FINAL

Philadelphia Naval Complex

Screening-Level Risk Assessment OF Reserve Basin Sediments

PREPARED FOR

U.S. Department of the Navy
Northern Division
Naval Facilities Engineering Command
10 Industrial Highway, Mail Stop No. 82
Lester, Pennsylvania 19113-2090

National Oceanic and Atmospheric Administration
Office of Response and Restoration
Coastal Protection and Restoration Division
7600 Sand Point Way N.E.
Seattle, Washington 98115

WITH

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200 West Mercer Street, Suite 403
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LIST OF ACRONYMS

AE	assimilation efficiency
ARCS	USEPA Assessment and Remediation of Contaminated Sediments program
AVS	acid-volatile sulfide
BW	body weight
COC	contaminant of concern
COPC	contaminant of potential concern
FMR	field metabolic rate
GE	gross energy
HPAH	high-molecular-weight polynuclear aromatic hydrocarbon
HQ	hazard quotient
LC_{50}	median lethal concentration
LD_{50}	median lethal dose
LOEL	Lowest observed effects level
LPAH	low-molecular-weight polynuclear aromatic hydrocarbon
ME	metabolizable energy
NOAEL	no observed adverse effects level
OC	organic carbon
PAH	polynuclear aromatic hydrocarbon
PCB	polychlorinated biphenyl
PEL	Probable Effects Level
PNC	Philadelphia Naval Complex
ROC	receptor of concern
RM	river mile
SEM	simultaneously extracted metals
SLERA	screening level ecological risk assessment
TEL	Threshold Effects Level
TRV	toxicity reference value
USEPA	U.S. Environmental Protection Agency

1.0
INTRODUCTION

This document is a Screening-Level Ecological Risk Assessment (SLERA) of sediments in the Reserve Basin at the Philadelphia Naval Complex (PNC). This document was prepared in accordance with U.S. Environmental Protection Agency (USEPA) Risk Assessment Guidance (USEPA 1997) and Navy policy (CNO 1999). This report is organized into ten sections. Section 2.0 presents results from the problem formulation, including discussions of resources and risk; selection of contaminants of potential concern (COPCs); the fate, transport, and ecotoxicity of COPCs; selection of receptors of concern (ROCs); identification of exposure pathways; and assessment and measurement endpoints. Section 3.0 evaluates risk to benthic invertebrates, Section 4.0 evaluates risk to fish, and Section 5.0 is an evaluation of risk to birds. Section 6.0 is a qualitative analysis of the uncertainty in these risk evaluations. Section 7.0 provides the results of additional evaluations that were conducted. Section 8.0 summarizes the results of risk evaluations conducted as part of the SLERA. Section 9.0 recommends further work that could be conducted as part of a baseline risk assessment. Section 10.0 is a list of references.

The screening-level problem formulation was developed using only existing information contained in the following reports:

> EA Engineering, Science, and Technology, Inc 1998. *Characterization of Aquatic Habitats and Resources Near the Philadelphia Naval Complex*. Lester, Pennsylvania: U.S. Department of the Navy, Northern Division.;
>
> EA Engineering, Science, and Technology, Inc. 1996. *IR Program Site 8 (Reserve Basin) Further Characterization of Sediments in Philadelphia Naval Complex, Philadelphia PA, Draft*. Lester, Pennsylvania: U.S. Department of the Navy, Northern Division.; and,
>
> Boyd, T.J., M.T. Montgomery, B.J. Spargo, R.B. Coffin. 1999. *Characterization of intrinsic bioremediation within the Philadelphia Naval Complex Reserve Basin, Interim Report*. NRL/PU/6115-99-374. Washington, D.C.: Naval Research Laboratory. 91 pp.

2.0
ENVIRONMENTAL SETTING AND CONTAMINANTS

2.1 SITE HISTORY

The Reserve Basin is a mooring basin in an industrial area located in the western portion of the PNC (Figure 1). The Reserve Basin covers approximately 44 hectares, with water depths ranging from 6 to 9.5 m (mean lower low water). The basin opens to the Schuylkill River, a tidally influenced freshwater system. The mouth of the basin is located approximately 1,000 m north of the confluence of the Schuylkill and Delaware rivers. The confluence is located at river mile (RM) 92, where RM 0 is at the entrance to the Delaware estuary, near Cape Henlopen.

The Reserve Basin serves as a moorage area for the Navy's inactive ships. Ship moorage in the Reserve Basin may result in the release of contaminants. For example, antifoulant paints on the ships' hulls may act as a source of metal contamination, especially for copper and zinc. Because inactive ships are sealed and do not normally collect bilge water, discharges of this sort are not considered a probable source of contamination in the Reserve Basin.

There have been three reported oil spills in the Reserve Basin: the first, in 1964, was a 75,000-gallon spill of bunker C oil; in 1980, two 400-gallon spills of No. 6 fuel oil occurred near the entrance to the Reserve Basin. In all incidents, an oil skimmer reclaimed as much oil as possible. The presence of residual oil may be reflected in the fact that sediment polynuclear aromatic hydrocarbon (PAH) concentrations measured in the Reserve Basin are higher than concentrations in Schuylkill River sediments (EA Engineering 1996a).

Several contaminated upland sites adjacent to the Reserve Basin, including the Girard Point Landfill (Site 4), the Girard Point Blasting Grit Disposal Area (Site 5), and an aboveground fuel storage tank area (Site 12), may add to the contamination of basin sediments (Figure 2). It has been suggested that the spent blasting grit disposed of at the Girard Point Blasting Grit Disposal Area may have eroded into the Reserve Basin as a result of tidal action (EA Engineering 1996a).

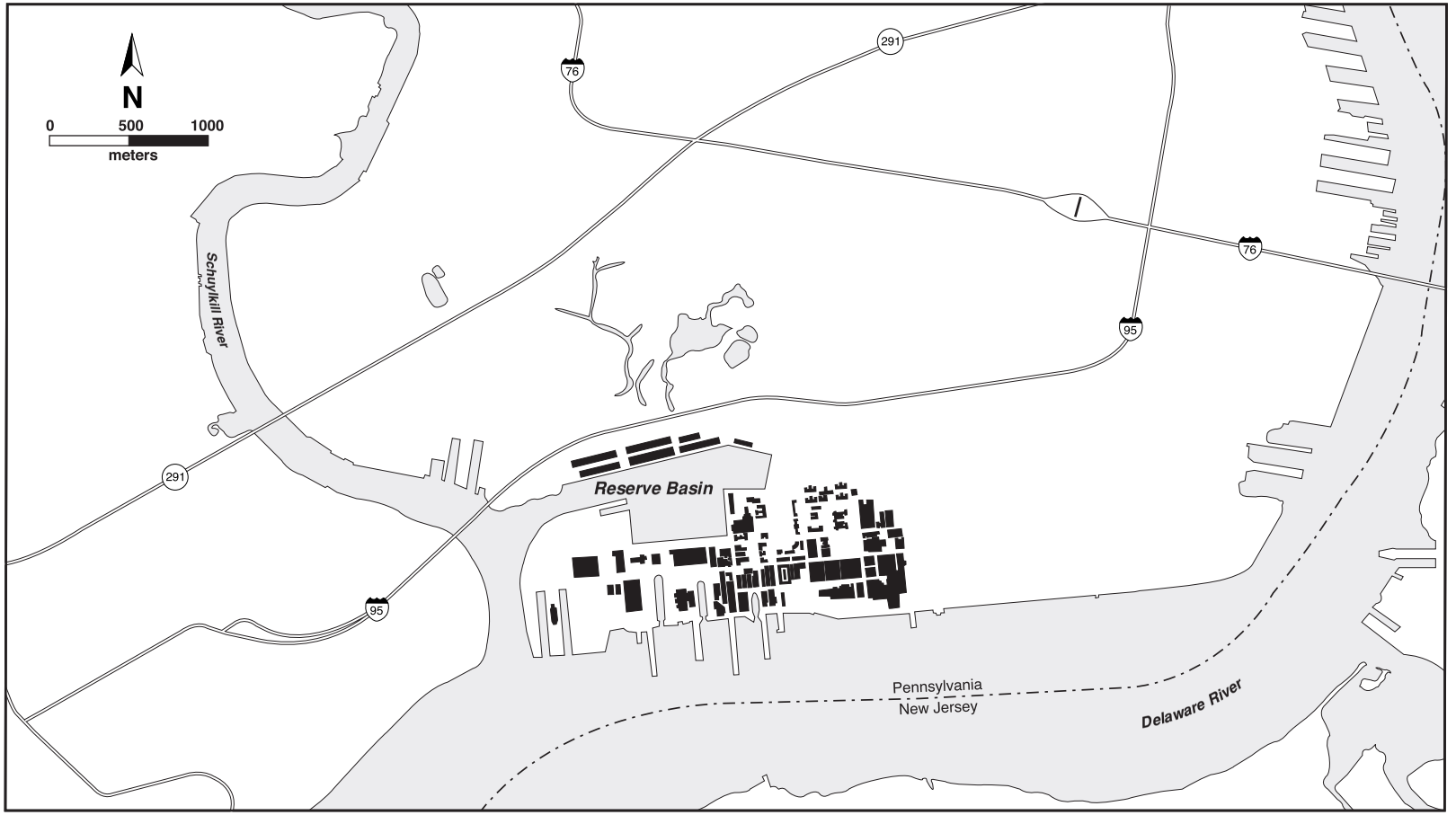

Figure 1. Philadelphia Naval Complex, Philadelphia, Pennsylvania

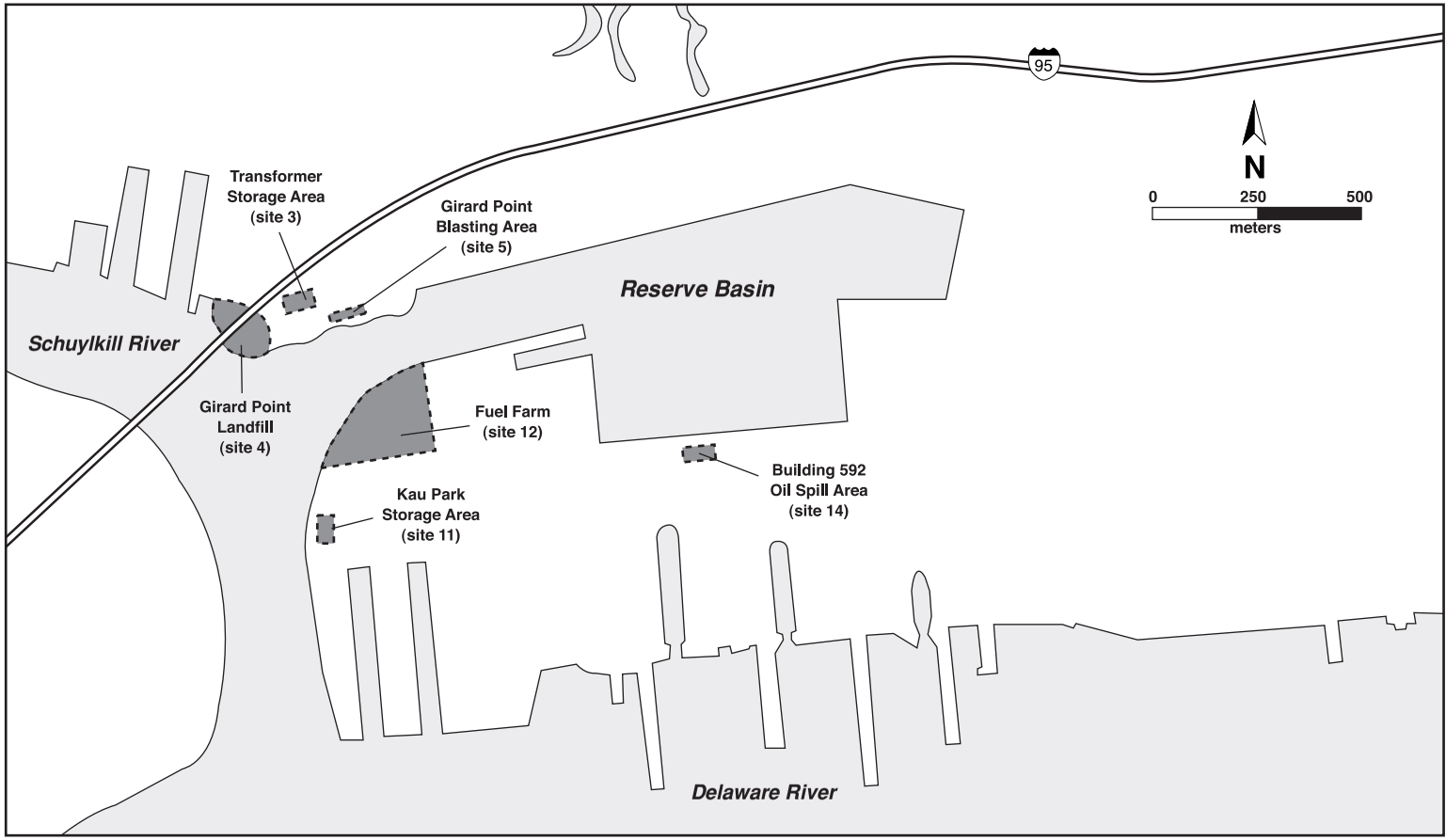

Figure 2. Philadelphia Naval Complex – Reserve Basin

Several actions have been taken to address upland sources of contamination to the Reserve Basin including:

- River bank stabilization at Site 4 in 1994 and Site 5 in 1996
- Stormwater sewer engineering survey, including line cleaning for Sites 4 and 5 in 1997
- Underground storage tank removals for all of the Girard Point Management Area in 1996, 1997, and 1998
- Removal of PCB contaminated soil from areas throughout the Girard Point Management Area in 1992, 1995, and 1996.

The final remedy selected for Sites 4 and 5 was the development of vegetative cover.

In addition to upland sources, other potential sources of contamination to the Reserve Basin include stormwater discharge into the basin and tidal exchange with the Schuylkill River. There are 14 permitted outfalls at the PNC, four of which discharge directly to the Reserve Basin (EA Engineering 1996a). Additionally, 40 pipes discharge stormwater or non-stormwater to either the Reserve Basin or the entrance to the Basin. The Reserve Basin receives stormwater runoff from both the PNC and the City of Philadelphia storm drainage system. Stormwater also enters the Reserve Basin directly as sheet flow.

