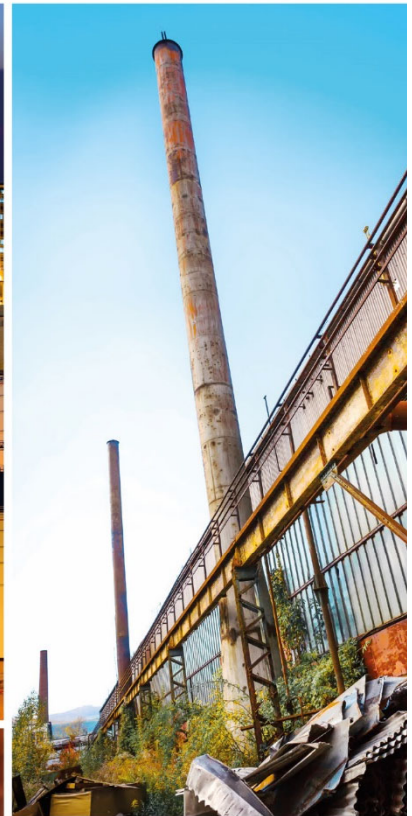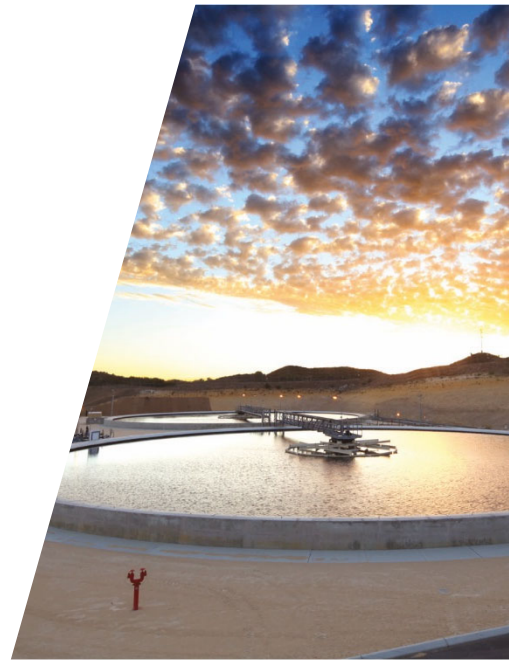# Ecological Risk Assessment

Sauer Dump Superfund Site
EPA ID: MDD981038334
Dundalk, Maryland

The Sauer Dump Superfund Site
RI/FS Performing Parties Group

Revised September 2019June 2020

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1. Introduction

On behalf of the Sauer Dump Superfund Site Remedial Investigation/Feasibility Study (RI/FS) Performing Parties Group, GHD Services, Inc. (GHD) hereby submits to the United States Environmental Protection Agency (USEPA) this Ecological Risk Assessment (ERA) for the Sauer Dump Superfund Site (Site). This report is being submitted in accordance with the requirements of the Administrative Settlement Agreement and Order on Consent (ASAOC), Docket No. CERC 03 2012 0177DC, effective March 11, 2013 (USEPA ID: MDD981038334). This Report was prepared to meet specific requirements included in Section 6.0 of the RI/FS Work Plan approved by USEPA on March 13, 2014 and USEPA guidance for Superfund Sites (USEPA, 1997, 1998).

This Report presents Steps 1 and 2 of the ERA.

Step 1 – Screening Level Problem Formulation and Ecological Effects Evaluation includes the problem formulation, preliminary ecological conceptual site model (CSM) and exposure pathways. Step 1 of the ERA was previously prepared by GHD (formerly CRA) as a Memo dated February 2, 2015 and submitted to USEPA. USEPA provided comments on the Step 1 Memo dated April 29, 2015. GHD submitted responses to USEPA's comments in a letter dated May 28, 2015. Step 1, including consideration of USEPA's comments, is presented in Sections 2 and 3 of this report.

Step 2 – Screening Level Exposure Estimate and Risk Calculation includes estimating exposure levels and screening for ecological risks. Step 2 is presented in Sections 4 through 6 of this report.

