Effect-level HQs for all preliminary COCs are presented in Table 13-2.

[130] Preliminary COCs were identified as those COPECs with HQs ≥ 1.0 based on any LOE and effect-level concentration (i.e., HQ ≥ 1.0 based on a LOAEL for tissue and diet LOEs, HQ ≥ 1.0 based on acute or chronic surface water TRVs; HQ ≥ 1.0 based on plant-specific sediment TRVs).

Table 13-1. Summary of ecological COPECs and preliminary COCs

Selected Receptor Group and Species Evaluated	LOE	COPECs[a]	
Protection and maintenance (i.e., survival, growth, and reproduction) of the benthic invertebrate community, both as an environmental resource in itself and as one that serves as a forage base for fish and wildlife populations			
Benthic invertebrate community	SQT (benthic community metrics; toxicity test data; surface sediment chemistry)	not identified using the SQT analysis	No preliminary COCs were ident follows: • No, low, or likely low impacts (conditions were observed at ~ ~63% of the SQT locations. • Likely or high impacts were ob • At ~32% of the SQT locations, to confounding factors).
Benthic invertebrates (including benthic invertebrate community, macroinvertebrates, and mollusks)	surface water	cadmium, chromium, copper, lead, mercury, selenium silver, zinc, TBT (estuarine), anthracene, benzo(a)anthracene, benzo(a)pyrene, fluoranthene, pyrene, BEHP, BBP, total PCBs, 2,3,7,8-TCDD, 4,4′-DDE, 4,4′-DDT, dieldrin (estuarine), hexachlorobenzene, total chlordane, total DDx, cyanide	2,3,7,8-TCDD, copper, cyanide
Protection and maintenance (i.e., survival, growth, and reproduction) of healthy populations of benthic invertebrates (worms, blue crab and crayfish, and bivalve mussels) that serve as a forage base for fish and wildlife populations and as a base for sports fisheries[c]			
Benthic invertebrates (worms, blue crab, and caged mussels)	tissue	arsenic, cadmium, chromium, cobalt, copper, lead, mercury, methylmercury, nickel, selenium, silver, vanadium, zinc, total LPAHs, total HPAHs, total PCBs, PCB TEQ-fish, 2,3,7,8-TCDD, PCDD/PCDF TEQ - fish, total TEQ - fish, dieldrin, heptachlor expoxide, total DDx	arsenic,[c] chromium,[c] copper,[c] lea 2,3,7,8-TCDD, PCDD/PCDF TE
Protection and maintenance (i.e., survival, growth, and reproduction) of omnivorous, invertivorous, and piscivorous fish populations that serve as a forage base for fish and wildlife populations and as a base for sports fisheries			
Fish populations (mummichog/other forage fish, common carp, white perch, channel catfish, white catfish, brown bullhead, American eel, largemouth bass, smallmouth bass, and northern pike)	tissue	arsenic, cadmium, chromium, copper, lead, methylmercury, selenium, silver, zinc, total HPAHs, total LPAHs, total PCBs, PCB TEQ - fish, 2,3,7,8-TCDD, PCDD/PCDF TEQ - fish, total TEQ - fish, dieldrin, endosulfan I, total DDx	copper,[c] methylmercury/mercury fish, total PCBs, dieldrin, total DI
	diet	cadmium, chromium, cobalt, copper, mercury methyl mercury, nickel, selenium, vanadium, zinc, TBT, total PAHs, benzo(a)pyrene, total PCBs, PCT TEQ - fish, PCDD/PCDF TEQ-fish, total TEQ - fish, total DDx	cadmium, mercury, PCB TEQ - f
	surface water	cadmium, chromium, copper, lead, mercury, selenium, silver, zinc, TBT (estuarine), anthracene, benzo(a)anthracene, benzo(a)pyrene, fluoranthene, pyrene, BEHP, BBP, total PCBs, PCB TEQ - fish, 2,3,7,8-TCDD, PCDD/PCDF TEQ - fish, total TEQ - fish, 4,4′-DDE, 4,4′-DDT, dieldrin (estuarine), hexachlorobenzene, total chlordane, total DDx, cyanide	copper and cyanide
	egg tissue (mummichog)	mercury, methylmercury, total PCBs, PCDD/PCDF TEQ - fish, total TEQ - fish	mercury, total PCBs
	mummichog egg count	none identified based on qualitative LOE	none identified based on qualitat
	health assessment	none identified based on qualitative LOE	none identified based on qualitat
Protection and maintenance (i.e., survival, growth, and reproduction) of herbivorous, omnivorous, sediment-probing, and piscivorous bird populations			
Bird populations (spotted sandpiper, belted kingfisher, and great blue heron)	diet	cadmium, chromium, copper, lead, methylmercury, nickel, selenium, vanadium, zinc, total LPAHs, total HPAHs, total PCBs, PCB TEQ - bird, PCDD/PCDF TEQ - bird, total TEQ - bird, total DDx	copper, lead, methylmercury, tota total DDx
	egg tissue	methylmercury/mercury, total PCBs, PCB TEQ - bird, PCDD/PCDF TEQ - bird, total TEQ - bird, total DDx, dieldrin	total PCBs, PCDD/PCDF TEQ -
Protection and maintenance (i.e., survival, growth, and reproduction) of aquatic mammal populations			
Mammal populations (river otter and mink)	diet	arsenic, cadmium, copper, lead, methylmercury/mercury, nickel, selenium, vanadium, and zinc, total HPAHs, total PCBs, PCB TEQ - mammal, PCDD/PCDF TEQ - mammal, total TEQ - mammal, dieldrin	total PCBs, PCB TEQ - mammal
Maintenance of the zooplankton community that serves as a food base for juvenile fish			
Zooplankton community	surface water	cadmium, chromium, copper, lead, mercury, selenium, silver, zinc, TBT (estuarine), anthracene, benzo(a)anthracene, benzo(a)pyrene, fluoranthene, pyrene, BEHP, BBP, total PCBs, 2,3,7,8-TCDD, 4,4′-DDE, 4,4′-DDT, dieldrin (estuarine), hexachlorobenzene, total chlordane, total DDx, cyanide	copper and cyanide
Protection and maintenance (i.e., survival, growth, and reproduction) of healthy amphibian and reptile populations			

Table 13-1. Summary of ecological COPECs and preliminary COCs

Selected Receptor Group and Species Evaluated	LOE	COPECs[a]	
Amphibians and reptile populations (multiple species represented)	surface water	chromium, copper, lead, mercury, nickel, silver, zinc	none identified
Maintenance of healthy aquatic plant populations as a food resource and habitat for fish and wildlife populations			
Aquatic plant populations (multiple species represented)	sediment	antimony, arsenic, cadmium, chromium, cobalt, copper, lead, mercury, nickel, selenium, vanadium, zinc, and acenaphthene	chromium, copper, lead, mercury
	surface water	cadmium, chromium, copper, lead, mercury (estuarine), zinc, TBT, total PCBs (estuarine), 2,3,7,8-TCDD, 4,4'-DDE, cyanide (estuarine)	copper, zinc, TBT, cyanide

[a] COPECs are those COIs for which the maximum concentration exceeded its TSV in the SLERA. If a TSV was exceeded based on any species in a receptor group, it was retained as a COPEC for all species in that receptor group. COPECs for surface water are for both estuarine (RM 0 to RM 13) and freshwater (RM 4 to RM 17.4) unless noted otherwise.