Contaminants associated with surface water and suspended sediments may also be transported into the Reserve Basin through tidal exchange with the Schuylkill River. EA Engineering conducted hydrodynamic and sediment transport modeling (EA Engineering 1996a). To perform the modeling, it was necessary to make several very broad assumptions. Within the constraints of this effort, and with the numerous assumptions made for certain input parameters, this modeling suggested that the majority of the suspended particulate material derived from stormwater outfalls may be deposited within the basin. However, this amount may represent only a small fraction of the total amount of sediment deposited, since the majority of fluid input may occur from the river. So, the model also predicted possible river sediment deposition. Additionally, this modeling effort suggested that predicted currents within the basin are not strong enough to scour the previously deposited sediments from the Reserve Basin to the Schuylkill River.

2.2 BIOLOGICAL COMMUNITIES

This screening of ecological risk will evaluate the Reserve Basin. No data specific to the aquatic biological communities within the Reserve Basin were available for review within the scope of this study. However, the following description of biological communities was obtained from studies summarized by EA Engineering (1998) conducted in the

immediate vicinity of the Reserve Basin between RM 90 and 95 in the Delaware River. Because of the close proximity of the Reserve Basin and river, it is expected that similar habitats and species would occupy both areas.

A macroinvertebrate community associated with fine-grained, organically enriched sediments is present in this reach of the Delaware River as well as populations of fish and aquatic birds.

2.2.1 Macroinvertebrate Community

There have been several studies of the benthic community within the region, though typically at low spatial resolution. Most studies of the lower, tidal freshwater regions of the Delaware River have found benthic infauna dominated by a few taxa. Oligochaetes (freshwater aquatic worms) were the most abundant taxon in the study area and composed 52.4 percent of the total mean density of macroinvertebrates. Tubificidae (sludge worms) was the dominant family with *Limnodrilus* the most abundant genus. The Enchytraeids (microdrile worms) were the second most abundant oligochaete family. Chironomids (midges), the second most abundant taxon, composed 23.5 percent of the total density in the lower river near the study area. Within the chironomids, the subfamily Chironominae was the most common group, and *Polypedilum* was the most abundant genus, followed by *Cladotanytarsus* and *Cryptochironomus*. The amphipods (scuds), isopods (aquatic sow bugs), and polychaetes (marine worms) were the third, fourth, and fifth most abundant taxa, respectively.

The benthic communities observed in the vicinity of the study area sediments are considered tolerant of pollution and disturbance. Most occupy habitats of very fine sediments with high organic content. These area studies have not identified specific gradients or sites of ecological impairment, although some studies have noted a temporal trend of increased health in the macroinvertebrate community. The community is still considered pollution-tolerant, but compared to earlier studies, there is an increased diversity of oligochaetes and chironomids, and the most pollution-tolerant tubificid taxa have been absent in the most recent surveys.

2.2.2 Fish Community

Finfish surveys have been conducted in the Delaware River between RM 92.5 (e.g., the mouth of the Schuykill) and 95.5 (Table 1). Data indicate that the community is composed of numerous anadromous, resident, and estuarine species. The anadromous white perch dominated the fish community in the study area, comprising 27.6 percent of the total number of fish sampled. The anadromous blueback herring and American shad,

resident killifish, and the estuarine bay anchovy were all found in relatively equal numbers, each between 9 and 11 percent of the total number sampled. Two resident minnows (eastern silvery minnow and spottail shiner), the resident channel catfish, and the estuarine hogchoker each composed 4 to 7 percent of the community. The remaining species were found in minor numbers, each composing less than 2 percent of the total number of fish sampled.

Table 1. Finfish species observed in the vicinity of the Philadelphia Naval Shipyard (information summarized from EA Engineering 1998).

COMMON NAME	SCIENTIFIC NAME	NUMBER	PERCENT COMPOSITION
Anadromous Fish			
Shortnose sturgeon	*Acipenser brevirostrum*	2	0.1
Atlantic sturgeon	*Acipenser oxyrhynchus*	1	0
American eel	*Anguilla rostrata*	16	0.5
Blueback herring	*Alosa aestivalis*	370	11.5
Alewife	*Alosa pseudoharengus*	42	1.3
American shad	*Alosa sapidissima*	295	9.2
Gizzard shad	*Dorosoma cepedianum*	9	0.3
White perch	*Morone americana*	887	27.6
Striped bass	*Morone saxatilis*	54	1.7
Resident Fish			
Common carp	*Cyprinus carpio*	10	0.4
Eastern silvery minnow	*Hybognathus regius*	143	4.5
Spottail shiner	*Notropis hudsonius*	155	4.8
White catfish	*Ameiurus catus*	2	0.1
Brown bullhead	*Ameiurus nebulosus*	2	0.1
Channel catfish	*Ictalurus punctatus*	188	5.9
Pumpkinseed	*Lepomis gibbosus*	9	0.3
Smallmouth bass	*Micropterus dolomieu*	1	0
Tiger muskellunge	*Esox masquinonogy x lucius*	1	0
Tessellated darter	*Etheostoma olmstedi*	48	1.5
Estuarine Fish			
Banded killifish	*Fundulus diaphanus*	273	8.5
Mummichog	*Fundulus heteroclitus*	75	2.3
Rough silverside	*Membras martinica*	5	0.2
Inland silverside	*Menidia beryllina*	40	1.2
Bay anchovy	*Anchoa mitchilli*	354	11
Atlantic croaker	*Micropogonias undulatus*	9	0.3
Naked goby	*Gobiosoma bosc*	2	0.1
Atlantic menhaden	*Brevoortia tyrannus*	4	0.1
Hogchoker	*Trinectes maculatus*	214	6.7

The data also indicated that the study area is used for rearing juvenile fish. Juveniles constituted slightly more than half (51 percent) of the species measured. All of the

blueback herring, American shad, and hogchoker observed in samples were juveniles. Both juvenile and adult white perch were observed in survey samples.

Studies indicate that the study area is used as a spawning ground for American shad and possibly striped bass. American shad spawn in a large reach of the river between Trenton and Philadelphia, and in recent years appear to be extending to areas downstream of Philadelphia and the study area. Spawning striped bass have not been observed, but their presence can be inferred by the presence of young-of-the-year fish near an island at the mouth of the Schuylkill River within the study area and downstream at Tinicum Island (RM 86) and Chester Island (RM 83).

Spawning in many of the fish species observed in the region is in response to thermal cues (Lee et al. 1980). Trigger temperatures vary, but generally spawning begins in spring and peaks in summer. Many species are serial spawners and may have multiple spawning periods during the season.

The federally endangered shortnose sturgeon and the Atlantic sturgeon, a candidate species for federal protection, are both present in the study area. Two shortnose sturgeon and one Atlantic sturgeon were collected in fish surveys in the study area.

Most of the species observed in the fish community near the study area are benthic feeders, including white perch, blueback herring, American shad, killifish, minnows, and hogchoker. Predatory fish observed in the area include adult striped bass and channel catfish.

2.2.3 Aquatic Bird Community

Relevant observations from the study area are provided to demonstrate the potential exposure of birds through aquatic food chains. Several groups of aquatic feeding birds have a seasonal presence at the study area. The dabbling ducks seasonally present include the mallard (*Anas platyrhynchus*), American black duck (*Anas rubripes*), and pintail (*Anas acuta*). The diving ducks seasonally present include the ruddy duck (*Oxyura jamaicensis*), canvasback (*Aythya valisineria*), and lesser scaup *(A. affinis*). Dabbling and diving ducks, which typically feed on aquatic vegetation and aquatic invertebrates, migrate through the study area in the fall and spring and sometimes overwinter.

Aquatic birds in the area that feed primarily on benthic organisms and fish include the bufflehead (*Bucephala albeola*), which feeds on benthic invertebrates, and the hooded merganser *(Mergus merganser*), which feeds primarily on fish. These species are fall and spring migrants, and sometimes overwinter.

2.3 POTENTIAL CONTAMINANTS OF CONCERN

Sediment chemistry data collected during the Navy's environmental studies of the Reserve Basin and the surrounding area were reviewed to identify COCs for evaluation in the screening-level risk assessment. The classes of contaminants that were initially considered included PAHs, polychlorinated biphenyls (PCBs), trace elements, pesticides, and dioxins and furans (EA Engineering 1996a).

Threshold Effects Levels (TELs) from the USEPA Assessment and Remediation of Contaminated Sediments (ARCS) program (USEPA 1996) were used as sediment benchmarks to identify COCs. These values were derived from freshwater exposures of *Hyalella azteca* using 28-day survival, growth, and reproductive endpoints. TELs were calculated as the geometric mean of the lower 15th percentile concentrations of the effects data and the 50th percentile of the no-effects data. The TELs were selected as protective COC screening criteria because they are considered to represent sediment concentrations rarely associated with adverse effects to benthic organisms. In addition, TELs derived for freshwater sediments (Smith et al. 1996) were used for comparison for the pesticides due to the fact that there were no ARCS TELs for these contaminants. A sediment benchmark associated with low risk to avian wildlife was used for dioxin (USEPA 1993).

Maximum measured sediment concentrations of contaminants of potential concern observed are compared to the corresponding TEL in Table 2. Several classes of contaminants were reported as not detected in Reserve Basin sediment samples, including PCBs, some pesticides, dioxins, and furans. The reported detection limits are used for screening comparison for contaminants that were reported as below detection in Reserve Basin sediment samples. This provides for the conservative assumption that these compounds are present at the detection limit, regardless of how high the detection limit may have been.

All of the maximum reported sediment concentrations of PAHs and trace elements exceeded the corresponding TELs, so these contaminants were retained as COCs. The detection limits reported for PCBs exceeded the corresponding TELs, so PCBs were retained as COCs. The maximum sediment concentrations of several pesticides were less than the corresponding TELs (chlordane, endrin, and DDT) and so were not retained. The remaining pesticides, with maximum sediment concentrations higher than corresponding TELs, were retained as COCs. These contaminants included dieldrin, heptachlor epoxide, gamma-BHC, DDD, and DDE. Finally, the detection limit reported for 2,3,7,8-TCDD was an order of magnitude below the proposed sediment benchmark. Therefore, dioxins and furans were not retained as COCs.

Table 2. Maximum sediment concentrations and screening guidelines.

	SEDIMENT CONCENTRATION (mg/kg)	
	MAXIMUM SEDIMENT CONC.	TEL
PAHs		
Benz(a)anthracene	4.99	0.032
Benzo(a)pyrene	3.26	0.032
Chrysene	6.61	0.057
Fluoranthene	9.41	0.11
Phenanthrene	11.7	0.042
Pyrene	9.46	0.053
LPAHs	35.8	0.076
HPAHs	32.7	0.193
Total PAHs	55.2	0.264
PCBs		
Aroclor 1016	<1.3	0.032[a]
Aroclor 1221	<1.3	0.032[a]
Aroclor 1232	<1.3	0.032[a]
Aroclor 1242	<1.3	0.032[a]
Aroclor 1248	<1.3	0.032[a]
Aroclor 1254	<2.6	0.032[a]
Aroclor 1260	<2.6	0.032[a]
Trace Elements		
Arsenic	36.5	5.9
Cadmium	8.93	0.60
Chromium	336	37.3
Copper	5,210	35.7
Lead	1,820	35
Mercury	4.03	0.17
Nickel	281	18
Zinc	16,000	123.1
Pesticides		
Chlordane	0.0033	0.005[b]
Dieldrin	0.012	0.0029[b]
Endrin	0.0019	0.0027[b]
Heptachlor epoxide	<0.130	0.0006[b]
Gamma-BHC	0.0034	0.0009[b]
DDD	0.240	0.00354[b]
DDE	0.270	0.0014[b]
DDT	0.0035	0.0070[b]
Dioxins and Furans	(µg/kg)	(µg/kg)
2,3,7,8- TCDD	<0.0074	0.021[c]

[a] TEL based on total PCB concentration.
[b] TEL value from Smith et al. 1996.
[c] Proposed sediment benchmark values associated with low risk to avian wildlife (USEPA 1993).

2.4 CONTAMINANT FATE AND TRANSPORT

The fate and transport of contaminants in the Reserve Basin will depend on a wide range of characteristics of the nearby Schuylkill River system and on the physical and chemical properties of the individual contaminants. In the following discussion, a brief overview of potentially important physical characteristics of the Reserve Basin is followed by a more detailed discussion of the fate and transport of the classes of contaminants that have been selected as COCs—trace elements, PAHs, PCBs, and pesticides.

2.4.1 Polynuclear Aromatic Hydrocarbons

PAHs are a class of nonpolar organic compounds characterized by highly aromatic, fused-ring structures. Environmental sources of PAHs include petroleum products and combustion residue (i.e., soot particles). Because of their low aqueous solubilities (0.0003 to 34 mg/L) and high octanol-water partition coefficients (log K_{ow} = 3.4 to 7.6), PAHs in aquatic systems tend to be associated with sediments and biota.

In the Schuylkill River and Reserve Basin, the fate and transport of PAHs will be largely controlled by sediment organic carbon content and dissolved organic carbon concentrations in the water column and sediment porewater. The extent to which an individual PAH compound will tend to be associated with either sediment or dissolved organic carbon depends on the relative hydrophobicity of the compound, which can be predicted from its molecular weight.

Low-molecular-weight PAHs (LPAHs), with three aromatic rings or less, are more water soluble and more easily degraded. High-molecular-weight PAHs (HPAHs) will tend to predominate in sediments where they are subjected to burial, resuspension, and degradation reactions. The available literature suggests that HPAHs are degraded by microbes slower than LPAHs. Half-lives for these compounds range from months to years. Furthermore, biodegradation probably occurs more slowly in aquatic systems than in soil (Clement Associates 1985).

PAH biodegradation rates and bacterial productivity have been measured in sediments collected from the Reserve Basin and the Delaware and Schuylkill Rivers (Boyd et al. 1999). this study suggests that active microbial communities in the Reserve Basin sediments are able to degrade PAH compounds.

Uptake of PAHs by aquatic biota is rapid. However, PAHs are also quickly metabolized and eliminated from most fish. Invertebrates, especially mollusks, do not metabolize PAHs as efficiently and may accumulate high tissue concentrations (Eisler 1987; Varanasi

et al. 1989). Bioconcentration factors for those species that do not metabolize PAHs tend to increase as the molecular weights and the octanol-water partition coefficients of the PAHs increase.

2.4.2 Trace Elements

Key factors that affect the speciation, and thus the bioavailability, of sediment-associated trace elements include Eh (redox conditions), pH, porewater hardness, sulfides, and the organic carbon content of the sediment. The redox conditions, pH, and the concentration of dissolved organic carbon in porewater influence the oxidation state and the dissolved concentration of the trace element. Trace elements exhibit a range of binding affinities, with both organic and inorganic phases present in the sediment, resulting in varying concentrations of dissolved versus particulate metals. In addition, trace elements exhibit a range of stability constants with dissolved ligands, which determines the ratio of complexed to freely dissolved species in solution.