The ERA has been revised to address the comments provided by U.S.EPA and Maryland Department of the Environment (MDE) and HydroGeoLogic, Inc. (HGL) on June 19, 2019. U.S. EPA had conditionally approved the ERA in their letter dated May 11, 2020; however, they requested the results of the February 2020 chromium investigation, as summarized in the April 14, 2020 GHD letter (GHD, 2020), be incorporated into the final ERA.

2. Background (Step 1)

The Site is located adjacent to Lynhurst Road in Dundalk, Baltimore County, Maryland. The Back River, a tidal estuary of the Chesapeake Bay, borders the Site to the south. Figure 2.1 presents a Site Location and Vicinity Map.

The Site is composed of several properties including all of Parcel 425 and parts of Parcels 574, 503, 464, 295, and 137, and any areas where Site-related hazardous substances have come to be located as a result of Site operations" (ASAOC, 2013). Based on ENSAT's 2002 review of aerial photographs, the former operations area (FOA) was determined to be an approximate 3 acre portion of the Site where Mr. Frederick Sauer conducted salvage operations (Malcolm Pirnie, 2008). The entire area is made land and as a result there is a potential for the presence of other constituents related to the filling and not related to site activities. The FOA boundary is shown as an orange line on Figure 2.2. The majority of the FOA is located on Parcel 425 (approximately 2.5-acres) and overlaps onto residential parcels 137, 295, 425, 464, 503, and 574. The FOA delineation was determined based on aerial photography interpretation rather than subsurface explorations, and is only representative of surficial expressions of salvage operations, and may or may not define the

extent of contamination, either lesser of greater. Therefore, the FOA boundary is considered approximate. The FOA was originally a topographically-low tidal inundation area that was filled in with soil and construction material, scrap metal, and other debris. The majority of the FOA is covered with approximately 10 feet of fill principally consisting of soil and construction material, scrap metal, and other debris. Drainage ways containing wetlands extend along the east and west sides of the FOA and discharge to a cove of the Back River. These drainage ways have historically been referenced, respectively, as the Southeast Finger and Southwest Finger. A non-tidal freshwater pond ("the Pond") is also present on the northwest margin of the FOA. The groundwater flow direction is from north to south across the Site with discharge to the Back River.

The majority of the FOA is located on Parcel 425 (approximately 2.5-acres), which is currently owned by the Wittstadt Hunting Club. However, the Site has not been and is not currently used for hunting or fishing activities. Access by the general public to Parcel 425 is precluded by a fence and isolation from public rights-of-way by intervening private property. A fence also occurs along the southern boundary of Parcel 425, precluding access from the Back River. A portion of the FOA extends onto the bordering residential properties to the east (portions of Parcels 503, 464, 295, and 137). Historic sample results indicate that some portions of residential Parcels 137, 209, 295, 464, 503 (beyond the FOA on those parcels) may have been impacted by the same COCs as found on the Site. Some of these affected parcels contain structures that are used for residential and commercial purposes. The residential lots are served by public water.

The surrounding land uses consist of:

- Northeast, east and southeast – residential properties

- South – a cove that is part of the Back River, which is a tributary of the Chesapeake Bay

- Northwest and west – small wetlands, pond, and forest

3. Screening-Level Problem Formulation and Ecological Effects Evaluation (Step 1)

3.1 Environmental Setting and Contaminants at the Site

3.1.1 Environmental Setting

Figure 2.1 presents a Site Location and Vicinity Map. This figure depicts the surrounding regional topography and surface water hydrology based on the United States Geological Survey (USGS) 7.5-minute-series Middle River Quadrangle Map. The elevation of the Site ranges from approximately zero feet mean sea level to nearly 17 feet above mean sea level (AMSL). As noted above, the majority of the FOA is bounded by a fence, which precludes access by the public. A mounded area, present in the western and central portions of the Site (Figure 2.2), exhibits the irregular topography typically associated with sites subject to dumping activity. The topography along the eastern and southern portions of the Site is generally more level.

The Site is located in the Back River watershed which encompasses approximately 39,000 acres of largely developed land, with nearly 90% of the land use in the watershed classified as "Urban" (MDE, 2011). The Back River, a tidal estuary of the Chesapeake Bay, borders the Site to the south.