[b] Preliminary COCs are those COPECs with HQs ≥ 1.0 based on any LOE and effect-level concentration (i.e., HQ ≥ 1.0 based on a range of LOAELs for tissue and diet LOEs, HQ ≥ 1.0 based on acute or chronic surface water TRVs; HQ ≥ 1.0 based on plant-specific sediment TRVs).

[c] Preliminary COCs for regulated metals based on the tissue residue LOE were based on EFs rather than HQs.

BBP – butyl benzyl phthalate
BEHP – bis(2-ethylhexyl) phthalate
COC – chemical of concern
COI – chemical of interest
COPEC – chemical of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
EF – exceedance factor
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
LOE – line of evidence
LPAH – low-molecular-weight polycyclic aromatic hydrocarbon
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
PCDD – polychlorinated dibenzo-*p*-dioxin
PCDF –polychlorinated dibenzofuran
RM – river mile
SLERA – screening-level ecological risk assessment
SQT – sediment quality triad
TBT – tributyltin
TCDD – tetrachlorodibenzo-*p*-dioxin
TEQ – toxic equivalent
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT and 4,4′-DDT)
TRV – toxicity reference value
TSV – toxicity screening value

Table 13-2. Summary of LOAEL HQs and EFs for preliminary COCs

Preliminary COC	Risk Results (Range of LOAEL HQs and EFs[a])					
	Benthic Invertebrates	Fish	Birds	Mammals	Aquatic Plants	Zooplankton
Metals						
Arsenic	tissue: worm (2.2); blue crab (2.2)	*no unacceptable risk*	*no unacceptable risk*	*no unacceptable risk*	*no unacceptable risk*	*no unacceptable risk*
Cadmium	*no unacceptable risk*	diet: mummichog (1.3); common carp (1.2); white perch (1.1); white sucker (1.2); American eel (0.70–1.2)	*no unacceptable risk*	*no unacceptable risk*	*no unacceptable risk*	*no unacceptable risk*
Chromium	tissue: worm (6.0); mussel (3.7)	*no unacceptable risk*	*no unacceptable risk*	*no unacceptable risk*	sediment (160)	*no unacceptable risk*
Copper	surface water (estuarine: 0.14–2.7; freshwater: 0.034–1.0) tissue: blue crab (2.1)	surface water (estuarine: 0.14–2.7; freshwater: 0.023-1.0) tissue: mummichog (2.1), other forage fish (2.7), white perch (9.3), American eel (1.7)	diet: spotted sandpiper (0.30-3.6); great blue heron (0.029–1.3)	*no unacceptable risk*	sediment (2.4) surface water (estuarine: 1.8)	surface water (estuarine: 0.14–2.7; freshwater: 0.023–1.0)
Lead	tissue: worm (0.16–2.5)	*no unacceptable risk*	diet: spotted sandpiper (0.20–10); belted kingfisher (0.015–1.1)	*no unacceptable risk*	sediment (2.3)	*no unacceptable risk*
Methylmercury/mercury	tissue: blue crab: (1.3–1.5)	tissue: white catfish (0.71–1.1); American eel (0.74–1.1); largemouth bass (1.5–2.6); smallmouth bass (0.63–1.1) diet: mummichog (1.3); common carp (1.1); white perch (1.3); white catfish (1.1); American eel (1.1–1.3) Egg tissue: mummichog (0.11–1.1)	diet: great blue heron (0.031–1.6); belted kingfisher (0.13–1.6)	*no unacceptable risk*	sediment (9.7)	*no unacceptable risk*
Nickel	tissue: worm (12); mussel (6.0)	*no unacceptable risk*	*no unacceptable risk*	*no unacceptable risk*	*no unacceptable risk*	*no unacceptable risk*
Selenium	tissue worm (1.1); blue crab (1.5)	*no unacceptable risk*	*no unacceptable risk*	*no unacceptable risk*	sediment (1.8)	*no unacceptable risk*
Silver	tissue: blue crab (1.0)	*no unacceptable risk*	*not evaluated (no toxicity data available)*	*not evaluated (no toxicity data available)*	*no unacceptable risk*	*no unacceptable risk*
Vanadium	*no unacceptable risk*	*no unacceptable risk*	*no unacceptable risk*	*no unacceptable risk*	sediment (14)	*no unacceptable risk*
Zinc	*no unacceptable risk*	*no unacceptable risk*	*no unacceptable risk*	*no unacceptable risk*	sediment (3.1) surface water (estuarine; 21)	*no unacceptable risk*
Organometals						
TBT	*no unacceptable risk*	*no unacceptable risk*	*no unacceptable risk*	*no unacceptable risk*	surface water (estuarine: 1.1; freshwater: 50)	*no unacceptable risk*
PAHs						
HPAHs	tissue: worms (0.090–3.0)	*no unacceptable risk*	diet: spotted sandpiper (1.9–10)	*no unacceptable risk*	*no unacceptable risk*	*no unacceptable risk*
PCBs						
Total PCBs	tissue: worm (0.46–14.1), blue crab (0.67–21) , mussels (0.046–1.4)	tissue: mummichog (0.16–1.1), other forage fish (0.14–1.0), common carp (1.4–9.8), white perch (0.66–4.7), channel catfish (0.45–3.2), brown bullhead (0.37–2.6), white catfish (0.89–6.4), white sucker (0.76–5.5), American eel (0.53–3.8), largemouth bass (2.1–15), northern pike (0.53–3.8), smallmouth bass (0.37–2.6)	diet: spotted sandpiper (0.047–1.2), great blue heron (0.031–1.1)	diet: mink (0.94–3.1); river otter (2.6–3.7)	*no unacceptable risk*	*no unacceptable risk*