Total concentrations of trace elements in sediment are generally not predictive of their bioavailability. Concentrations of certain metals in porewaters have been correlated with biological effects (DiToro et al. 1990). For several divalent metals, a key partitioning phase controlling cationic metal activity and toxicity in sediments appears to be acid-volatile sulfide (AVS) (DiToro et al. 1990, 1992; Carlson et al. 1991; Allen et al. 1993; Ankley et al. 1993). Simultaneously extracted metals (SEM) and AVS measurements can be made to assess the potential bioavailability of SEM—cadmium, copper, lead, nickel, and zinc. The SEM:AVS models may extend to silver as well, but the suitability and predictability for silver have yet to be fully resolved.

The bioavailability of trace elements that form stable complexes with organic compounds is particularly complex. For example, methylmercury compounds are extremely toxic and are efficiently bioaccumulated through aquatic food chains (Wiener and Spry 1996). Methylmercury is formed in aquatic sediments by microbial methylation of inorganic mercury.

2.4.3 Polychlorinated Biphenyls

The fate and transport of PCBs in environmental systems is controlled by distribution or partitioning of PCBs among sediment, suspended particulates, surface water, and biota. The observed partitioning of nonionic organic chemicals, such as PCBs, is due to sorption to organic matter. The extent to which PCB congeners are associated with organic matter relative to their dissolved aqueous concentrations is related to their levels of chlorination.

The more chlorinated congeners have stronger tendencies to be associated with particulate and dissolved organic matter than the less chlorinated congeners.

Bioaccumulation of PCBs occurs as a result of the partitioning of the congeners between an organic phase (the organism's lipid content) and aqueous solution. Therefore, bioaccumulation is highly dependent on the organism's lipid content and trophic level, and on the hydrophobicity of the PCB congener. In addition, PCBs are subject to biomagnification through the food chain.

Although PCBs are generally persistent, they can be degraded in situ to a limited extent by resident microorganisms.

2.4.4 Chlorinated Pesticides

Chlordane, dieldrin, endrin, and heptachlor epoxide are members of the cyclodiene pesticides, environmentally persistent and bioaccumulative chemicals that are stable in soil and resistant to photodegradation. As a result, they were used in greatest quantity as soil insecticides for the control of termites and soil-borne insects (Ware 1997).

Gamma-BHC is a moderately toxic compound that is highly persistent in the environment. It is very stable in both freshwater and marine environments, and is resistant to photodegradation (EXTOXNET 1996). It will disappear from water through secondary mechanisms such as adsorption to sediment, biological breakdown, and adsorption by aquatic animals through gills, skin, and food (EXTOXNET 1996).

DDT and its metabolites (DDD and DDE) are highly persistent in the environment, with reported half-lives of between 2 and 15 years in soil (ATSDR 1992). Because of its persistence in soil, DDT can reach surface waters through erosion and atmospheric transport. The reported half-life for DDT in lake water is 56 days, while that in river water is 28 days. Field and laboratory studies have demonstrated very little breakdown of DDT in estuarine sediments over the course of 46 days (EXTOXNET 1996)

2.5 ECOTOXICITY AND POTENTIAL RECEPTORS

2.5.1 Ecotoxicity of Contaminants of Concern

This section discusses ecotoxicity of the COCs selected for each potential ecological receptor group. The COCs were identified by screening the maximum measured concentrations against TELs that were chosen as benchmark values for ecotoxicity. For PAHs, endpoints are discussed for benthic invertebrates and fish. Relevant toxicological

endpoints of metals are presented for benthic invertebrates. Relevant toxicological endpoints of PCBs are presented for benthic invertebrates, fish, and birds.

2.5.1.1 Polynuclear Aromatic Hydrocarbons

PAHs vary substantially in their toxicity to aquatic organisms. LPAHs (2- and 3-ring compounds) such as naphthalene, fluorene, phenanthrene, and anthracene are acutely toxic to aquatic organisms. Acute toxicity increases with increasing alkyl substitution on the lower-molecular-weight compounds (Van Luik 1984). Many of the HPAHs, compounds with 4–7 aromatic rings, such as chrysene and benzo(a)pyrene, are less toxic but demonstrably carcinogenic, mutagenic, or teratogenic to a wide variety of organisms including fish, amphibians, birds, and mammals (Moore and Ramamoorthy 1984; Eisler 1987). Among aquatic organisms, acute toxicity is most pronounced among crustaceans and least pronounced among teleosts (Eisler 1987).

Benthic Invertebrates — Effects of PAHs observed in benthic invertebrates include inhibited reproduction, delayed emergence, sediment avoidance, and mortality (Eisler 1987; Landrum et al. 1991). In a study of PAH toxicity to the amphipod *Diporeia*, the mechanism identified as most likely responsible for observed acute toxic responses to PAHs was narcosis (Landrum et al. 1991). Generally, aquatic invertebrates are less able to metabolize PAHs than aquatic vertebrates, although metabolization rates vary widely within and among phyla (Meador et al. 1995).

Fish — PAHs are generally hydrophobic compounds and must be metabolized to more water-soluble forms before they are excreted. In most fish, PAHs are rapidly taken up, metabolized, and excreted so that concentrations found in edible tissues are generally low. The major route of elimination is through excretion into bile. The biotransformation and excretion rates can vary widely among fish species (Meador et al. 1995). Fish exposed to PAHs may be induced to produce higher levels of mixed-function oxidase (MFO) enzymes capable of transforming PAHs to more excretable, but occasionally more carcinogenic, metabolites (O'Connor and Huggett 1988). MFO enzymes are often used as an general, non-specific indicator of exposure to inducing compounds (e.g., PAHs, dioxins, furans).

Because fish rapidly metabolize and excrete PAHs, fish tissue residue concentrations of parent PAH compounds do not provide a useful measure of exposure to fish (Varanasi et al. 1989). Determining concentrations of PAHs in sediment is a useful measure of exposure because PAH-contaminated sediment has been linked to adverse effects in fish, including reproductive impairment, immune dysfunction, increased incidence of liver

lesions, and other histopathological endpoints (Malins et al. 1987; Johnson et al. 1988; Varanasi et al. 1992, Baumann et al. 1996). Fin erosion and liver abnormalities have also been observed in fish exposed to extracts from PAH-contaminated sediments (Fabacher et al. 1991). Other studies report sublethal effects on the cellular immune system (reduced macrophage activities) in fish exposed to PAH-contaminated sediments, that could result in increased susceptibility to disease (Weeks and Warinner 1984,1986; Weeks et al. 1986). The most common diseases generally affect the liver, although cataracts and pollution-related disorders of the skin and gills may also occur (O'Connor and Huggett 1988).

Birds – Very little data are available on the toxicity of PAHs in birds. PAHs are components of petroleum products such as oil and gas, and the toxicity of petroleum products in birds has been studied. However, toxicity studies that expose birds to petroleum products cannot be used without a direct measure of the PAH concentration associated with the exposure. In one study, Patton and Dieter (1980) fed mallards diets that contained 4,000 mg PAHs/kg for a period of 7 months. No mortality or visible signs of toxicity were evident during the exposure. However, the ducks' liver weights increased 25 percent and blood flow to the liver increased 30 percent when compared to controls (Eisler 1985). In addition, PAH mixtures applied to the surface of mallard eggs have been shown to increase embryo mortality and deformation (Hoffman and Gay 1981)

2.5.1.2 Metals

In fresh water, increasing water hardness decreases the toxicity of cadmium, chromium, copper, lead, nickel, silver, and zinc. The form of metal also affects toxicity; for example, methylmercury is more toxic than inorganic mercury. The combination of trace elements in the environment may result in additive, synergistic, or antagonistic effects, with the overall effect depending on the toxicity of the metals in question, the specific physical and chemical conditions of the site, and internal synergistic or antagonistic effects within organisms. In some species, certain trace elements are physiologically regulated to levels above ambient environmental concentrations (e.g., copper used in respiratory enzymes).

Benthic Invertebrates—Toxicity of trace elements to benthic organisms ranges widely, from slight reduction in growth rates to mortality. Mollusks are generally less sensitive than other aquatic phyla (Leland and Kuwabara 1985). The most sensitive life stages of benthic organisms are generally the embryonic and larval stages.

Fish – Fish are exposed to trace elements both in the water column and by consuming contaminated prey organisms. Freshwater fish are generally more sensitive to the effects

of trace elements than are marine species, and the larval stages are generally most sensitive. Commonly observed effects include reduced growth, survival, and fecundity.

Birds – Avian dietary toxicity studies have been conducted with a wide range of trace elements. The observed acute toxicity of the trace element can depend on the organism's body burden of metallothioneins. Ducks contained the highest levels of metallothioneins of a range of wildlife species surveyed (Brown et al. 1970 as cited in Eisler 1985). Sublethal effects can include reproductive and behavioral modifications. Teratogenic effects have been documented in chicken embryos after eggs were injected with chromium (Ridgeway and Karnofsky 1952, Gilani and Marano 1979 as cited in Eisler 1986). Similarly, the immersion of mallard eggs in solutions of methylmercury resulted in a significant incidence of skeletal embryonic malformations (Hoffman and Moore 1979 as cited in Eisler 1987).

2.5.1.3 Polychlorinated Biphenyls

Benthic Invertebrates — PCBs have a wide variety of effects on aquatic organisms. There are significant interspecies differences in sensitivities to PCBs, even among species that are closely related taxonomically (Eisler 1986). Most studies of the effects of PCBs on benthic invertebrates have shown effects on mortality, growth, and reproductive impairment.

Fish — Effects of PCBs on fish include mortality, growth-related effects, behavior responses, biochemical alterations, and adverse reproductive effects. Of particular concern are the effects of dioxin-like PCB congeners, which have the same toxic mechanism as 2,3,7,8-TCDD (Walker and Peterson 1991; Zabel et al. 1995). 2,3,7,8-TCDD and these dioxin-like PCB congeners cause early life stage mortality associated with blue-sac disease, which involves subcutaneous yolk sac edema (Wisk and Cooper 1990; Walker et al. 1991).

In addition, numerous field studies have reported increased mortality, pathologic anomalies, and biochemical changes in feral fish collected from ecosystems where PCBs have been reported and correlated with PCB tissue burdens (Niimi 1996). These observations include reduced hatchability and poor survival of larvae taken from feral organisms and reared in the laboratory (Ankley et al. 1991; Mac and Schwartz 1992). This impact is clearly important from an ecological perspective. Other impacts, such as behavioral responses and biochemical alterations, are more difficult to interpret, although some biochemical alterations may adversely affect reproduction (Sivarajah et al. 1978; Chen et al. 1986; Thomas 1988).

Birds — A substantial amount of research has documented adverse reproductive effects in piscivorous bird populations exposed to PCBs and dioxins (Tillitt et al. 1992; Jones et al. 1993, 1994; Giesy et al. 1994 a,b). The bulk of the research has focused on double-crested cormorants because deformities were first discovered in this species. Some work has done to evaluate reproductive effects of PCBs in the great blue heron (). Piscivorous birds display a number of symptoms similar to those observed in other avian species exposed to planar halogenated hydrocarbons in the laboratory, including altered biochemical homeostasis, physical deformities, fetotoxicity, and teratogenesis. In addition to embryo mortality, PCBs cause edema and beak malformations often recognized as crossed beaks in double-crested cormorants (Firestone 1973, Schrankel et al. 1982; Brunström and Darnerud 1983: all as cited in Brunström 1990).

2.5.1.4 Chlorinated Pesticides

Benthic Invertebrates — Relatively little information was found on the toxicity of dieldrin to benthic invertebrates. The 5-hour median lethal concentration (LC_{50}) for brown shrimp was reported to range between 25 and 500 µg/L.

Heptachlor is highly toxic to freshwater invertebrates. LC_{50}s for freshwater invertebrates range from 0.9 µg/L for a 96-hour exposure with the stonefly, *Pteronarcella badia*, to 80 µg/L for a 48-hour exposure in the cladoceran, *Simnocephalus serrulatus* (USEPA 1980).

The only information found relating sediment concentrations of gamma-BHC to toxicity to benthic invertebrates was the freshwater TEL of 0.94 µg/L (NOAA 1997). In the water column, gamma-BHC is highly toxic to aquatic invertebrates. Reported 96-hour LC_{50}s in aquatic invertebrates ranged from 4.5 µg/L in stoneflies to 460 µg/L in *Daphnia* sp. (EXTOXNET 1996).

The toxicity of DDT-contaminated sediments to aquatic organisms has not been extensively studied. Spiked sediment bioassays using the freshwater amphipod *Hyalella azteca* found that the 10-day LC_{50}s ranged from 11.0 to 49.7 mg/kg (TOC = 3.0 to 10.5 percent) (Nebeker 1988).

Fish — Dieldrin is a relatively potent toxin in fish. The 96-hour LC_{50} for bluegill is 7.9 µg/L, while that in goldfish is 37 µg/L. In one study, exposure to 50 µg/L for five hours resulted in 100 percent mortality in mullet (Environmental Health Data Search, no date).

Freshwater fish species are generally less sensitive to heptachlor than are invertebrate species. Ninety-six hour LC_{50}s for freshwater fish species ranged from 10.0 µg/L in

rainbow trout to 320 µg/L in goldfish (USEPA 1986). A 40-week chronic study was conducted using fathead minnows (*Pimephales promelas*) in which growth, reproduction, and survival were monitored. Concentrations of heptachlor tested were 1.84, 0.86, 0.43, 0.20, and 0.11 µg/L. All fish exposed to 1.84 µg/L were dead within 60 days. No adverse effects were reported in parental fish or their offspring at the other concentrations (USEPA 1980).

Gamma-BHC is also highly toxic to fish, with reported 96-hour LC_{50}s ranging from 1.7 to 90 µg/L in freshwater fish species.

Numerous acute toxicity studies demonstrate that DDT is acutely toxic to many aquatic organisms at low concentrations (Table 3). Additionally, the DDT metabolites DDD and DDE have also been shown to be acutely toxic to a number of fish species. Table 3 summarizes toxicity data for DDT, DDD, and DDE in a number of freshwater fish species.

In addition to its toxic effects, DDT bioaccumulates significantly in fish and other aquatic species, leading to long-term exposure. A half-time for elimination of DDT from rainbow trout was estimated to be 160 days (EXTOXNET 1996). The reported bioconcentration factor for DDT ranges between 1,000 and 1,000,000 in various aquatic species (EXTOXNET 1996). DDT and its metabolites biomagnify through the food web (ATSDR 1992).