A tidal wetland area is present along the southern border of the Site adjacent to the Back River. Non-tidal wetland areas are present on adjacent properties along the northwestern, southwestern ("Southwest Finger") and southeastern ("Southeast Finger") borders of the Site. A non-tidal freshwater pond ("the Pond") is present along the northwest boundary of the FOA. Adjacent land use includes residential properties to the northeast, east, and southeast. Forested uplands occur to the north and west. Figure 3.1 presents a Habitat Cover Type Map showing the land use cover types on the Site and adjoining the Site.

Currently, the FOA is heavily overgrown with vegetation and exhibits irregular topography. In general, runoff from the interior of the FOA will tend to travel to the southeast and southwest wetland fingers. Runoff from the southernmost portion of the Site will migrate by sheet flow toward the marsh along the shoreline of the Back River. No noticeable channels or runoff gullies were noted during multiple site visits. The uplands are now occupied by dense vegetation consisting of small trees, shrubs, and grasses (Malcolm-Pirnie, 2007; ATSDR, 2012). The southeast and southwest wetland fingers as well as a majority of the wetlands along the Back River and Embamay contain dense stands of common reed (*Phragmites australis*).

The NRCS Web Soil Survey (Figure 3.2) maps the following soil series within and immediately adjoining the Site:

- UaB: Udorthents; 0 to 8% slopes; well drained; non-hydric

- MhB: Mattapex silt loam; 2 to 5% slopes; moderately well drained; predominantly

     non-hydric

- MT: Mispillion and Transquaking soils; 0 to 1% slopes; tidally flooded; very poorly

     drained; predominantly hydric

A review of the U.S. Fish and Wildlife Service (USFWS) National Wetland Inventory (Figure 3.3) indicates the presence of a pond to the northwest of the FOA. This pond is classified as palustrine, with an unconsolidated bottom that is permanently flooded and diked / impounded (PUBHh). Estuarine, intertidal, emergent wetlands (E2EM5P) which are dominated by *Phragmites australis* and are irregularly flooded are mapped offsite, along the western boundary of the Embamay. The Back River, which occurs immediately south of the Site, is classified as estuarine, sub-tidal (permanently flooded) with an unconsolidated bottom (E1UBL).

The Pond is small and very shallow (less than 2 feet deep) with a bottom of very soft, organic rich mucks. When sampled, bottom sediments were semi-liquid, and, black in color and, thus, appeared to be anoxic. The Pond is surrounded by trees, which will shade much of the water column and contribute considerable leaf litter. Given this, the Pond may be subject to periodic oxygen depletions during summer, and during the winter when ice forms. A review of Google Earth photographs over time also suggests that the Pond may completely dry up during periods of drought. Thus, the Pond's habitat is likely more comparable to a swamp than open water. Consistent with this, previous reports suggest that the Pond lacks fish, and no fish were observed during any of the sampling events associated with the GHD RI. Consequently, primary receptors in the Pond are likely water column and benthic invertebrates, amphibians, and turtles. Insectivorous birds and bats could potentially forage on insects emerging from the pond and its surrounding wetlands.

Both the Southeast and Southwest Fingers and fringe marsh along the shoreline of the Back River are dense, essentially monospecific stands of *Phragmites australis*, a non-native and nuisance species. Site activities, including land manipulation, likely contributed to the presence and character of these *Phragmites* stands. In comparison to native emergent marsh plants (e.g., cattail), *Phragmites* provides poor habitat both because it is not directly consumed by many herbivorous vertebrates or insects and because it so aggressively excludes colonization of other plant species, resulting in very low plant diversity. At the same time, *Phragmites* provides physical cover for some wildlife species. *Phragmites* also has distinct advantages over native wetland plant species in terms of wetland protection against sea level rise and erosion from storm surges. Because of its prodigious growth rates and relative insensitivity to herbivory, *Phragmites* marshes have sediment accretion rates that are double those seen in stands of native vegetation (MDE, 2005).