Table 13-2. Summary of LOAEL HQs and EFs for preliminary COCs

Preliminary COC	Risk Results (Range of LOAEL HQs and EFs[a])					
	Benthic Invertebrates	Fish	Birds	Mammals	Aquatic Plants	Zooplankton
		diet: northern pike (1.3)				
		egg tissue: mummichog (2.2–18)	egg tissue: great blue heron (0.078–284); belted kingfisher (0.22–76)			
PCB TEQ	*no unacceptable risk*	tissue: common carp (0.037–2.4), white perch (0.018–1.2), channel catfish (0.015–1.0); white catfish (0.029–1.9), white sucker (0.027–1.8), largemouth bass (0.14–9.4), northern pike (0.019-1.3)	diet: spotted sandpiper (0.073–3.9); great blue heron (0.030–1.6), belted kingfisher (0.10–1.5)	diet: mink (0.12–1.1); river otter (0.31–1.4)	*not evaluated*	*not evaluated*
		diet: white perch (1.0); American eel (0.95–1.8); largemouth bass (1.6); smallmouth bass (1.5); northern pike (2.1)	egg tissue: great blue heron (0.56–36); belted kingfisher (0.46–12)			
PCDD/PCDFs						
2,3,7,8-TCDD	surface water (estuarine: 0.0028–4.3)	tissue: mummichog (0.41–27), other forage fish (0.38–26), common carp (5.1–340), white perch (1.6–110), channel catfish: (0.80–53), brown bullhead (1.3–83), white catfish (1.8–120), white sucker (1.1–72), American eel (0.19–13), largemouth bass (1.5–100), northern pike (0.79–53), smallmouth bass (0.63–42)	*not evaluated*	*not evaluated*	*no unacceptable risk*	*no unacceptable risk*
	tissue: worm (0.013–29); blue crab (0.019–44); mussel (0.00073–1.7)					
PCDD/PCDF TEQ	tissue: worm (0.013–29), blue crab (0.021–48), mussel (0.00077–1.8)	tissue: mummichog (0.43–28), other forage fish (0.41–27), common carp (5.2–340), white perch (1.7–110) channel catfish (0.83–56), brown bullhead (1.3–89), white catfish (1.8–120), white sucker (1.1–72), American eel (0.20–13), largemouth bass (1.5–100), northern pike (0.83–56) smallmouth bass (0.63–42),	diet: spotted sandpiper (0.014–21), great blue heron (0.020–1.9), belted kingfisher (0.090–1.9)	diet: mink: (0.79–8.7), river otter (1.8–10)	*not evaluated*	*not evaluated*
		diet: mummichog (200), common carp (200), white perch (170), channel catfish (190) white catfish (160), white sucker (190), American eel (180-190) largemouth bass (150) smallmouth bass (140), northern pike (200)	egg tissue: great blue heron (0.42–37), belted kingfisher (0.38–14)			
Total TEQ	tissue: worm (0.013–30); blue crab (0.021–48); mussel (0.00077–1.8)	tissue: mummichog: (0.43–28), other forage fish: (0.41–27), common carp: (5.2–340), white perch: (1.7–110), channel catfish: (0.83–56), brown bullhead: (1.3–89), white catfish: (1.9–130), white sucker: (1.1–72), American eel: (0.21–14), largemouth bass: (1.5–100), northern pike: (0.92–61), smallmouth bass: (0.68–46)	diet: spotted sandpiper (0.089–25), great blue heron (0.044–3.5), belted kingfisher (0.18–3.1)	diet: mink (1.0–9.9), river otter (2.4–12)	*not evaluated*	*not evaluated*
		diet: mummichog (210); common carp (200); white perch (170); channel catfish (190); white catfish (160); white sucker (190); American eel (190-200); largemouth bass (150); smallmouth bass (140); northern pike (200)	egg tissue: great blue heron (1.0–74), belted kingfisher (0.85–23)			
Pesticides						
Total DDx	tissue: worm: (0.12–1.6), blue crab (0.52–6.8)	tissue: common carp (1.3–1.7)	diet: spotted sandpiper (0.018–1.4); great blue heron (0.020–2.4); belted kingfisher (0.066–1.8)	*no unacceptable risk*	*no unacceptable risk*	*no unacceptable risk*
			egg tissue: great blue heron (0.14–18); belted kingfisher (0.37–4.6)			

Table 13-2. Summary of LOAEL HQs and EFs for preliminary COCs

Preliminary COC	Risk Results (Range of LOAEL HQs and EFs[a])					
	Benthic Invertebrates	Fish	Birds	Mammals	Aquatic Plants	Zooplankton
Dieldrin	*no unacceptable risk*	tissue: common carp (0.28–1.4), channel catfish (0.24–1.2), American eel (0.27–1.4), largemouth bass (0.20–1.0), northern pike (0.22–1.1)	*no unacceptable risk*	*no unacceptable risk*	*no unacceptable risk*	*no unacceptable risk*
Other						
Cyanide	surface water (estuarine: 1.3–4.1; freshwater: 0.23–1.0)	surface water (estuarine: 1.6–5.3)	*not evaluated*	*not evaluated*	surface water (estuarine: 2.0)	surface water (estuarine: 1.6–5.3)

Note – Preliminary COCs are identified as those COPECs with HQs ≥ 1.0 based on any LOE and effect-level concentration (i.e., HQ ≥ 1.0 based on a range of LOAELs for tissue and diet LOEs, HQ ≥ 1.0 based on acute or chronic surface water TRVs; HQ ≥ 1.0 based on plant-specific sediment TRVs).

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

BERA – baseline ecological risk assessment
COC – chemical of concern
COPEC – chemical of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
EF – exceedance factor
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
LOE – line of evidence
LPRSA – Lower Passaic River study Area
NJDEP – New Jersey Department of Environmental Protection
NOAEL – no-observed-adverse-effect level
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
PCDD – polychlorinated dibenzo-p-dioxin
PCDF –polychlorinated dibenzofuran
TBT – tributyltin
TCDD – tetrachlorodibenzo-p-dioxin
TEQ – toxic equivalent
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT and 4,4′-DDT)
TRV – toxicity reference value

The results of this BERA will be used in the FS as a tool for risk managers to make potential remedial decisions for the LPRSA. Determining the potential for unacceptable ecological risk at the population level provides information regarding decisions to be made in the FS or other programmatic environmental management changes. The TRVs used to evaluate risk to various ecological receptor groups in this BERA are organism-level effects, including those that affect particular attributes of organisms within a population, such as survival, growth, and reproduction. Survival, growth, and reproduction of individual organisms may, in turn, affect populations of those organisms depending upon the magnitude and severity of the effect. However, population-level effects—such as size or density of population, population growth, or population survival—are more direct measures of influence on the population as a whole. Since BERAs evaluate ***populations*** as assessment endpoints, not individuals, a number of other factors—including the potential magnitude and severity of the effect, the ecological significance of the risk to the population, and the certainty of the assessment—should be evaluated to determine if a risk driver should be used to develop PRGs or RALs.