Table 3. Acute toxicity data for DDT and its metabolites in fish

SPECIES	CONCENTRATION (µg/L)	EFFECT	REFERENCE
DDT			
Fish (24 species)	0.6 – 180	LC_{50}	USEPA 1980
Fish (19 species)	1.8 – 21.5	96-hour LC_{50}	Mayer & Ellersieck 1986
Fathead minnow	0.74	Chronic	USEPA 1980
DDE			
Fish (3 species	32-240	96-hour LC_{50}	Mayer & Ellersieck 1986
DDD			
Fish (6 species)	14 – 4,400	96-hour LC_{50}	Mayer & Ellersieck 1986

Birds — Birds are less sensitive to dieldrin than aquatic organisms. A medial lethal dose (LC_{50}) of 381 mg/kg body weight/day was reported in unspecified waterfowl (Environmental Health Data Search).

Heptachlor is moderately to highly toxic to birds. The reported acute oral LD_{50} in mallard ducks was 2,080 mg/kg, while the five-day dietary LC_{50} in Japanese quail was 99 mg/kg. Other reported 8-day dietary LC_{50}s for heptachlor were 450 to 700 mg/kg in bobwhite quail, and 250 to 275 mg/kg in pheasant (EXTOXNET 1996).

Gamma-BHC is slightly to moderately toxic to birds, with a reported LD50 of more than 2,000 mg/kg body wt./day in the mallard duck. The five-day dietary LC_{50} of gamma BHC in Japanese quail was 490 mg/kg, while an LC_{50} of 561 mg/kg has been reported in pheasant. Eggshell thinning and reduced egg production has also occurred in birds exposed to gamma-BHC (EXTOXNET 1996).

DDT may be slightly toxic to nearly non-toxic in birds. Reported dietary LD_{50}s ranged from greater than 2,240 mg/kg in mallard to 841 mg/kg in Japanese quail (EXTOXNET 1996). There has been much concern over chronic exposure of bird species to DDT and effects on reproduction, especially eggshell thinning and embryo mortality. The mechanisms of eggshell thinning are not fully understood, although it is believed that predatory species of birds may be more sensitive to these effects. Laboratory studies on avian reproduction have demonstrated the potential for DDT and DDE to cause subtle changes in courtship behavior, delays in pairing and egg laying, and decreases in egg weight in ring doves and Bengalese finches (EXTOXNET 1996).

2.5.2 Receptors of Concern

Biological studies indicated that a diverse aquatic community occupies several trophic levels in the study area near the Philadelphia Naval Shipyard. The selection of ROCs is based upon their potential presence in the Reserve Basin, their sensitivity to contamination, and their potential for exposure to contaminants based on the identification of primary exposure pathways. The availability of appropriate toxicity information, exposure factors, and consumption data are also important considerations in the selection of ROCs.

Individual species have been selected as potential ROCs for the fish and bird populations. These species were selected as surrogate representatives of specific feeding strategies and for the potential transfer of specific classes of contaminants through the food chain. These receptor organisms will be assessed during the screening-level exposure assessment.

The spottail shiner was selected as a representative forage fish species. Spottail shiners were one of the common forage species observed in the study area. This species is an opportunistic benthic feeder known to feed heavily on chironomids and other parts of the

benthic community in the study area. Spottail shiners and other minnows are also prey for piscivorous fish and birds.

The white perch was selected as a representative opportunistic benthic feeder. White perch, the most commonly observed species in fish surveys near the study area, are known to feed heavily on amphipods, oligochaetes, and isopods.

The channel catfish was selected as a representative aquatic piscivore. The species is a common resident predator in the study area, feeding primarily on minnows and other forage species.

The effects of contaminated sediments may be passed to birds by several trophic pathways, including from benthic organisms to benthic-feeding birds, and from forage fish to piscivorous birds. The bufflehead was selected as a representative benthic-feeding bird, and the hooded merganser was selected as a piscivorous bird. Both are common seasonal inhabitants of the study area. These selections were made with input from the US Fish and Wildlife Service.

2.6 EXPOSURE PATHWAYS

In this section, potential exposure pathways are evaluated to determine which pathways are complete and important at the site. Identifying complete exposure pathways prior to a quantitative evaluation allows the assessment to focus on only those contaminants that can reach ecological receptors (USEPA 1997). An exposure pathway is considered complete if a contaminant can travel from the source to ecological receptors and can be taken up by the receptors via one or more exposure routes (USEPA 1997). Often many pathways are complete, but are of varying importance. It is therefore important to identify the key pathways that reflect maximum exposures within the ecosystem and constitute exposure pathways to ecological receptors sensitive to the contaminant (USEPA 1997). As described in Section 4.1, pathways of PAHs, PCBs, pesticides, and trace elements to benthic invertebrates, fish, and aquatic birds are relevant at this site. The conceptual exposure pathways for ROCs in the Reserve Basin are illustrated in Figure 3.

For benthic invertebrates, direct contact with water or sediment by the gills or integument are the primary exposure pathways (USEPA 1997). For lower-trophic-level fish, diet and direct contact with water or sediment by the gills are the primary exposure pathways (USEPA 1997). For higher-trophic-level fish, diet can be an important exposure pathway for COCs that are bioaccumulated or biomagnify, such as certain PCB congeners and

methylmercury. The primary exposure pathway for avian species is their consumption of prey that have accumulated site-related contamination.

2.7 ASSESSMENT ENDPOINTS

As defined in USEPA (1992), assessment endpoints are explicit expressions of the actual environmental values that are to be protected, such as ecological resources. Assessment endpoints are generally tied to the response of ecological receptor species to environmental stresses. Unless an ecological receptor is listed as a protected or endangered species, assessment endpoints are selected that are relevant to population-level rather than individual effects.

Assessment endpoints for the Reserve Basin of the Philadelphia Naval Complex include:

- **Benthic Invertebrates:** Assess the potential for adverse effects on diversity, abundance, growth, and survival of the benthic community
- **Fish:** Assess the potential for reproductive impairment and other adverse effects in fish (e.g., impaired growth)
- **Birds:** Assess the potential for adverse growth or reproductive effects and survival in piscivorous and benthic-feeding birds

Measurement endpoints are measurable biological responses to the valued characteristics chosen as assessment endpoints; this definition is subject to change (USEPA 1997). The measurement endpoints corresponding to the assessment endpoints selected for this SLERA are presented below:

- **Benthic Invertebrates:** Compare concentrations of COCs in sediment to relevant screening concentrations
- **Fish:** Use BSAFs to estimate tissue concentrations, and compare to literature toxicity values
- **Birds:** Estimate concentrations of COCs in prey, estimate dose to ROCs, and compare with literature toxicity values

3.0
BENTHIC INVERTEBRATES EVALUATION

3.1 SCREENING LEVEL EXPOSURE ESTIMATE

Most of the sediment samples that have been collected in the Reserve Basin have been surface sediment samples (Dynamac Corp. 1992 and EA Engineering 1996b as presented in EA Engineering 1996a, Boyd et al. 1999). In addition, several studies have collected vibracore sediment samples and reported concentrations in composite samples that represent up to 7ft of sediment depth (Normandeau Associates 1991 and 1995 as presented in EA Engineering 1996a).

Maximum sediment contaminant concentrations were selected from the sediment data sets compiled in EA Engineering (1996a) and the Reserve Basin sediment data presented by Boyd et al. 1999 (Table 4). These are the data which are used for exposure point concentrations.

3.2 SCREENING LEVEL EFFECTS ASSESSMENT

Sediment benchmark concentrations were selected from the literature to represent effects to benthic invertebrates. The selected sediment benchmark concentrations are presented in Table 4.

Threshold Effects Levels from the USEPA ARCS program (USEPA 1996) were used as sediment benchmarks. TELs have been defined as the concentrations below which toxic effects are rarely observed (USEPA 1996). TELs were derived from freshwater exposures of *Hyalella azteca* using 28-day survival, growth, and reproductive endpoints (USEPA 1996). TELs were calculated as the geometric mean of the lower 15th percentile concentrations of the effects data and the 50th percentile of the no-effects data. In addition, TELs derived for freshwater sediments (Smith et al. 1996) were used to compare the pesticides because ARCS did not provide TELs for these contaminants. These TELs were calculated in the same manner as the ARCS values (USEPA 1996), but are based on a wider variety of bioassays and benthic community metrics.

Table 4. Sediment benchmarks, maximum concentrations, and calculated hazard quotients.

CONTAMINANT OF CONCERN	TEL[a] (mg/kg)	MAXIMUM SEDIMENT CONCENTRATION (mg/kg dw)	HQ (TEL)
PAHs			
Benz(a)anthracene	0.032	4.99	156
Benzo(a)pyrene	0.032	3.26	102
Chrysene	0.057	6.61	116
Fluoranthene	0.11	9.41	85.5
Phenanthrene	0.042	11.71	279
Pyrene	0.053	9.46	179
LPAHs	0.076	35.8	471
HPAHs	0.193	32.7	169
Total PAHs	0.264	55.2	209
Total PCBs	0.032	<2.6	<81
Trace Elements			
Arsenic	10.8	36.5	3.38
Cadmium	0.6	8.93	14.9
Chromium	36.3	336	9.26
Copper	28.0	5,210	186
Lead	34.2	1,818	53.2
Mercury	0.17	4.03	23.7
Nickel	19.5	281	14.4
Zinc	94.2	15,991	170
Pesticides			
Dieldrin	0.0029[b]	0.012	4.14
Heptachlor Epoxide	0.0006[b]	<0.130	<217
Lindane	0.0009[b]	0.0034	3.78
DDD	0.00354[b]	0.240	67.8
DDE	0.0014[b]	0.270	193

[a] ARC TELs (USEPA 1996).
[b] Smith et al. (1996).

3.3 SCREENING LEVEL RISK CALCULATION

For screening-level risk calculations, the exposure estimates and the screening ecotoxicity values were combined using the hazard quotient approach to estimate risk (USEPA 1997). Hazard quotients (HQs) are defined as the ratio of the estimated dose or environmental exposure point concentration at the site to a no observed effects level (NOAEL) for the contaminant. An HQ less than 1 indicates that the contaminant alone is unlikely to cause adverse ecological effects (USEPA 1997).

HQs were calculated from the ratios of the maximum sediment concentrations to the corresponding TEL, and are presented in Table 4. HQs calculated from maximum concentrations reported were greater than 1 for all COCs. TEL-HQs greater than 100 were calculated for the maximum sediment concentrations of PAHs, including benz(a)anthracene, benzo(a)pyrene, chrysene, phenanthrene, pyrene, LPAHs, HPAHs, and total PAHs; copper; zinc; and two pesticides, heptachlor epoxide (based on the detection limit) and DDE.

4.0
FISH EVALUATION

The exposure of fish in the Reserve Basin to mercury, PCBs, pesticides, and PAHs was evaluated using two approaches:

- The exposures to the bioaccumulative contaminants, PCBs, mercury, and pesticides, were assessed using a tissue residue effects approach. The concentrations of these compounds that accumulate in tissues are an integrative measure of all exposure pathways. These tissue concentrations were compared to those known to cause injuries or those that have been associated with observations of adverse impacts.

- For PAHs, tissue effects concentrations are not relevant, as fish rapidly metabolize and excrete PAH compounds (Varanasi et al. 1989). Risks to fish from exposure to PAHs were evaluated by contrasting concentrations of the PAHs in sediments to concentrations known to be injurious to fish or associated with observations of adverse effects.

4.1 SCREENING LEVEL EXPOSURE ESTIMATE

Tissue concentrations of contaminants in fish from the Reserve Basin have not been measured. Biota-sediment accumulation factors (BSAFs) from the literature were used to estimate potential fish tissue concentrations of PCBs and pesticides associated with exposure to sediments in the Reserve Basin. BSAFs are calculated by dividing the lipid-normalized concentrations of a chemical in an organism by the organic-carbon-normalized (OC-normalized) concentrations of the same chemical in sediment
(Equation 1):

$$\text{BSAF} = \frac{\text{lipid - normalized tissue concentration}}{\text{OC - normalized sediment concentration}} \qquad \textbf{Eq. 1}$$

An estimated tissue concentration can thus be calculated by multiplying the BSAF by the OC-normalized sediment concentration (Equation 2):

$$\text{Estimated lipid-normalized tissue concentration} = \text{BSAF} \times \text{OC-normalized sediment concentration} \qquad \textbf{Eq. 2}$$

BSAFs used to estimate tissue concentrations of PCBs and pesticides are presented in Table 5. BSAFs for PCBs and pesticides represent median BSAFs for benthic organisms, including benthic fish (spot, channel catfish, croaker white perch, fathead minnow, and scup), calculated by Tracey and Hansen (1996). An extensive analysis of differences in BSAFs for individual species and trophic levels concluded that the calculated BSAFs were similar for various species both within the same trophic level and between different trophic levels (Tracey and Hansen 1996). However, the use of a single BSAF for an entire class of contaminants, such as PCBs or pesticides, is a simplification of the complexity of the chemistry of the individual compounds.

Table 5. BSAF values for PCBs and pesticides

CONTAMINANT	BSAF	REFERENCE
PCBs	1.64	Tracey and Hansen (1996)
Pesticides	1.96	Tracey and Hansen (1996)

A biota-to-sediment ratio of 1 for mercury appears to be conservative based on the sediment and tissue concentrations presented by Suter et al. 1999 for largemouth bass, bluegill, gizzard shad, and channel catfish collected from a large river-reservoir system. The mercury concentrations in the sediments ranged 0.0029 to 139 mg/kg dry weight (Jones et al. 1999). This range of concentrations encompasses the range of mercury concentrations measured in the sediments of the Reserve Basin.

Tissue concentrations of PCBs, mercury, and pesticides were estimated, using these values, and are presented in Table 6. Tissue lipid concentrations were assumed to be 5 percent, based on the reported whole-body lipid content reported for the guppy (Henderson and Tocher 1987). Moisture content was assumed to be 80 percent for fish (Stephan et al. 1985). The sediment concentrations used in the calculation were the maxima from Table 5. Sample specific TOC values were used when available.

4.2 TISSUE CONCENTRATION SCREENING

There are no promulgated criteria for evaluating concentrations of contaminants in fish. However, concentrations associated with effects in toxicity tests and field studies were reviewed. Lowest observed effects levels (LOELs) were selected from these studies to screen the estimated tissue concentrations. Studies were identified from the U.S. Army Corps of Engineers (ACOE) ERED electronic database and from recent reviews of tissue residue effects data (ESI 1998; Monosson 1999; Suter et al. 1999).

Table 6. Maximum sediment concentrations and estimated fish tissue concentrations.

CONTAMINANT	SEDIMENT CONCENTRATIONS (mg/kg)		ESTIMATED FISH TISSUE CONCENTRATIONS (mg/kg)		
	DRY WEIGHT	OC-NORMALIZED	LIPID-NORMALIZED	DRY WEIGHT[a]	WET WEIGHT[c]
Total PCBs	<2.6	<17.3	<28.4	<1.41	< 0.282
Mercury	4.03	na	na		4.03
Dieldrin	0.012	0.24[b]	0.47	0.024	0.005
Heptaclor Epoxide	<0.130	<0.867	<1.70	<0.085	< 0.017
Lindane (gamma BHC)	0.0034	0.068[b]	0.13	0.0065	0.001
4,4'-DDD	0.240	0.41 / 4.1[d]	0.80 / 8.04[d]	0.04 / 0.402[d]	0.008 / 0.08[d]
4,4'-DDE	0.270	0.45/ 4.5[d]	0.882 / 8.82[d]	0.044 / 0.441[d]	0.009 / 0.088[d]

NOTE: na – not applicable

[a] Calculated assuming a tissue lipid content of 5 percent (Henderson and Tocher 1987).

[b] No organic carbon data available for this station: values were calculated using the average sediment organic carbon content of 5 percent (Boyd et al. 1999).

[c] Calculated assuming a tissue moisture content of 80 percent (Stephan et al. 1985).

[d] Organic carbon data for this station was reported as 60%: this extreme value is thought to be in error and that it maybe should have been only 6%. Values are calculated for both TOC values.

4.2.1 Polychlorinated Biphenyls

PCBs were marketed in the United States as mixtures of congeners known as Aroclors. Individual Aroclors contain different amounts of toxicologically important congeners. Although it can be helpful to know the identity of the individual Aroclors being summed to calculate a reported total PCB concentration, PCBs undergo diferential transport, degradation, and environmental fate. This occurs to such an extent that measurements of PCBs released in the environment rarely mimic the exact suite of congers which constituted the original, commercial Aroclor. There can also be considerable vairation in the analytical computations of PCBs as Aroclor mixtures versus a sum of the congeners observed. For the purposes of the screening-level risk evaluation, total PCB concentrations were screened. The type of PCB associated with each of the residue effects concentrations are identified in Table 7.

Selected LOELs for freshwater fish are presented in Table 7. The LOELs were selected as the lowest whole body concentrations associated with effects in freshwater species from two recent reviews (Suter et al. 1999, ESI 1998). The results are consistent with those of another review which estimated that whole body Aroclor 1254 concentrations of 5 µg/g wet weight or greater can result in reduced larval survival (Monosson 1999).

Lower tissue concentrations associated with reproductive effects were reported for ovaries from Baltic flounder (Von Westernhagen et al. 1981) and eggs from starry flounder, lake trout, and rainbow trout (Hogan and Brauhn 1975; Hendricks et al. 1981; Spies et al. 1985; and Mac and Edsall 1991). In addition, increased mortality was seen in yearling coho salmon associated with PCB concentrations of 0.5 to 1.2 µg/g wet weight in the liver (Folmar et al. 1982). However, in order to be consistent with the data set used to derive the BSAFs, only whole-body concentrations were selected as screening concentrations.

Table 7. Lowest observed effects levels of PCBs in fish

SPECIES	PCB TYPE	LOEL (µg/g wet wt)	EFFECTS ENDPOINT	REFERENCE
Lake trout	Aroclor 1254	0.7[a]	fry mortality	Berlin et al. 1981 as cited in Suter et al. 1999
Atlantic salmon	Mixture of Aroclors 1016, 1221,1254,1260	3.0	reduced growth in alevins	Fisher et al. 1994
Adult fathead minnow	Aroclor 1254	13.7	reduced fecundity and frequency of reproduction	ACOE 1988
Fingerling channel catfish	Aroclor 1242	14.33	reduced growth	Hansen et al. 1976

[a] Geometric mean of the LOEL and NOAEL.

4.2.2 Mercury

Almost all of the studies reviewed were laboratory studies that dosed fish with either methylmercury or mercuric chloride in food or water. Uptake of methylmercury by aquatic organisms is both more rapid and more extensive than uptake of inorganic mercury (Biesinger et al. 1982 and others). Furthermore, the uptake of organic vs. inorganic mercury may be different from a toxicological perspective. Following uptake, inorganic mercury can be converted in the fish gut to methylmercury (Rudd et al. 1980, as cited in Wiener and Spry 1996), or detoxified by binding to metallothioneins (Roesijadi 1992, as cited in Wiener and Spry 1996). In field-exposed adult freshwater fish, methylmercury is generally 95 to 99 percent of the total mercury concentration in the tissue (Wiener and Spry 1996). The speciation of mercury in the Reserve Basin sediment has not been determined.

LOELs for mercury in freshwater fish species are presented in Table 8. The selected concentrations represent the lowest whole-body mercury concentrations associated with

effects from an extensive literature review (ESI 1998). Lower effects concentrations were reported for the gonads of adult rainbow trout, associated with reduced larval survival in their offspring (Birge et al. 1979). For the purposes of this assessment, only whole-body tissue concentrations were selected as screening concentrations. The form of mercury to which fish were exposed in each study is noted in Table 8.

Table 8. Lowest observed effects concentrations of mercury in fish

SPECIES	MERCURY SPECIES	EFFECTS CONCENTRATION (μG/G WET WT)	EFFECTS ENDPOINT	REFERENCE
Larval fathead minnow	Mercuric chloride	1.24	Reduced growth (length)	Snarski and Olson 1982
Larval fathead minnow	Mercuric chloride	1.36	Reduced growth (weight)	Snarski and Olson 1982
Rainbow trout	Methylmercury	1.8	Mortality	Hawryshyn and Mackay 1979
Juvenile walleye	Methylmercury	2.37	Reduced growth	Friedmann et al. 1996

4.2.3 Pesticides

There were fewer LOELs available for pesticides than for PCBs and mercury. Whole-body tissue concentrations associated with effects in freshwater fish species obtained from the ACOE ERED database were reviewed and the lowest reported LOEL or NOAEL value was selected for each pesticide. The effects concentrations are presented in Table 9.

Table 9. Lowest observed effects concentrations of pesticides in fish

CONTAMINANT	SPECIES	EFFECTS CONCENTRATION (μg/g ww)	EFFECTS ENDPOINT	REFERENCE
Dieldrin	Guppy	10.7	Reduced growth	Burnett and Liss 1990
Heptaclor Epoxide	Spot	0.016[a]	Mortality	Schimmel et al. 1976
Lindane	Fathead minnow	0.537[a]	Mortality	Macek et al. 1976
4,4'-DDD	Fathead minnow	0.6	Reproduction	Jarvinen et al. 1977
4,4'-DDE	Lake trout	1.09	Mortality	Burdick et al. 1964

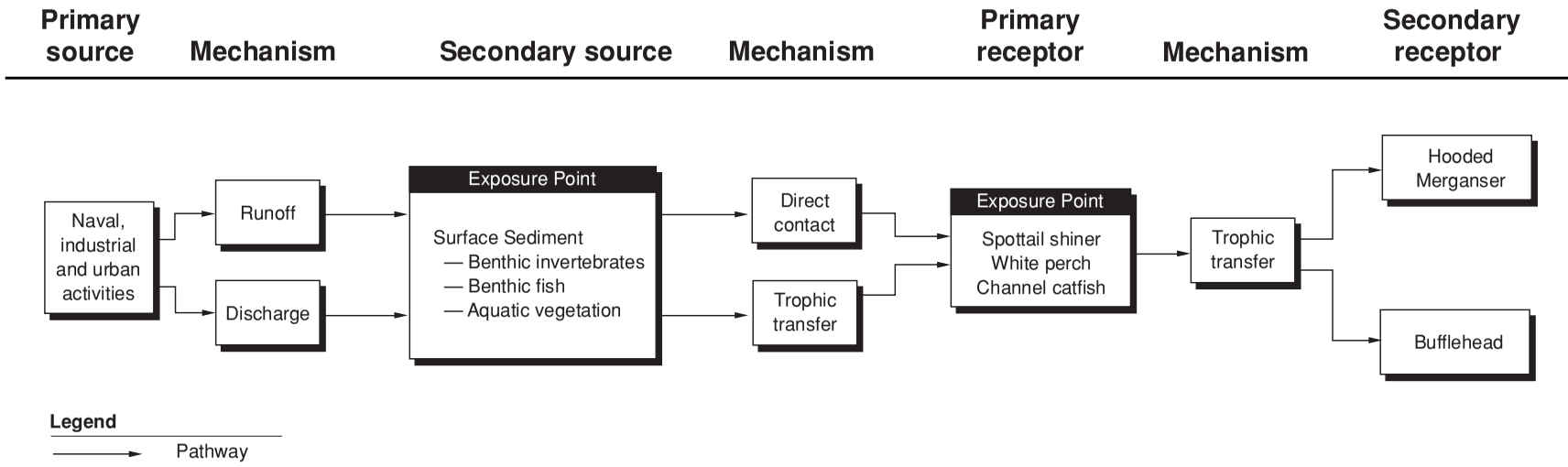

Figure 3. Potential exposure pathways for ecological receptors

4.3 SEDIMENT THRESHOLD SCREENING OF PAHS

Correlations between sediment PAH concentrations and neoplasms in feral fish, and the induction of neoplasms in fish by exposure to contaminated sediment extracts both support the hypothesis that some fish neoplasms and chronic responses result from exposure to PAHs in the fish's environment (Black et al. 1980; Baumann et al. 1982 1991; Baumann 1984; and Horness et al. 1998).

A sediment quality threshold of 2 μg/g, derived from the relationship observed between sediment PAH concentrations and the prevalence of hepatic lesions in English sole (Horness et al. 1998), was selected for evaluating the potential exposure of fish to the PAHs in the Reserve Basin sediment.

4.4 SCREENING-LEVEL RISK CALCULATIONS

4.4.1 PCB, Mercury, and Pesticide Screening

Risks to fish were estimated as HQs, calculated using the lowest of the LOELs for PCBs, mercury, and pesticides from Tables 7, 8, and 9, respectively. Calculated HQs exceeded 1 only for mercury and heptaclor epoxide (Table 10). Mercury was the only contaminant with a measured sediment concentration that resulted in an estimated tissue residue-based HQ greater than 1. Tissue concentrations of total PCBs and heptaclor epoxide were estimated based on sediment detection limits, not measured concentrations. The residues of dieldrin, lindane, DDD, and DDE estimated in fish were lower than the corresponding whole-body tissue concentrations of these contaminants associated with effects.

4.4.2 PAH Screening

The maximum total PAH concentration measured in the Reserve Basin sediments, 55.2 µg/g, was compared to the sediment quality threshold of 2 µg/g derived from the relationship observed between PAH concentrations in sediment and the prevalence of hepatic lesions in English sole (Horness et al. 1998). The resulting HQ was 27.6.

Table 10. Screening and estimated fish tissue concentrations (both in µG/G WET WT),and calculated hazard quotients

COMPOUND	SCREENING CONCENTRATION	ESTIMATED TISSUE CONCENTRATION	HQ
Total PCBs	0.7	< 0.282	<0.41
Mercury	1.24	4.03	**3.25**
Dieldrin	10.7	0.005	0.0005
Heptaclor Epoxide	0.016	< 0.017	**<1.06**
Lindane (gamma BHC)	0.537	0.001	0.002
4,4'-DDD	0.6	0.008 / 0.08 [a]	0.013 / 0.13
4,4'-DDE	1.09	0.009 / 0.088[a]	0.008 / 0.08

NOTE: **Bold** – HQ values greater than one

[a] Organic carbon data for this station was reported as 60%: this extreme value is thought to be in error and that it maybe should have been only 6%. Values have been calculated for both TOC values.

5.0
BIRD EVALUATION

5.1 SCREENING LEVEL EXPOSURE ESTIMATE

Exposure of avian receptors to COCs associated with Reserve Basin sediments was estimated based on the assumption that contaminant exposure was entirely through diet. Aquatic invertebrates are the primary constituent of buffleheads' diet although they also consume some aquatic vegetation, seeds, and fish (Gough et al. 1998). In this SLERA, 100 percent of bufflehead diet was assumed to consist of benthic invertebrates. Mergansers were assumed to derive 100 percent of their diet from fish within the Reserve Basin, although aquatic invertebrates are also included in their diet (Gough et al. 1998). Exposure to site-related contaminants through water consumption and the incidental ingestion of sediments from the Reserve Basin were not evaluated.

Food ingestion rates for the merganser and bufflehead were estimated using the procedures outlined in US EPA's *Wildlife Exposure Factors Handbook* (1993). The rate of food consumption that an animal must achieve to meet its metabolic needs can be calculated by dividing its free-living (or field) metabolic rate (FMR) by the metabolizable energy in its food. Metabolizable energy (ME) is the gross energy (GE) in a unit of food consumed minus the energy lost in urine and feces. Assimilation efficiency (AE) equals the ratio ME/GE, or the fraction of GE that is metabolizable. AE is relatively constant among different groups of consumer species of mammals and birds that are carnivorous, insectivorous, herbivorous, or granivorous (USEPA 1993).

The generic equation for estimating oral contaminants doses in food for wildlife species is:

$$ADD_{pot} = \sum_{k=1}^{m} (C_k \times FR_k \times NIR_k) \qquad \textbf{Eq. 3}$$

where: ADD_{pot} = potential average daily dose (e.g., mg/kg/day)

C_k = average contaminant concentration in food type k (e.g., mg/kg wet weight)

Fr_k = fraction of intake of food type k that is contaminated (unitless). For example, if k in an animal's diet were salmon, FR_k for salmon would equal the fraction of the salmon consumed that is contaminated at level C_k.

NIR_k = Normalized ingestion rate of food type k on a wet-weight basis (e.g., in g/g-day)

m = number of contaminated food types

Estimation of contaminant dose for avian receptors at the PNC is based only on consumption of a single type of prey item for each of the species considered, and it is also being assumed that 100 percent of the prey items are contaminated, so that FR_k is equal to 1. Therefore, the only item left to calculate is the normalized ingestion rate.