The preliminary COCs were further evaluated based on a comparison to background concentrations (USEPA 2016d) and the uncertainty of the assessment to identify risk drivers to be further evaluated in the FS.

13.1 Preliminary COCs Recommended as Risk Drivers

The following preliminary COCs are recommended as risk drivers for further evaluation in the FS:

- 2,3,7,8-TCDD
- PCDD/PCDF TEQ (fish, bird, and mammal)
- Total TEQ (fish, bird, and mammal)
- Total PCBs
- PCB TEQ (fish, bird, and mammal)
- Total DDx

The above-listed risk drivers are based on effect-level HQs exceeding 1.0 for various ecological receptor groups and LOEs. Some LOEs are more certain than others and should be evaluated prior to making any management decisions. Table 13-3 presents a summary of the risk drivers and considerations for risk management decisions regarding the assumptions used to derive HQs.

Table 13-3. Considerations for risk management on ecological risk drivers

Risk Driver and LOE	LOAEL HQ range[a,b] Based on TRV-A[c]	LOAEL HQ range[a,b] Based on TRV-B[d]	Risk Management Considerations
Benthic invertebrate community	No risk drivers were identified using the SQT analysis; however, SQT locations with impacts were identified as follows: • No, low, or likely low impacts (indicative of insignificant benthic invertebrate risk) relative to urban reference conditions were observed at ~37% of the 97 SQT locations. Medium, likely, or high impacts were observed at 63% of the 97 SQT locations. • Likely or high impacts were observed at ~31% of the 97 SQT locations. • At ~32% of the SQT locations, risk was unclear (medium impacts). Medium impacts suggest moderate chemical risk.		• The reference area chemistry and toxicity screens were conservative, which resulted in a dataset that may not represent realistic reference conditions. The quantitative analysis of uncertainty (Appendix P) provides an alternative screening process. • The sediment chemistry LOE was conservative and potentially unreliable for predicting actual effects in the LPRSA. The quantitative analysis of uncertainty (Appendix P) provides an alternative chemistry LOE. The multivariate analysis of SQT data (Appendix P) indicates that sediment chemical factors are potentially related to benthic community impacts and exacerbated by habitat variables. • The comparison of LPRSA SQT data to non-urban reference data was less relevant than the comparison of LPRSA data to urban reference data. Effects in the LPRSA associated with its urban setting were not addressed by the comparison of LPRSA SQT data to non-urban reference data. • Medium-impact conclusions of the SQT WOE analysis were uncertain because of disagreement between or within LOEs. The quantitative analysis of uncertainty (Appendix P) attempts to address these uncertainties. Moderate effects are possible at medium-impact stations. • Impacts at freshwater LPRSA SQT locations LPRT17A and LPRT17D were potentially influenced (at least in part) by differences between habitat conditions immediately below Dundee Dam and those in the area above Dundee Dam. The area above the dam has finer sediments than the area just below, which is predominately composed of coarse sand and cobble. In general, such sediments are not expected to have elevated sediment contamination.
Total PCBs			
Benthic invertebrate tissue	0.046–0.67 (mussels, worm, and blue crab)	**1.4**–**21** (mussels, worm, and blue crab)	• TRV-A based on an SSD value less than lowest measured LOAEL; TRV-A results in HQs < 1.0 • TRV-B based on whole-body tissue concentrations interpolated from measured egg tissue concentrations
Fish tissue	0.14–**2.1** (all LPRSA fish species evaluated)	**1.0**–**15** (all LPRSA fish species evaluated)	• TRV-A based on an SSD value less than lowest measured LOAEL • TRV-B based on changes in smolt seawater preference in Atlantic salmon • EPC for largemouth bass based on maximum tissue concentration due to sample size
Fish diet	**1.3** (northern pike)	ne	• LOAEL based on fecundity (number of eggs per female), but no significant reduction on egg weight or hatching rate was reported.
Fish egg	**2.2–3.6** (mummichog)	**11–18** (mummichog)	• TRV-A and TRV-B based on same literature source; TRV-A based on observed adverse effect on reproduction (reduced hatchability), and TRV-B based on reduced fecundity, but no effect on egg weight or hatchability • Mummichog egg concentration modeled using literature-based CFs and LPRSA mummichog-specific lipid content
Bird diet	0.031–0.70 (spotted sandpiper, great blue heron)	0.11–**1.2** (spotted sandpiper, great blue heron)	• TRV-A based on non-chicken reproduction; TRV-A results in HQs < 1.0 • TRV-B based on interpolated value from chicken hatchability data • Low HQs and HQs < 1.0 conflicted with bird egg LOE, which relied on literature-based BMFs to estimate egg tissue concentrations
Bird egg	0.078–**1.0** (great blue heron and belted kingfisher)	**1.0**–**284** (great blue heron and belted kingfisher)	• TRV-A based on non-chicken reproduction and limited dataset (two studies) • TRV-B based on interpolated value from chicken hatchability data • Uncertainty associated with use of literature-based BMFs to predict bird egg concentrations; species-specific BMF for heron and kingfisher based heron and kingfisher data, respectively, for comparison to TRV-A, and range of BMFs evaluated for comparison to TRV-B
Mammal diet	0.94–**3.2** (mink and river otter)	**1.1**–**3.7** (mink and river otter)	• TRV-A and TRV-B based on same literature source with slightly different ingestion rates and body weight assumptions used to derive TRV • HQ based on range of dietary scenarios and 2 exposure areas; HQs slightly lower based on site-wide exposure area (vs. > RM 10); mink HQs lower based on consumption of aquatic prey and assuming zero exposure from terrestrial prey
PCB TEQ			
Fish tissue	0.014–0.037 (0.010-0.74) (common carp, white perch, channel catfish, white catfish, white sucker, largemouth bass, northern pike)	**1.0**–**9.4** (common carp, white perch, channel catfish, white catfish, white sucker, largemouth bass, northern pike)	• TRV-A based on SSD within range of measured LOAELs evaluated; TRV-A results in HQs < 1.0 • Alternative TRV-A based on SSD with relatively poor visual and statistical fit to the empirical data and likely over-predicts risk; alternative SSD less than lowest measured LOAEL • TRV-B based on interpolated larvae concentration from egg tissue
Fish diet	**1.5**–**2.1** (American eel - large; largemouth bass; smallmouth bass; northern pike)	ne	• LOAEL TRV 2 orders of magnitude less than LOAELs reported for 2 other species