The normalized ingestion rate can be calculated as follows:

$$NIR = FMR / (BW \times ME_{avg}) \qquad \textbf{Eq. 4}$$

where:

NIR = normalized ingestion rate (g/g BW/day)
FMR = estimated field metabolic rate (kcal/day)
BW = body weight in grams
ME_{avg} = estimated average metabolizable energy of diet (kcal/g wet wt)

FMRs have been calculated for a number of animal species, including birds. The FMR for non-passerine birds is:

$$FMR\ (kcal/day) = 1.146\ (BW)^{0.749} \qquad \textbf{Eq. 5}$$

The body weights for the bufflehead and merganser were derived from Dunning (1993) in which mean body weights were reported for the male and female of each species. For this risk assessment, body weights were calculated by taking the average of the mean male and female body weights (Table 11).

Table 11. Mean weights of bufflehead and merganser

GENDER	WEIGHT IN GRAMS	
	BUFFLEHEAD	MERGANSER
Male	473	1,709
Female	334	1,232
Average	404	1,470

SOURCE: Dunning (1993)

The gross energy, assimilation efficiency, and metabolizable energy for each of the food items used by the bufflehead and the merganser are presented in Table 12. The estimated normalized ingestion rates for bufflehead and merganser are presented in Table 13.

Table 12. Gross energy, assimilation efficiency, and metabolizable energy values used in calculating normalized ingestion rates for bufflehead and merganser

Dietary Item	Proportion of Diet		Gross Energy (kcal/g wet wt)	Assimilation Efficiency	Metabolizable Energy (kcal/g wet wt)
	BH	M			
Fish	0	1	1.2	0.79[a]	0.95
Aquatic invertebrates	1	0	1.1[b]	0.77	0.84

SOURCE: USEPA (1993)
NOTE: BH - bufflehead
M - merganser

[a] Based on the assimilation efficiency of fish in eagles and seabirds
[b] Based on the gross energy content of amphipods, shrimp, and isopods

Table 13. Estimated normalized ingestion rates for bufflehead and merganser.

Species	Normalized Ingestion Rate (g/g BW/day)
Bufflehead	0.30
Merganser	0.19

The estimated dose for each COC was calculated by multiplying the estimated normalized ingestion rate by the concentration of each COC in the appropriate prey item. Bufflehead were assumed to consume benthic invertebrates containing contaminant concentrations identical to those in the sediments (i.e.,.BSAF = 1), so doses were calculated for each COC. In addition to the review for mercury (Jones et al. 1999), a review of almost 500 records of BSAFs for organics and invertebrates from the US Army Corps of Engineers database (1999) suggests a BSAF of 1 is appropriate assumption. A similar review for metals and invertebrates also indicates BSAF values are typically less than 1 for many metals (PTI 1995).

Contaminant exposure in the merganser was assumed to occur entirely through fish consumption. COC concentrations in fish within the Reserve Basin were estimated as described in Section 4.2. A BSAF of 1 was used for mercury and also assumed for other metals. The estimated doses of the COC in the bufflehead and the merganser are presented in Table 14.

Table 14. Estimated dose to bufflehead and merganser

Contaminant of Concern	Bufflehead[a] Concentration in Prey[c] (mg/kg)	Bufflehead[a] Dose (mg/kg BW/day)	Merganser[b] Concentration in Prey[c] (mg/kg)	Merganser[b] Dose (mg/kg BW/day)
PCBs (Total)	<2.6	<0.79	< 0.282	< 0.054
Trace Elements				
Arsenic	36.5	11	36.5	6.9
Cadmium	8.93	2.7	8.93	1.7
Chromium	336	101	336	63.8
Copper	5,210	1,575	5,210	990
Lead	1,818	550	1,818	345.4
Mercury	4.03	1.2	4.03	0.77
Nickel	281	85	281	53.4
Zinc	15,991	4,837	15,991	3,038
Pesticides				
Dieldrin	0.012	0.003	0.005	0.0001
Heptachlor epoxide	<0.130	<0.039	<0.017	< 0.003
Gamma-BHC	0.0034	0.001	0.001	0.0002
4,4'-DDD	0.240	0.07	0.008 / 0.08	0.0002 / 0.015
4,4'-DDE	0.270	0.08	0.009 / 0.088	0.0002 / 0.017

[a] The concentrations of organic contaminants and trace elements in benthic invertebrates (primary prey of the bufflehead) of the Reserve Basin were assumed to be identical to their maximum concentrations in sediments of the Reserve Basin.

[b] The concentrations of PCBs and pesticides in fish (primary prey of the merganser) within the Reserve Basin were estimated as described in Section 2.2.1. The concentrations of trace elements in fish tissues were assumed to be identical to their maximum concentrations in the sediments. Estimated concentrations of PAHs in fish tissues were not calculated due to the rapid metabolism of PAHs by fish.

[c] The toxicity reference value for avian receptors is the NOAEL expressed as mg chemical/kg body weight/day as reported in Sample et al. (1996).

5.2 Effects Assessment

The estimated wildlife NOAELs for avian species reported in Sample et al. (1996) for each COC evaluated were derived from chronic studies using bird species other than the bufflehead and the merganser (Table 15). All of the studies were of sufficient duration to include a sensitive life-stage of the test species. Toxicity endpoints evaluated in the studies included growth, reproduction, behavior, and/or mortality.

Table 15. Summary information on derivation of TRVs for the contaminants of concern as reported in Sample et al. (1996)

COC	Test Species	Endpoint	Duration	Test Species NOAEL (mg/kg BW/day)
Total PCBs	Ring-necked pheasant	Reproduction	17 weeks	0.180
Trace Elements				
Arsenic	Mallard	Mortality	128 days	5.1
Cadmium	Mallard	Reproduction	90 days	1.45
Chromium	Black duck	Reproduction	10 months	1.00
Copper	Chicken	Growth and mortality	10 weeks	47
Lead	American kestrel	Reproduction	7 months	3.85
Mercury	Mallard	Reproduction	3 generations (>1 year)	0.006
Nickel	Mallard	Growth, behavior, and mortality	90 day	77.4
Zinc	Chicken	Reproduction	44 weeks	14.5
Pesticides				
Dieldrin	Barn owl	Reproduction	2 years	0.077
Gamma-BHC	Mallard	Reproduction	8 weeks	2.00
4,4'-DDD	Brown pelican	Reproduction	5 years	0.003 (as DDT)
4,4'-DDE	Brown pelican	Reproduction	5 years	0.003 (as DDT)

5.3 Screening Level Risk Calculation

Hazard quotients for avian receptors were calculated by dividing the calculated contaminant dose by the TRV for that contaminant (Table 16). The TRVs for the avian receptors were estimated wildlife NOAELs reported in Sample et al. (1996). HQ values of 1 or less are interpreted as an exposure resulting in negligible risk of adverse effects in receptor species. HQ values greater than 1 indicate risk, although they must be viewed in the context of the degree of protectiveness and uncertainty built into the exposure calculations and effects studies.

5.3.1 Bufflehead

The primary exposure route for bufflehead was assumed to be their consumption of benthic invertebrate species within the Reserve Basin. The conservative assumption was made that the concentrations of COCs in benthic invertebrates were identical to the maximum sediment concentrations. HQs were greater than 1 for all of the COCs evaluated , except for dieldrin and gamma-BHC (Table 16). The HQs of the trace elements chromium, lead, mercury and zinc exceeded 100, while the HQs of arsenic, cadmium, copper, nickel, total PCBs, 4,4'-DDD, and 4,4'-DDE were less than 30. The estimated dose of PCBs was based on a detection limit while all others were based on measured concentrations.

Table 16. Toxicity reference values, estimated daily doses, and hazard quotients for avian receptors at the Philadelphia Naval Complex

CONTAMINANT OF CONCERN	BUFFLEHEAD			MERGANSER		
	TRV[a]	DOSE[b]	HQ	TRV[a]	DOSE[b]	HQ
PCBs (total)	0.180[a]	<0.79	**<4.4**	0.180[a]	< 0.054	<0.3
Trace Elements						
Arsenic	5.1	11	**2.2**	5.1	6.9	**1.4**
Cadmium	1.45	2.7	**1.9**	1.45	1.7	**1.2**
Chromium	1.00	101	**101**	1.00	63.8	**63.8**
Copper	47	1,575	**33**	47	990	**21.1**
Lead	3.85	550	**143**	3.85	345.4	**89.7**
Mercury	0.006	1.2	**203**	0.006	0.77	**128**
Nickel	77.4	85	**1.1**	77.4	53.4	0.7
Zinc	14.5	4,837	**333**	14.5	3,038	**209.5**
Pesticides (μg/kg)						
Dieldrin	0.077	0.003	0.05	0.077	0.0001	0.001
Heptachlor epoxide	na	<0.039	nc	na	< 0.003	nc
Gamma-BHC	2.00	0.001	0.0005	2.00	0.0002	0.0001
4,4'-DDD [c]	0.003	0.07	**24**	0.003	0.0002 / 0.015	0.06 / **5**
4,4'-DDE [c]	0.003	0.08	**27**	0.003	0.0002 / 0.017	0.06 / **5.7**

NOTE: **Bold** – HQ values greater than one
na – not available
nc – not calculated

[a] The TRV for avian receptors is the NOAEL expressed as mg chemical/kg body weight/day as reported in Sample et al. (1996).
[b] Dose expressed in mg/kg body weight/day.
[c] Organic carbon data for this station was reported as 60%: this extreme value is thought to be in error and that it maybe should have been only 6%. Values have been calculated for both TOC values.

5.3.2 Merganser

Merganser's consumption of fish was assumed to be their primary route of exposure in the Reserve Basin. Mercury exceeded benchmarks for bioaccumulative CoCs. DDT metabolites (4,4'-DDD and 4,4'-DDE) may have also exceeded these benchmarks. The mercury HQ was the greatest at 128 (Table 16). There is uncertainty associated with the HQs for the DDT compounds related to the unusual TOC values for those samples. HQs greater than 50 were calculated for chromium, lead, and zinc. These latter

compounds are not generally considered to be prone to biomagnification through the food chain. Therefore, the assumption that the fish tissue concentrations would be equal to the sediment concentrations of these contaminants may be conservative. The estimated dose of total PCBs was based on a detection limit. All of the other doses were based on food web modeling using measured concentrations.

6.0
UNCERTAINTY ANALYSIS

6.1 BENTHIC INVERTEBRATES ANALYSIS

Using detection limits as the maximum sediment concentrations for total PCBs and the pesticide heptachlor epoxide is one source of uncertainty in evaluating exposure of benthic invertebrates to contaminants associated with the Reserve Basin sediments. Using these values to screen will provide a conservative estimate (i.e., an overestimate) of the potential for exposure to these contaminants.

In addition, the use of the Reserve Basin as a moorage area and the potential erosion of spent blasting grit from the Girard Point Blasting Grit Disposal Area suggest that tributyltin (TBT) may be present in the Reserve Basin sediments. This compound has been used in marine antifouling paints for decades. Exposure to TBT has been shown to result in both chronic and acute effects in marine invertebrates (Langston and Burt 1991; Gibbs and Bryan 1996; Meador 1997; and others). Sediments in the Reserve Basin have not been analyzed for TBT. Therefore, the potential for TBT exposure cannot be evaluated at this time.

6.2 FISH ANALYSIS

The lack of surface water data makes it difficult to estimate the potential exposure of fish within the Reserve Basin. The unconsolidated nature of the Reserve Basin sediments, combined with turbulence in the water column caused by the movement of ships, could result in substantial resuspension events that would release sediment contaminants into the water column.

Using BSAFs to estimate fish tissue concentrations also introduces uncertainty into the evaluation. The BSAFs that were used were average values derived from an extremely large data set. The use of an average value for entire classes of compounds such as PCBs and pesticides simplifies the complex chemistry of the individual compounds within the contaminant class. For the purposes of this assessment, these BSAFs provide a reasonable estimate of concentrations in fish resulting from exposure to sediment contaminants. However, site-specific conditions could influence the uptake of contaminants by fish in the Reserve Basin. Default assumptions for lipid content and sediment organic carbon were also used in the tissue modeling. These assumptions may add uncertainty to the predicted tissue residues.

Determining tissue residue effects concentrations in fish using the limited information available in the published literature is complicated by a number of factors, including:

- Differences in species sensitivity;
- Differences in the tissues analyzed;
- Differences in exposure scenarios in laboratory and field studies;
- Lack of consistency in the endpoints evaluated; and
- Differences in the species of the contaminant (for example, inorganic mercury vs. methylmercury)

Finally, there is some uncertainty associated with the exposure assessment used to screen sediment PAH concentrations. Although PAHs are known to cause adverse effects in fish, methodologies are still being developed to quantify this risk. Some researchers have used PAH concentrations in sediment associated with elevated rates of neoplasia in fish as a screening value, but this approach is subject to confounding factors such as the presence of co-occurring contaminants in the field. In addition, the database that was used for this purpose contained a comparatively small number of data points. The ecological significance of neoplasms in fish populations has yet to be firmly established, and no further information is available for determining sediment effects concentrations for other, possibly more sensitive endpoints such as reproductive impairment and immune dysfunction.

6.3 AVIAN RECEPTOR ANALYSIS

Uncertainty associated with the exposure of avian receptors relates primarily to the calculation of the dose for each contaminant. No tissue data were available for prey species in the Reserve Basin, so conservative assumptions were used to estimate the contaminant concentrations in fish and benthic invertebrates in the basin. The factors that lead to uncertainty in estimating fish tissue residues above would also apply to estimations of dose to birds. Also, the maximum measured sediment concentrations were used to estimate prey tissue concentrations rather than attempting to integrate the exposure throughout the area as a bird would as it foraged.

The use of generic equations to estimate the normalized ingestion rate, the field metabolic rate, and the estimated average metabolizable energy of birds' diets provides reasonable estimates of these values for the calculation of normalized ingestion rates for the receptor

species in the absence of species-specific data. However, there is some uncertainty with regard to the resulting estimated species-specific ingestion rates. Variability in parameters used to model ingestion rates (e.g., body weight, assimilation efficiency) was not quantified, and so each introduces a source of uncertainty to the overall result.

Assumptions of a diet composed to entirely invertebrates or fish were used to simplify the modeling process. This assumption, and a lack of information about the actual diet of avian receptors in the vicinity, are additional sources of uncertainty. Incidental consumption of water and sediment, a factor for some duck species, was not considered. This factor would tend to underestimate exposures if one of these species was assessed.

The dose model assumed that the assimilation efficiency of contaminants was complete. This is a conservative assumption. There assumption is not widespread information on the assimilation efficiency of contaminants to birds, but available data suggests that it does not reach 100%. The actual assimilation efficiency of contaminants is another source of uncertainty to the estimation of actual dose.

Uncertainties due to interspecies differences in toxicity of the COCs were not addressed in the development of the TRV values or in the risk calculation. Toxicity data for ducks, chicken, kestrel, pheasant, barn owl, and pelicans were used to derive TRV values for great blue heron (Sample et al. 1996). These TRV values were then used to calculate risks to the bufflehead and the merganser.

Finally, the potential for exposure of bufflehead to PAHs accumulated by benthic invertebrates could not be evaluated due to the lack of TRVs for PAHs in birds. Extensive reviews of several databases failed to produce any toxicity data for birds resulting from a dietary exposure to PAHs. Without this toxicity data it is impossible to evaluate the potential exposure of bufflehead in the Reserve Basin.

7.0
ADDITIONAL EVALUATIONS

Screening level risk assessments are simplified assessments that can be conducted with limited data, using conservative assumptions for the parameters for which site-specific data are lacking (USEPA 1997). Conservative approaches are used to minimize the potential of failing to identify a potential risk. This approach ensures that sites with the potential to pose risks to ecological receptors are further evaluated in a baseline risk assessment. The decision at the end of a SLERA is either that a site passes the screening or the site fails and moves on to a second tier of assessment or possibly directly to an interim action (CNO 1999). The first step of a Tier 2 Baseline Ecological Risk Assessment should be refining the conservative exposure assumptions to move the assessment closer to one that uses site-specific values instead of default values.

Additional evaluations were conducted for benthic invertebrates, fish, and birds using assumptions that are more realistic and less conservative than those used in the evaluations described in Sections 3.0, 4.0, and 5.0. Examples of these refinements include , using mean sediment contaminant concentrations instead of maximum concentrations. Such refinements constitute Step 3a of the Tier 2 Baseline Ecological Risk Assessment as promulgated in the Navy Policy for Conducting Ecological Risk Assessments (CNO 1999). Preliminary Step 3a refinements are presented in the following sections.

7.1 BENTHIC INVERTEBRATES

For the Screening Level Ecological Risk Assessment, maximum sediment contaminant concentrations were selected from the sediment data sets compiled in EA Engineering (1996a) and from the Reserve Basin sediment data presented by Boyd et al. 1999 (Table 17), according to USEPA Risk Assessment Guidance (USEPA 1997). A maximum value does not represent the distribution of exposure levels to which populations of invertebrates within the basin would be subjected to. If a single parameter is to be used to characterize exposure, the general exposure level for an entire population may be better characterized by use of a mean value. Therefore, the mean sediment PAH and trace element concentrations calculated for the 28 stations sampled in 1997 by Boyd et al. (1999) are presented in Table 17. All of the PAH compounds and trace elements, except for mercury, were detected at all 28 stations. The mean mercury concentration was calculated using the sediment mercury concentrations measured within the Reserve Basin in four sediment studies (EA Engineering 1996a). Sediment mercury concentrations were

greater than the detection limits at 25 of the 30 stations sampled. Mean sediment concentrations were not calculated for PCBs because they were not reported as detected in any Reserve Basin sediment samples. Pesticides were detected at relatively few stations. Therefore, only maximum concentrations were reported for pesticides.

The SLERA also used the Threshold Effect Level as a toxicological benchmark. TELs are efficient for predicting samples that have a low probability of imparting adverse biological effects. However, exceedance of a single TEL is not necessarily an efficient predictor of toxicity (Long et al. 1998). Therefore, PELs were also selected as sediment benchmarks. PELs were derived by calculating the geometric mean of the 50th percentile of the effect data set and the 85th percentile of the no-effect data set. PELs were intended to estimate the sediment concentration of a chemical above which adverse biological effects frequently occurred (Smith et al. 1996). Analysis of the predictive performance of sediment quality benchmarks has indicated that as the degree of exceedance rises and as the number of individual compounds exceeding their respective benchmark rises, the incidence of observing toxicity also rises, while the incidence of observing a non-toxic sample decreases. Long et al. (1998) indicate that for marine PELs, the probability of observing a highly toxic sediment sample approaches 90% when eleven or more PELs are exceeded.

Most of the PELs for freshwater sediments presented in Table 17 were obtained from Smith et al. (1996). The PELs reported for LPAH, HPAH, and total PAH concentrations were taken from the ARCS data for the 28-day exposure of *Hyalella azteca* to freshwater sediments (USEPA 1996).

HQs were then calculated based on the PELs along with average sediment concentrations. PEL-HQs were greater than one for the maximum and mean concentrations of all contaminants except the mean concentrations of benzo(a)pyrene and cadmium (Table 17). The larger HQs (those greater than 30) were calculated for the maximum reported concentrations of LPAHs, zinc, heptaclor epoxide, and DDE.

Table 17. Sediment benchmarks, concentrations, and calculated HQs

CONTAMINANT OF CONCERN	TEL[a] (mg/kg)	PEL[b] (mg/kg)	Sediment concentration (mg/kg)[c]		HQ (TEL)	HQ (PEL)
PAHs						
Benz(a)anthracene	0.032	0.385	Mean:	1.29	**40.3**	**3.35**
Benzo(a)pyrene	0.032	0.782	Mean:	0.72	**22.5**	0.92
Chrysene	0.057	0.862	Mean:	1.63	**28.6**	**1.89**
Fluoranthene	0.11	2.35	Mean:	2.96	**26.9**	**1.26**
Phenanthrene	0.042	0.515	Mean:	5.18	**123**	**10.1**
Pyrene	0.053	0.875	Mean:	3.14	**59.2**	**3.59**
LPAHs	0.076	1.17[a]	Mean:	13.27	**175**	**11.3**
HPAHs	0.193	2.34[a]	Mean:	9.68	**50.2**	**4.14**
Total PAHs	0.264	3.37[a]	Mean:	22.94	**86.9**	**6.81**
Total PCBs	0.032	0.277	<2.6		**<81**	**<9.39**
Trace Elements						
Arsenic	10.8	17	Mean:	20.9	**1.94**	**1.23**
Cadmium	0.6	3.53	Mean:	3.5	**5.83**	0.99
Chromium	36.3	90	Mean:	130	**3.58**	**1.44**
Copper	28.0	197	Mean:	391	**14.0**	**1.98**
Lead	34.2	91.3	Mean:	343	**10.0**	**3.76**
Mercury	0.17	0.486	Mean:	0.932[d]	**5.48**	**1.92**
Nickel	19.5	35.9	Mean:	68.4	**3.51**	**1.91**
Zinc	94.2	315	Mean:	1,420	**15.1**	**4.51**
Pesticides						
Dieldrin	0.0029[b]	0.0067	Max:	0.012	**4.14**	**1.79**
Heptachlor epoxide	0.0006[b]	0.00274	Max:	<0.130	**<217**	**<47.4**
Lindane	0.0009[b]	0.00138	Max:	0.0034	**3.78**	**2.46**
DDD	0.00354[b]	0.00851	Max:	0.240	**67.8**	**28.2**
DDE	0.0014[b]	0.00675	Max:	0.270	**193**	**40.0**

NOTE: **Bold** – HQ values greater than one

[a] ARC TELs and PELs (USEPA 1996).
[b] Smith et al. (1996).
[c] Mean calculated from sediment concentrations presented in Boyd et al. (1999).
[d] Mean calculated from sediment concentrations presented in EA Engineering (1996a).

7.2 FISH

Fish exposure to PAHs in the Reserve Basin for the SLERA was evaluated by comparing the maximum measured sediment PAH concentration to sediment benchmark concentrations. The resulting HQ was 27.6. For fish whose home range is the entire basin, the maximum sediment concentration would be a biased estimate for their exposure. When mean values are used to characterize overall exposure, the resulting HQ for mercury drops to 0.75. If the mean sediment PAH concentration is compared to the sediment benchmark, the resulting HQ is 11.5.

The benchmark for sediment PAHs of 2 µg/g, was derived from relationships observed between sediment PAH concentrations and the prevalence of various hepatic lesions in English sole (Horness et al. 1998). There may be even more sensitive species than the English sole (e.g., winter flounder) as well (Collier 1999). If any cancerous or precancerous lesion prevalence is considered, the threshold drops to 0.62 µg/g. Therefore, the 2 µg/g can be thought of as a robust benchmark.

7.3 BIRDS

Bird exposure to contaminants associated with the Reserve Basin sediments was estimated based on a food web model. The maximum sediment contaminant concentrations were used to estimate concentrations in prey that comprise a dietary exposure to two avian species, the bufflehead and the merganser (Section 5.0). If birds fed exclusively within the basin, their exposure would not be most accurately characterized by an exposure route based solely upon the maximum sediment concentration. Even if their feeding range was limited to the basin, they would be feeding upon organisms that themselves would represent a range of bioaccumulation from an entire distribution of sediment contamination levels within the basin. If a single parameter is to be used to characterize exposure for the avian population, a mean value may be a better representation of the degree of contamination of prey items throughout the basin than the maximum. Therefore, the mean trace element concentrations were also used to further estimate exposure for both species (Table 18).

The HQs calculated using the mean sediment concentrations were greater than 1 for both species for all trace elements except cadmium and nickel, and for arsenic for the merganser. Given the degree of exceedance and the conservative assumptions used (e.g., feeding range, assimilation), it is not likely that arsenic and copper are risk factors.

Table 18. Bird TRVs, sediment concentrations, doses, and hazard quotients

Trace Elements	TRV[a]	Sediment Concentration (mg/kg)		Bufflehead		Merganser	
				Dose[b]	HQ	Dose[b]	HQ
Arsenic	5.10	Mean:	20.90	6.32	**1.2**	3.97	0.78
Cadmium	1.45	Mean:	3.50	1.1	0.7	0.67	0.46
Chromium	1.00	Mean:	130.00	39.3	**39**	24.70	**24.70**
Copper	47.00	Mean:	391.00	118.3	**2.5**	74.29	**1.58**
Lead	3.85	Mean:	343.00	103.7	**27**	65.17	**16.93**
Mercury	0.01	Mean:	0.93	0.28	**47**	0.18	**29.51**
Nickel	77.40	Mean:	68.40	20.7	0.3	13.00	0.17
Zinc	14.50	Mean:	1,420.00	429.5	**30**	269.80	**18.61**

NOTE: **Bold** – HQ values greater than one

[a] The TRV for avian receptors is the NOAEL expressed as mg chemical/kg body weight/day as reported in Sample et al. (1996).

[b] Dose expressed in mg/kg body weight/day.

8.0
SUMMARY

8.1 BENTHIC INVERTEBRATES ASSESSMENT

Potential effects in benthic invertebrates were assessed by comparing the concentrations of COCs in sediment to their corresponding screening benchmarks. As a result, HQs were calculated as the ratio of the sediment concentration to the corresponding screening benchmark. TEL-HQ values greater than 100 were reported for the maximum sediment concentrations of all the PAHs except fluoranthene. The mean sediment PAH concentrations resulted in TEL-HQs greater than 20 for all individual PAHs. For the trace elements, the maximum sediment concentrations resulted in TEL-HQs greater than 10 for all COCs except arsenic and chromium. The TEL-HQs calculated with the mean trace element concentrations were all greater than 1. In addition, TEL-HQs greater than 1 were calculated for total PCBs and pesticides. However, the total PCB concentration was based on the highest reported detection limit, as was the heptachlor epoxide concentration. Since these values were censored by the detection limit, the resulting HQs only indicate that risk due to these COCs is unknown. Overall, these results indicate that we cannot eliminate the potential for risk to benthic invertebrates due to contaminant exposure. Therefore, the site fails the screening assessment.

When Step 3a refinements were made to the assessment and concentrations in sediment were compared to PELs, the results were similar. PEL-HQ values exceeded 1 for not only the maximum concentrations, but even the mean concentrations, of PAHs and all trace metals in sediment, except the mean concentration of cadmium. PEL-HQs based on mean values were greater than 1 for pesticide concentrations as well.

Given the degree of exceedance and the number of COCs exceeding the PELs, these results indicate a high probability that exposure to COC concentrations in Reserve Basin sediments will cause harm to benthic invertebrates. These results also suggest that the adverse effects expected would be widespread throughout the basin, and are not likely isolated to particular “hotspots.”

8.2 FISH ASSESSMENT

Fish exposure in the Reserve Basin to mercury, PCBs, pesticides, and PAHs was evaluated using two approaches. The exposures to mercury, PCBs, and pesticides were evaluated by comparing estimated tissue concentrations to tissue concentrations associated with adverse impacts in freshwater fish. PAHs exposure was evaluated by comparing the measured sediment PAH concentrations to sediment concentrations associated with adverse effects.

Mercury and PAHs were the only contaminants for which measured maximum sediment concentrations resulted in HQ values greater than 1. The mercury HQ was 3.25; the maximum sediment PAH concentration resulted in a HQ of 27.6. Therefore the site fails the screening assessment according to this endpoint as well.

An HQ just slightly greater than 1 was also calculated for the pesticide heptaclor epoxide, which was reported as not detected in the Reserve Basin sediments. The exposure assessment was based on the reported detection limits. Therefore, the potential for risk due to exposure to this contaminant cannot be excluded, but has likely been overestimated.

Refinements to the assessment using the mean sediment PAH concentration resulted in an HQ of 11.5. These results for the assessment of risk to fish reinforce the conclusion that fish within the Reserve Basin are likely exposed to PAH concentrations that would result in adverse effects.

8.3 BIRD ASSESSMENT

Birds' food consumption was assumed to put them at risk from COCs in Reserve Basin sediments. The estimated doses of COCs derived from food-web modeling and sediment concentrations to two species, the bufflehead and the merganser, were compared to avian TRVs (Sample et al. 1996).

The resulting HQs for trace elements were greater than 1 for all trace elements in both species, except nickel in the merganser. For the bufflehead, the HQs for chromium, lead, mercury, and zinc were greater than 100. For the merganser, the HQs for these elements were greater than 50. The maximum HQs calculated for DDD and DDE for both species were also greater than 1. The HQs for bufflehead were 24 and 27, respectively, and the maximum merganser HQs were 5 and 5.7 for both compounds. Questionable TOC values associated with those sediment concentrations makes the assessment suspect. Finally,

HQs calculated for total PCBs exceeded 1 for the bufflehead, with an HQ of 4.33. PCBs were reported as below detection throughout the basin though. This HQ value is based on the detection limit.

This assessment suggests a potential for trace elements to adversely affect birds feeding on benthic invertebrates and fish within the Reserve Basin. In addition pesticides and PCBs may also affect these birds. Mercury, total PCBs, 4,4'-DDD, and 4,4'-DDE are of particular concern because of their tendency to biomagnify through the food web. Thus, the site fails the screening process for the assessment of risk to birds.