Table 13-3. Considerations for risk management on ecological risk drivers

Risk Driver and LOE	LOAEL HQ range[a,b]		Risk Management Considerations
	Based on TRV-A[c]	Based on TRV-B[d]	
Bird diet	0.030–0.78 (all bird species evaluated)	0.13–**3.9** (all bird species evaluated)	• TRV-A and TRV-B based on same literature source based on weekly injection of pheasants; TRV-A results in HQs < 1.0 • TRV-B extrapolated from study using interspecies extrapolation factor of 5 • High variability of bird TEFs and differences among species sensitivities to dioxin-like compounds • Low HQs and HQs < 1.0 conflicted with bird egg LOE, which relied on literature-based BMFs to estimate egg tissue concentrations
Bird egg	0.46–**7.2** (great blue heron and belted kingfisher)	0.57–**36** (great blue heron and belted kingfisher)	• TRV-A based on SSD with no chicken reproduction data (SSD not expected to have changed significantly with inclusion of chicken data) • TRV-B based on SSD inclusive of chicken reproduction data • TEQ sensitivities varied with Ah receptor; chickens in high-sensitivity group and great blue heron in low-sensitivity group • Uncertainty associated with use of literature-based BMFs to predict bird egg concentrations; species-specific BMF for heron and kingfisher based on heron and kingfisher data, respectively, for comparison to TRV-A, and range of BMFs evaluated for comparison to TRV-B
Mammal diet	0.12–0.34 (mink and river otter)	0.49–**1.4** (mink and river otter)	• TRV-A based on mink fed laboratory-prepared diet and TRV-B based on mink fed field-collected carp; TRV-A results in HQs < 1.0 • HQ based on range of dietary scenarios and 2 exposure areas; HQs slightly lower based on site-wide exposure area (vs. > RM 10); mink HQs lower based on consumption of aquatic prey and assuming zero exposure from terrestrial prey
PCDD/PCDF and total TEQ			
Benthic invertebrate tissue	0.00077–0.021 (PCDD/PCDF TEQ; worm, blue crab and mussels) 0.00077–0.021 (total TEQ; worm, blue crab and mussels)	**1.8**–**48** (PCDD/PCDF TEQ; worm, blue crab and mussels) **1.8**–**48** (total TEQ; worm, blue crab and mussels)	• TRV-A based on injected (not measured) concentration in crayfish; TRV-A results in HQs < 1.0 • TRV-B based on uncontrolled field data and limited sample size (n = 1 tissue composite); LOAEL based on relative reduction at Arthur Kill site compared to Sandy Hook site • Evaluation as TEQ (based on fish TEFs) questionable for invertebrates because there was limited evidence for ligand activation of the Ah (dioxin) cellular receptor in these organisms (i.e., they were not susceptible to the dioxin-like effects reported for vertebrates) (Van den Berg et al. 1998).
Fish tissue	0.20–**5.2** (**1.0**–**27**) (PCDD/PCDF TEQ-fish; all fish species evaluated) 0.21–**5.2** (**1.1**–**27**) (total TEQ-fish; all fish species evaluated)	**13**–**340** (PCDD/PCDF TEQ-fish; all fish species evaluated) **14**–**340** (total TEQ-fish; all fish species evaluated)	• TRV-A based on SSD within range of measured LOAELs evaluated • Alternative TRV-A based on SSD with relatively poor visual and statistical fit to the empirical data and likely over-predicts risk; alternative SSD less than lowest measured LOAEL • TRV-B based on interpolated larvae concentration from egg tissue
Fish diet	**140**–**200** (PCDD/PCDF TEQ-fish; all fish species evaluated) **140**–**210** (total TEQ-fish; all fish species evaluated)	ne	• LOAEL TRV 2 orders of magnitude less than LOAELs reported for 2 other species
Bird diet	0.014–**4.2** (PCDD/PCDF TEQ - bird; all bird species evaluated) 0.044–**5.0** (total TEQ - bird; all bird species evaluated)	0.071–**21** (PCDD/PCDF TEQ - bird; all bird species evaluated) 0.22–**25** (total TEQ - bird; all bird species evaluated)	• TRV-A and TRV-B based on same literature source based on weekly injection of pheasants • TRV-B extrapolated from study using interspecies extrapolation factor of 5 • High variability of bird TEFs and differences among species sensitivities to dioxin-like compounds
Bird egg	0.38–**7.5** (PCDD/PCDF TEQ - bird; great blue heron and belted kingfisher) **1.0**–**15** (total TEQ - bird; great blue heron and belted kingfisher)	0.43–**37** (PCDD/PCDF TEQ - bird; great blue heron and belted kingfisher) **1.0**–**74** (total TEQ - bird; great blue heron and belted kingfisher)	• TRV-A based on SSD with no chicken reproduction data (SSD not expected to have changed significantly with inclusion of chicken data) • TRV-B based on SSD inclusive of chicken reproduction data • TEQ sensitivities varied with Ah receptor; chickens in high-sensitivity group and great blue heron in low-sensitivity group • Species-specific BMF for heron and kingfisher based heron and kingfisher data, respectively

Table 13-3. Considerations for risk management on ecological risk drivers

Risk Driver and LOE	LOAEL HQ range[a,b] Based on TRV-A[c]	LOAEL HQ range[a,b] Based on TRV-B[d]	Risk Management Considerations
Mammal diet	0.79–**2.6** (PCDD/PCDF TEQ-mammal; mink and river otter) **1.0**–**2.9** (total TEQ-mammal; mink and river otter)	**3.2–10** (PCDD/PCDF TEQ-mammal; mink and river otter) **4.1**–**12** (total TEQ-mammal; mink and river otter)	• TRV-A based on mink fed laboratory-prepared diet and TRV-B based on mink fed field-collected carp • HQ based on range of dietary scenarios and 2 exposure areas; HQs slightly lower based on site-wide exposure area (vs. > RM 10); mink HQs lower based on consumption of aquatic prey and assuming zero exposure from terrestrial prey
Total DDx			
Benthic invertebrate tissue	0.15–0.62 (**1.6**–**6.8**) (worm and blue crab)	0.12–0.52 (worm and blue crab)	• TRV-A and alternative TRV-A based on SSD less than lowest measured LOAEL • Alternative TRV-A based on relatively poor visual and statistical fit to the empirical data and likely overestimates toxicity
Fish tissue	**1.3** (common carp)	**1.7** (common carp)	• TRV-A based on SSD less than lowest measured LOAEL evaluated • TRV-B based on SSD within range of measured LOAELs evaluated (which included TRVs based on field-collected organisms) • HQs < 1.0 for other 11 of 12 fish species evaluated
Bird diet	0.018–0.26 (all bird species evaluated)	0.16–**2.4** (all bird species evaluated)	• TRV-A based on SSD within range of measured LOAELs evaluated; TRV-A results in HQs < 1.0 • TRV-B based on field study of eggshell thinning in pelicans • Low HQs and HQs < 1.0 conflicted with bird egg LOE, which relied on literature-based BMFs to estimate egg tissue concentrations
Bird egg	0.14–**1.8** (great blue heron and belted kingfisher)	0.19–**18** (great blue heron and belted kingfisher)	• TRV-A based on SSD not inclusive of chicken reproduction data • TRV-B based on SSD inclusive of chicken reproduction data • Uncertainty associated with use of literature-based BMFs to predict bird egg concentrations; species-specific BMF for heron based heron data, and species-specific BMF for kingfisher based on geomean of 5 species for comparison to TRV-A and range of BMFs evaluated for comparison to TRV-B

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

[b] HQs presented are based on LOAEL TRVs.

[c] TRVs were derived from the primary literature review.

[d] TRVs based on USEPA's revised draft of the LPR restoration project FFS (Louis Berger et al. 2014) or first draft of the LPR restoration project FFS (Malcolm Pirnie 2007b).

Ah – aryl hydrocarbon
BERA – baseline ecological risk assessment
BMF – biomagnification factor
CF – conversion factor
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
EPC – exposure point concentration
FFS – focused feasibility study
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
LOE – line of evidence
LPR – Lower Passaic River
LPRSA – Lower Passaic River Study Area
ne – not evaluated
NJDEP – New Jersey Department of Environmental Protection
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
PCDD – polychlorinated dibenzo-*p*-dioxin
PCDF –polychlorinated dibenzofuran
RM – river mile
SSD – species sensitivity distribution
SQT – sediment quality triad
TEF – toxic equivalency factor
TEQ – toxic equivalent
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT and 4,4′-DDT)
TRV – toxicity reference value
USEPA – US Environmental Protection Agency
WOE – weight of evidence

13.2 Preliminary COCs Not Recommended as Risk Drivers

A number of preliminary COCs were not recommended as risk drivers to be carried forward to inform major risk management decisions. Preliminary COCs that were not retained as risk drivers were excluded primarily for two reasons:

- Background concentrations indicated that risks in the LPRSA would not be different or would be less than those in background (upstream or regional) areas.
- The LOE for which a LOAEL HQ was ≥ 1.0 could not reliably predict risks to a level appropriate for costly remedial decisions. This included the tissue residue LOE for metals[131] and the sediment LOE for aquatic plants.[132]

Eleven metals (arsenic, cadmium, chromium, copper, lead, mercury, nickel, silver, selenium, vanadium, and zinc), TBT, HPAHs, dieldrin, and cyanide were not recommended as risk drivers based on background concentrations and/or the uncertainty of the LOE for remedial decisions.

- **Arsenic** – Arsenic was identified as a preliminary COC based on benthic invertebrate tissue LOE (worm HQ = 2.2 and blue crab HQ = 2.2). Arsenic was not recommended as a risk driver because of the uncertainty associated with the evaluation of regulated metals in tissue[133] as a LOE. In addition, the LPRSA exposure point concentration (EPC) for sediment (9.6 mg/kg) was less than regional background (i.e., Jamaica Bay and Mullica River/Great Bay) maximum concentrations (20.7 and 32.8 mg/kg at Jamaica Bay and Mullica River/Great Bay, respectively) and the upper confidence limit on the mean (UCL) for the Mullica River/Great Bay (12 mg/kg). However, the LPRSA EPC for sediment (9.6 mg/kg) was slightly greater than the UCL for Jamaica Bay (7.3 mg/kg) and above Dundee Dam (6.4 mg/kg).

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[131] The use of a tissue residue approach for metals (except for methylmercury and selenium) was highly uncertain because of the wide range of strategies used by organisms to store, detoxify, and excrete bioaccumulated metals (e.g., fish and invertebrates may regulate their body burdens of some metals, although metals regulation, and the strategy thereof, is species and metal specific) (Adams et al. 2011; USEPA 2007e). The USEPA framework for metals risk assessment (USEPA 2007e) recommends against the use of a tissue residue approach, stating that the CBR approach for metals "does not appear to be a robust indicator of toxic dose."

[132] The relevance of soil toxicity thresholds generated for agricultural crops to the sediment exposure of aquatic plants was unknown. Soil toxicity data were primarily from studies of agricultural crops and/or waste soils. Many of these soil toxicity data were based on a highly bioavailable chemical form that is not representative of natural soils or sediment (Efroymson et al. 1997).

[133] The use of a tissue residue approach for metals (except for methylmercury and selenium) was highly uncertain because of the wide range of strategies used by organisms to store, detoxify, and excrete bioaccumulated metals (e.g., fish and invertebrates may regulate their body burdens of some metals, although metals regulation, and the strategy thereof, is species and metal specific) (Adams et al. 2011; USEPA 2007e).

- **Cadmium** – Cadmium was identified as a preliminary COC based on the fish diet LOE (HQs for mummichog, common carp, white perch, white sucker, and American eel ranged from 0.70 to 1.3). Cadmium was not identified as a preliminary COC for any other LOE or receptor group, and HQs for fish diet were just above 1.0 for several fish species. This identification was consistent with recommendations by USEPA (2007e). USEPA recommends a dietary assessment of inorganic metals for conservative screening purposes only, because the uptake by and toxicity of inorganic metals to fish can vary widely depending upon a number of factors, including (but not limited to) digestive physiology (e.g., gut residence time), food nutritional quality, distribution and chemical form of metals in prey tissue, and environmental conditions under which toxicity is evaluated (e.g., temperature).
- **Chromium** – Chromium was identified as a preliminary COC based on the benthic invertebrate tissue LOE (worm HQ = 6.0 and mussel HQ = 3.7) and aquatic plants and sediment LOE (HQ = 160). Chromium was not recommended as a risk driver based on the uncertainty of the aquatic plant assessment for sediment. This uncertainty was due to a screening level based on a highly bioavailable chemical form,[134] as well as the uncertainty associated with the evaluation of regulated metals in tissue.[135]
- **Copper** – Copper was identified as a preliminary COC based on the following LOEs: benthic invertebrate tissue (blue crab HQ = 2.1), fish tissue (mummichog, other forage fish, white perch, and American eel HQs ranged from 1.7 to 9.3), bird diet (sandpiper and great blue heron HQs ranged from 0.029 to 3.6), surface water (benthic invertebrate, fish, zooplankton, and aquatic plant estuarine and freshwater HQs ranged from 0.14 to 2.7 and from 0.023 to 1.0, respectively), and sediment for aquatic plant populations (HQ = 2.4). Copper was not recommended as a risk driver for the following reasons:
    - Uncertainty associated with the evaluation of regulated metals in tissue.[136]