It should be noted that extremely conservative assumptions were made to assess the exposure of the merganser. None of the trace elements, with the exception of mercury, would be expected to be biomagnified through the food chain. When Step 3a refinements were made to the screening process, the overall conclusions did not change since HQs still exceeded 1 for trace metals.

9.0 RECOMMENDATIONS

The results of the Screening Level Risk Assessment fail to meet the exit requirements for stopping the risk assessment process. Hazard quotient modeling indicates that the site does may pose significant risk to benthic invertebrates, fish, and aquatic birds. The next tier of assessment, which includes refinement of assumptions made in the screening assessments and re-evaluation, also fails to substantiate a conclusion that the site cannot pose any ecological risk, and therefore fails to meet exit criteria for the risk assessment process. According to the decision framework and phased process for ecological risk assessment (CNO 1999), scoping for a baseline risk assessment is the next step.

This section provides recommendations regarding the types of data that could be collected to support a baseline ERA, and also to aid in the selection of remedial alternatives. The recommendations for the baseline ERA are organized by receptor.

9.1 BENTHIC INVERTEBRATES

The results of the screening-level assessment and Step 3a refinement indicate a high probability of adverse bioeffects in benthic invertebrates exposured to sediment concentrations of PAHs, trace elements, and pesticides in the Reserve Basin. Due to elevated detection limit problems, however, these assessments failed to provide conclusive PCB findings.

To support a baseline ERA, additional sediment samples could be collected and analyzed for the suite of contaminants of concern. One goal of this sampling would be to achieve lower detection limits for PCBs and several pesticides that have not been detected in the Reserve Basin sediments. In addition, sediment toxicity could be tested, with Toxicity Identification Evaluations (TIE) conducted on the samples found to be toxic. This study would provide information both about the toxicity of the sediments, and about the contaminants that are driving the observed toxicity. In addition, the results of the sediment toxicity testing and TIE would be useful in deriving site-specific sediment remediation goals.

Additional sediment samples could also be evaluated for the presence of benthic infauna, either qualitatively or quantitatively. The level of effort, particularly for organism identification (and associated costs) could be tailored to the degree of information desired. This information, along with the additional sediment chemistry, would help confirm

conclusions of impacts to the benthic community. Alternatively, sediment profile imaging would allow fast, efficient surveying of benthic conditions. A sediment profile imaging survey could confirm conclusions of impacts to benthos, but would not provide any association between observations and contaminant exposure as the responsible stressor.

9.2 FISH

Because no fish tissue data were available, the screening-level risk evaluation of potential effects in fish due to exposure to Reserve Basin sediment was based on estimated fish tissue concentrations. The estimated tissue concentrations were calculated using the maximum reported sediment concentrations and BSAFs taken from the literature. There is no field verification of these modeled exposure levels however.

For a baseline ERA, fish could be collected within the Reserve Basin and their tissues analyzed for a selected suite of contaminants of concern to generate the actual exposure level of bioaccumulative COCs. Measured fish tissue concentrations would enable a more accurate assessment of potential effects in the resident fish population, and would provide data useful for estimating the food-web exposure of piscivorous birds.

For risk to fish associated with exposure to PAHs, alternative approaches could be used to confirm conclusions of the SLERA and Step 3a re-evaluation. For instance, the level of PAH metabolites in fish could be used to confirm their degree of exposure. The stomach contents of fish could also be analyzed to determine the actual dietary exposure level of PAHs. The presence of adducts of PAHs with DNA in blood could be measured as one measurement endpoint of impacts. Additionally, the degree of enzymatic induction, which is not only a measure of exposure but also a measure of potential for disruption of reproductive systems, could also be evaluated as another measurement endpoint.

9.3 BIRDS

In the screening-level assessment it was assumed that the entire diet of both the bufflehead and the merganser had contaminant concentrations equal to or greater than the maximum sediment concentration. This assumption results in an extremely conservative estimate of the exposure.

To support a baseline ERA, contaminant concentrations in benthic invertebrates and fish could be measured to provide a more realistic estimate of the dietary exposure of piscivorous and benthic-feeding birds within the Reserve Basin. In addition, incorporating information regarding the home ranges and dietary preferences of the receptor species in the ERA would result in a more accurate assessment of the exposure of the avian species.

10.0
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APPENDIX

COMMENTS

From Don Killian, Pennsylvania Department of Environmental Protection:

I have reviewed the Draft Screening-Level Ecological Risk Assessment (SLERA) of Reserve Basin Sediments for the Philadelphia Naval Complex (PNC) dated May 1999. According to the SLERA, it has been conducted in accordance with EPAs *Ecological risk assessment guidance for Superfund: Process for designing and conducting ecological risk assessments, interim final* (1997) and Navy Policy for Conducting Ecological Risk Assessments (1999). I do not have a copy of the EPA guidance (on order through Internet site); however, the SLERA does seem to follow Steps 1 and 2 (Fundamental Components and Preliminary Exposure Estimate and Risk Assessment) for site-specific ecological risk procedures as described in the Ecological Evaluation Training Workbook for the May 11, 1999 workshop conducted in Harrisburg by Ogden Environmental. These are the first steps in the EPA guidance.
Overall, the SLERA seems to be done well and contains very little "fluff." The general conclusion of the SLERA is that it is inadequate to determine that no substantial ecological risk exists from the PNC Reserve Basin sediments. Recommendations for further action are provided at the end of the SLERA.
I have a few specific comments relating to the SLERA, which follow:

1. Although it is mentioned in the SLERA that one endangered fish species (shortnose sturgeon) and one fish species (Atlantic sturgeon) candidate for federal protection exist in the Delaware River (in excess of 1000 meters from the Reserve Basin), no fish sampling has been done to determine the presence of either species in the Reserve Basin itself or within a 2500-foot radius of the Reserve Basin. Also, no discussion is provided concerning the potential effect that the contaminated sediment in the Reserve Basin may have on these species as they exist in the Delaware River. Endangered and candidate species should be addressed in the next steps of the site-specific ecological risk assessment. Also, in determining if an ecological impact has occurred or will occur, [as required by Section 250.402(d)(1) of the Act 2 regulations], such determination should be made to include the requirements of Section 250.311(d), i.e., species of concern and habitats of concern (as defined in Section 250.1) should be considered as well as presence or absence of threatened or endangered species and exceptional value wetlands within the 2500-foot radius of the Reserve Basin.

2. Although piscivorous fish have been mentioned in the SLERA, (i.e., channel catfish) estimated contaminant doses to piscivorous fish (from consumption of benthic feeding fish) have not been included in the dose estimates for piscivorous fish. Also, the effects of metals on fish (except for mercury) have not been included in the SLERA.
3. All of the estimated doses for bufflehead (table 14 of the SLERA) appear to be incorrectly calculated. The reason for this seems to be a parenthesis error in the NIR estimation equation .... This results in the wrong units for NIR. Also, the estimated doses for dieldrin and lindane for merganser appear to be incorrect. ... The SLERA states that (except for mercury and tin) metals are not bioaccumulated in fish implying that they are not very important with respect to toxicity to fish and therefore birds. Although metals may not generally be bioaccumulated, they could still be toxic to fish and birds irrespective of their ability to move through the food chain. There is considerable evidence that lead and other metals are toxic to fish and birds.
4. Tributyltin (TBT), which is very toxic to aquatic species, has not been included in the sampling and analysis or in the SLERA.
5. The general recommendation of the SLERA is that a more refined baseline ecological risk assessment needs to be conducted to acquire a better indication of existing and potential risks and to aid in selection of remedial alternatives. This refined baseline ecological risk assessment describes a variety of actions which could be undertaken to further define ecological conditions in the Reserve Basin. Most of these actions include additional sampling and analysis of sediments in the Reserve Basin (to include toxicity testing), sampling and analysis of fish and benthic invertebrates in the Basin, and more refined information regarding home ranges and dietary preferences of receptor bird species for a more accurate estimate of bird exposure to contaminants.
   - Although not stated as a recommendation, the assessment should include a survey of the number of individuals in each fish and bird species in the Reserve Basin relative to those in the survey of the Delaware River so that the density and diversity requirements of Section 250.311(d) of the Act 2 regulations can be determined.
   - As already stated, species and habitats of concern (as defined in the Act 2 regulations) should be included in this assessment as well as presence of threatened or endangered species and exceptional value wetlands.
   - Tributyltin should also be included in the sampling and analysis and dose estimation of the assessment.
- Fish sampling should include both benthic feeding and piscivorous fish species so that contaminant dose estimates for birds will be more accurate.

General Comment: The general conclusion of the SLERA is that it is inadequate to determine that no substantial ecological risk exists from the PNC Reserve Basin sediments.

Response: Perhaps a more appropriate summarization would be that the SLERA concludes that existing information cannot unequivocally determine that there is a non-existent or acceptably-low level of risk.

Comment #1: No fish sampling has been done to determine the presence of endangered and candidate fish species in the Reserve Basin itself or within a 2500-foot radius of the Reserve Basin. Also, no discussion is provided concerning the potential effect that the contaminated sediment in the Reserve Basin may have on these species as they exist in the Delaware River. Endangered and candidate species should be addressed in the next steps of the site-specific ecological risk assessment. Also, in determining if an ecological impact has occurred or will occur, [as required by Section 250.402(d)(1) of the Act 2 regulations], such determination should be made to include the requirements of Section 250.311(d), i.e., species of concern and habitats of concern (as defined in Section 250.1) should be considered as well as presence or absence of threatened or endangered species and exceptional value wetlands within the 2500-foot radius of the Reserve Basin.

Response: This comment is noted. The Baseline Ecological Risk Assessment (BERA) will be designed to address these species, either directly or via surrogate species. Modeling efforts discussed in the SLERA suggest that predicted currents within the basin are not strong enough to scour previously deposited sediments from the Reserve Basin and then transport them to the Schuylkill River. Therefore the expected impact of basin sediments to the river is predicted to be negligible. Impact to the river is expected to be limited to export of suspended solids.

Comment #2: Although piscivorous fish have been mentioned in the SLERA (i.e., channel catfish), estimated contaminant doses to piscivorous fish (from consumption of benthic feeding fish) have not been included in the dose estimates for piscivorous fish. Also, the effects of metals on fish (except for mercury) have not been included in the SLERA.

Response: As mentioned in the SLERA, an extensive analysis of BSAFs for individual species and trophic levels concluded that the calculated BSAFs were similar for various species both within the same trophic level and between different trophic levels. The SLERA also employed a conservative approach for this assessment. Therefore, the assessment of bioaccumulative CoCs in the SLERA should adequately address piscivorous fish species as well as benthic species.

We agree that not all exposure pathways for all CoCs have been included for fish. Those for which data existed to support a technically-valid assessment and which were deemed to represent significant risk factors were included. As noted, aqueous CoC exposure data was desired, but such data do not exist. There is a common presumption that protecting the benthic habitat necessary to support a healthy, diverse fish community does provide some degree of protection to the fish themselves from contaminant exposure. Therefore, this assessment indirectly addressed this issue. Also, inclusion of this additional risk factor would not have changed the overall conclusions that risk to the fish community must be further evaluated.

We agree that a BERA should address issues related to risk to the fish community further.

Comment #3: Estimated doses for bufflehead appear to be incorrectly calculated. Also, the estimated doses for dieldrin and lindane for merganser appear to be incorrect. The SLERA states that (except for mercury and tin) metals are not bioaccumulated in fish implying that they are not very important with respect to toxicity to fish and therefore birds. Although metals may not generally be bioaccumulated, they could still be toxic to fish and birds irrespective of their ability to move through the food chain. There is considerable evidence that lead and other metals are toxic to fish and birds.

Response: Doses have been corrected and the HQs adjusted appropriately. The impact for bufflehead is an across-the-board 40% inflation in dose and HQs. As the reviewer has noted, the overall conclusions of the SLERA do not change.

There was no statement in the SLERA to the effect that since most metals are not generally considered to be prone to biomagnification through the food chain, they were non-toxic. The statement was made relative to the assumption that the fish tissue concentrations would be equal to the sediment concentrations of these contaminants, and therefore by assuming metals were completely bioaccumulated (i.e., bioaccumulation

factor of 1) the assessment should be an extremely conservative one. References will be added to support the statement regarding biomagnification of metals.

Comment #4: Tributyltin (TBT) has not been included in the sampling and analysis or in the SLERA.

Response: TBT was listed as a potential CoC and was retained as a potential CoC in the draft SLERA due to its possible use at the facility. The potential for TBT exposure was not evaluated however since no data were available at time the SLERA was drafted. Very recent results for TBT analyses have become available. This data is from a very limited sampling (only two sampling stations, one in duplicate) conducted on April 28, 1999 near the marine railway (adjacent to the basin entrance channel). TBT concentrations were found to be 79.5 to 89.2 µg/Kg with corresponding TOC levels of 16.1% and 19% in these samples. An approach for a screening level for sediment TBT was developed by a EPA Region IX workgroup. This approach uses aqueous toxicity information contrasted with a predicted pore water concentration (calculated using a K_{oc} value). Using this approach with an aqueous toxicity value of 0.02 µg/L, which corresponds to a 5th percentile of chronic toxicity values (to match the 95th percentile protection level of AWQC), corresponding TBT benchmarks at the TOC levels observed in these samples were calculated to be 80.8 and 95.1 µg TBT/Kg dw. HQ's using these benchmarks are just under 1 (0.98 and 0.94 respectively). The TOC levels observed in these samples are quite elevated however, and they are also outside the scope of TOC levels used in studies which developed the partition coefficients. A TBT benchmark at 2% TOC- an average value for coastal sediment- or at 10% - commonly applied maximal TOC concentration- would be 10 and 50 µg/Kg respectively. HQ's using either of these values would exceed 1.

Given the uncertainty of the above analysis, and the very limited sampling effort upon which it is based (only two stations), firm conclusions regarding risk associated with TBT exposure within the entire basin cannot be determined. Therefore, the recommendation of the SLERA that TBT should be carried forward as a CoC remains unchanged.

Comment #5: The BERA should include a survey of the number of individuals in each fish and bird species in the Reserve Basin relative to those in the survey of the Delaware River so that the density and diversity requirements of Section 250.311(d) of the Act 2 regulations can be determined.

- As already stated, species and habitats of concern (as defined in the Act 2 regulations) should be included in this assessment as well as presence of threatened or endangered species and exceptional value wetlands.
- Tributyltin should also be included in the sampling and analysis and dose estimation of the assessment.
- Fish sampling should include both benthic feeding and piscivorous fish species so that contaminant dose estimates for birds will be more accurate.

Response: Comment noted. The issues will be addressed in the planning phase for the BERA.