[134] The relevance of soil toxicity thresholds generated for agricultural crops to the sediment exposure of aquatic plants was unknown. Soil toxicity data were primarily from studies of agricultural crops and/or waste soils. Many of these soil toxicity data were based on a highly bioavailable chemical form that is not representative of natural soils or sediment (Efroymson et al. 1997).

[135] The use of a tissue residue approach for metals (except for methylmercury and selenium) was highly uncertain because of the wide range of strategies used by organisms to store, detoxify, and excrete bioaccumulated metals (e.g., fish and invertebrates may regulate their body burdens of some metals, although metals regulation, and the strategy thereof, is species and metal specific) (Adams et al. 2011; USEPA 2007e).

[136] The use of a tissue residue approach for metals (except for methylmercury and selenium) was highly uncertain because of the wide range of strategies used by organisms to store, detoxify, and excrete bioaccumulated metals (e.g., fish and invertebrates may regulate their body burdens of some metals, although metals regulation, and the strategy thereof, is species and metal specific) (Adams et al. 2011; USEPA 2007e).

- Uncertainty associated with the aquatic plant assessment for sediment due to a screening level based on a highly bioavailable chemical form.[137]
- Evaluation of background. Dissolved estuarine surface water LPRSA EPCs for copper (2.61 µg/L) were less than the maximum (3.36 µg/L) and UCL (2.7 µg/L) background surface water concentrations above Dundee Dam. Sediment LPRSA EPCs for copper (170 mg/kg) were less than or similar to maximum (209 mg/kg) and UCL (150 mg/kg) background sediment concentration above Dundee Dam.

- **Lead** – Lead was identified as a preliminary COC based on benthic invertebrate tissue LOE (worm HQs ranged from 0.16 to 2.5), bird diet LOE (spotted sandpiper HQs ranged from 0.20 to 10, and belted kingfisher HQs ranged from 0.015 to 1.1), and the sediment LOE for aquatic plant populations (HQ = 2.3). Lead was not recommended as a risk driver based on benthic invertebrate tissue due to uncertainty associated with the evaluation of regulated metals in tissue,[138] uncertainty of the aquatic plant assessment for sediment due to a screening level based on a highly bioavailable chemical form,[139] and the background evaluation. The LPRSA EPC for lead in sediment (270 mg/kg) was less than the UCL (440 mg/kg) background concentration above Dundee Dam.
- **Methylmercury/mercury** – Methylmercury/mercury was identified as a preliminary COC based on the following LOEs: benthic invertebrate tissue (blue crab HQs ranged from 1.3 to 1.5), fish tissue (white catfish, American eel, largemouth bass, and smallmouth bass HQs ranged from 0.63 to 2.6), fish diet (mummichog, common carp, white perch, white catfish, and American eel HQs ranged from 1.1 to 1.3), fish egg tissue (mummichog HQs ranged from 0.11 to 1.1), bird diet (great blue heron and kingfisher HQs ranged from 0.031 to 1.6), and sediment for aquatic plant populations (HQ = 9.7). Methylmercury and mercury were not recommended as risk drivers for the following reasons:
  - Evaluation of background. The sediment LPRSA EPC for mercury (2,900 µg/kg) was less than the UCL (2,910 µg/kg) background sediment

[137] The relevance of soil toxicity thresholds generated for agricultural crops to the sediment exposure of aquatic plants was unknown. Soil toxicity data were primarily from studies of agricultural crops and/or waste soils. Many of these soil toxicity data were based on a highly bioavailable chemical form that is not representative of natural soils or sediment (Efroymson et al. 1997).

[138] The use of a tissue residue approach for metals (except for methylmercury and selenium) was highly uncertain because of the wide range of strategies used by organisms to store, detoxify, and excrete bioaccumulated metals (e.g., fish and invertebrates may regulate their body burdens of some metals, although metals regulation, and the strategy thereof, is species and metal specific) (Adams et al. 2011; USEPA 2007e).

[139] The relevance of soil toxicity thresholds generated for agricultural crops to the sediment exposure of aquatic plants was unknown. Soil toxicity data were primarily from studies of agricultural crops and/or waste soils. Many of these soil toxicity data were based on a highly bioavailable chemical form that is not representative of natural soils or sediment (Efroymson et al. 1997).

concentration above Dundee Dam. In addition, LPRSA methylmercury fish tissue EPCs were less than maximum concentrations for 7 of 10 species above Dundee Dam, and similar to or less than UCLs for 3 of 4 fish species for which UCLs above Dundee Dam could be calculated. Mummichog LPRSA EPCs for methylmercury were less than UCLs in mummichog from Jamaica Bay/Lower Harbor.

- Uncertainty associated with the aquatic plant assessment for sediment due to a screening level based on a highly bioavailable chemical form.[140]
- Uncertainty associated with the bird diet. The TRV resulting in HQs > 1.0, which was derived using an interspecies extrapolation factor of 3 (assumed mallards were three 3 less sensitive than the selected avian species evaluated), and was based on exposure to methylmercury dicyandiamide, a fungicide that is not a form of mercury expected to be associated with the LPRSA.

- **Nickel** – Nickel was identified as a preliminary COC based on the benthic invertebrate tissue LOE (worm HQ = 12 and blue crab HQ = 6.0). Nickel was not recommended as a risk driver based on the uncertainty associated with the evaluation of regulated metals in tissue.[141]
- **Silver** – Silver was identified as a preliminary COC based on the benthic invertebrate tissue LOE (blue crab HQ = 1.0). Silver was not recommended as a risk driver based on uncertainty associated with the evaluation of regulated metals in tissue.[142]
- **Selenium** – Selenium was identified as a preliminary COC based on the benthic invertebrate tissue (worm HQ = 1.1 and blue crab HQ = 1.5) and aquatic plant sediment (HQ = 14) LOEs. Selenium was not recommended as a risk driver based on the uncertainty of the aquatic plant assessment for sediment. This uncertainty was due to a screening level based on a highly bioavailable chemical

[140] The relevance of soil toxicity thresholds generated for agricultural crops to the sediment exposure of aquatic plants was unknown. Soil toxicity data were primarily from studies of agricultural crops and/or waste soils. Many of these soil toxicity data were based on a highly bioavailable chemical form that is not representative of natural soils or sediment (Efroymson et al. 1997).

[141] The use of a tissue residue approach for metals (except for methylmercury and selenium) was highly uncertain because of the wide range of strategies used by organisms to store, detoxify, and excrete bioaccumulated metals (e.g., fish and invertebrates may regulate their body burdens of some metals, although metals regulation, and the strategy thereof, is species and metal specific) (Adams et al. 2011; USEPA 2007e).

[142] The use of a tissue residue approach for metals (except for methylmercury and selenium) was highly uncertain because of the wide range of strategies used by organisms to store, detoxify, and excrete bioaccumulated metals (e.g., fish and invertebrates may regulate their body burdens of some metals, although metals regulation, and the strategy thereof, is species and metal specific) (Adams et al. 2011; USEPA 2007e).

form.[143] In addition, selenium was not recommended as a risk driver based on a comparison to background; the LPRSA sediment concentration (0.93 mg/kg) was less than the UCL and maximum concentrations above Dundee Dam (27 and 2.7[144] mg/kg, respectively) and the UCL from Jamaica Bay (1.4 mg/kg).

- **Vanadium** - Vanadium was identified as a preliminary COC based on the sediment LOE for aquatic plants (HQ = 14). Vanadium was not recommended as a risk driver based on the uncertainty of the aquatic plant assessment for sediment. This uncertainty was due to a screening level based on a highly bioavailable chemical form.[145]

- **Zinc** – Zinc was identified as a preliminary COC based on the LOEs for sediment for aquatic plants (HQ = 3.1) and surface water for aquatic plants (HQ = 21). Zinc was not recommended as a risk driver based on the uncertainty of the aquatic plant assessment for sediment. This uncertainty was due to a screening level based on a highly bioavailable chemical form.[146] In addition, zinc was not recommended as a risk driver based a comparison to background; LPRSA estuarine and freshwater surface water EPCs for dissolved zinc (8.5 and 7.5 µg/L, respectively) were less than the background maximum dissolved zinc concentration above Dundee Dam (9.8 µg/L). In addition, zinc concentrations in surface water based on general surface water criteria for the evaluation of other aquatic receptor groups (i.e., invertebrates, fish, and zooplankton) resulted in HQs < 1.0.

- **TBT** – TBT was identified as a preliminary COC based on aquatic plant populations (surface water HQs ranged from 1.1 to 50). TBT was not recommended as a risk driver based on the background evaluation; surface water EPCs for TBT were represented by maximum concentrations (0.026 µg/L) and DLs (0.05 µg/L) in the LPRSA. The maximum LPRSA TBT concentrations were less than the DL for background surface water above Dundee Dam

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143 The relevance of soil toxicity thresholds generated for agricultural crops to the sediment exposure of aquatic plants was unknown. Soil toxicity data were primarily from studies of agricultural crops and/or waste soils. Many of these soil toxicity data were based on a highly bioavailable chemical form that is not representative of natural soils or sediment (Efroymson et al. 1997).

144 Maximum background concentrations were derived excluding outliers. UCL background concentrations were derived including all data. Details of the background evaluation are provided in Appendix J of this BERA.

145 The relevance of soil toxicity thresholds generated for agricultural crops to the sediment exposure of aquatic plants was unknown. Soil toxicity data were primarily from studies of agricultural crops and/or waste soils. Many of these soil toxicity data were based on a highly bioavailable chemical form that is not representative of natural soils or sediment (Efroymson et al. 1997).

146 The relevance of soil toxicity thresholds generated for agricultural crops to the sediment exposure of aquatic plants was unknown. Soil toxicity data were primarily from studies of agricultural crops and/or waste soils. Many of these soil toxicity data were based on a highly bioavailable chemical form that is not representative of natural soils or sediment (Efroymson et al. 1997).

(0.05 µg/L), and the LPRSA DLs were equal to background DLs from above Dundee Dam. In addition, TBT had a low detection frequency in the surface water of the LPRSA (0 to 1%).

- **HPAHs -** Total HPAHs were identified as a preliminary COC based on the benthic invertebrate tissue LOE for worms (HQs ranged from 0.090 to 3.0) and the bird diet LOE for spotted sandpiper (HQs ranged from 1.9 to 10 by reach; HQ = 4.5 site wide). Total HPAHs were not recommended as a risk driver based on the background evaluation; the LPRSA sediment EPC (46,000 µg/kg) was less than both the EPC and the maximum sediment concentration above Dundee Dam (300,000 and 73,300 µg/kg[147], respectively). No background invertebrate tissue data were available for comparison to LPRSA invertebrate concentrations, so there was some uncertainty with this evaluation.
- **Dieldrin** – Dieldrin was identified as a preliminary COC based on the fish tissue LOE for several fish species: common carp, channel catfish, American eel, largemouth bass, and northern pike (HQs ranged from 0.20 to 1.4). The two TRVs used to determine the HQs were derived from the same study (Shubat and Curtis 1986). The higher LOAEL TRV was based on unadjusted data from the 16-week study wherein reduced growth of rainbow trout was observed, and the lower LOAEL TRV was based on 96-hr LC50 data adjusted using extrapolation factors. Given that the HQs were relatively low based on the LOAEL TRV that was adjusted using extrapolation factors, remedial action based on these predicted risks was not recommended. In addition, dieldrin was not recommended as a risk driver based on the background evaluation; the LPRSA sediment EPC (8.3 µg/kg) was less than the EPC above Dundee Dam (17 µg/kg).
- **Cyanide** – Cyanide was identified as a preliminary COC based on surface water (for invertebrate populations [estuarine and freshwater HQs ranged from 1.3 to 4.1 and from 0.23 to 1.0, respectively], fish and zooplankton populations [estuarine HQs ranged from 1.6 to 5.3], and aquatic plant populations [estuarine HQ = 2.0]). Cyanide was not recommended as a risk driver due to its low detection frequency in surface water in the LPRSA; less than 6% of samples in the estuarine portion had detected concentrations of cyanide.

[147] Maximum background concentrations were derived excluding outliers. UCL background concentrations were derived including all data. Details of the background evaluation are provided in Appendix J of this BERA.

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