While Phragmites marshes can serve as cover and foraging habitat for fish, this potential is limited at the Site. There is little to no standing water in most of the wetlands and fringe marsh, except at the extreme high tide and only in the marsh immediately adjacent to the Back River. Hence, fish and – their predators' potential exposure to chemicals in the marsh will be limited to the short periods when water depth is sufficient to support fish habitat. Based on continuous sampling at nearby Stansbury Point (latitude 39.2834, longitude -76.4497), the Back River is a slightly brackish system with salinity averaging 2.3 ppt (part per thousand) and ranging from less than 0.01 to about 8 ppt (MDNR, 2018). Consistent with the salinity data, fish species captured were a combination of freshwater species (American eel, brown bullhead, common carp, channel catfish) and more estuarine species (Atlantic menhaden and white perch) (MDE, 2011). The species noted were captured at various locations in the Back River, as reported in MDE (2011); no fish were captured as part of this RI investigation. Tides in the Back River are minimal: normally about 0.37 meters or 1.2 feet (MES, 1974) or less. Data from Rocky Point at the mouth of the Back River suggest even more moderate tides, on average about 0.9 ft (Anonymous, 2016).[1]

The onsite terrestrial habitat is mostly old field habitat covered by a combination of secondary growth deciduous upland tree (overstory) and shrub (understory) species and herbaceous (ground cover) vegetation typical of disturbed sites. Common species include maple, ash, black locust, sumac, honeysuckle, pokeweed, stilt grass, goldenrod, ragweed, poison ivy and common reed.

A review of the mapping available on the Maryland Environmental Resource and Land Information Network (MERLIN) was conducted for the Site and adjoining areas. This search was updated on August 16, 2019. The MERLIN search included the following data layers: Critical Areas, Wetlands, Protected Lands, and Living Resources; the search outputs are provided in Appendix A. The updated search did not identify any sensitive habitats or known occurrences of rare, threatened, or endangered species on or within the vicinity of the Site. The Critical Areas search indicates that the

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[1] According to available tidal information, most of the marsh associated with the wetland fingers (approximately 80% or more) is above the mean high tide line (about 0.55 feet on the Site figure contours). Mean high water occurs in a slightly larger area, up to about 0.8 feet, but the water is only at this level for minutes per day. Consequently, most of the marsh is literally wet land rather than fish habitat. Tidal information is provided on the Rocky Point, Back River, Maryland Tide Chart (http://tides.mobilegeographics.com/locations/5387.html).

eastern portion of the Site and adjoining areas to the east and north are classified as "Limited Development Area", while the western portion of the Site and adjoining areas to the west and north are classified as "Resource Conservation Area". MERLIN defines "Limited Development Area" as areas with low to moderate development intensity uses or areas that contain housing density ranging from one dwelling unit per five acres up to four dwelling units per acre. It is also defined as areas not dominated by agriculture, wetland, forest, barren land, surface water, or open space. MERLIN defines "Resource Conservation Area" as nature-dominated environments such as wetlands, forests and abandoned fields or areas of resource utilization activities such as agriculture, forestry, fisheries or aquaculture. It is also defined as areas where density is less than one dwelling unit per five acres; or areas with dominant land uses in agriculture, wetland, forest, barren land, surface water, or open space (MERLIN, 2014). The terrestrial areas and wetlands in the western portion of the Site and wetlands immediately adjoining the Site to the west have been impacted by historical Site activities.

A USFWS Information for Planning and Consultation (IPaC) search was conducted for the Site on August 16, 2019 (Appendix B). The species list identified one threatened species, the Northern Long-eared Bat (*Myotis septentrionalis*). However, no critical habitats for the bat occur within the project area under the jurisdiction of the USFWS. The IPaC also did not identify any national wildlife refuge lands or fish hatcheries within the project area.

3.1.2 Contaminants at the Site

Based upon the Data Usability Assessment approved by USEPA on December 3, 2014 (CRA, 2014) and subsequent soil sample collection on residential parcels in 2015, only sample data collected in 2007, 2014 and 2015 were evaluated for risk assessment purposes. Figure 3.4 presents the 2007, 2014 and 2015 sample locations. Based on the soil, groundwater, surface water, and sediment sampling results from these sampling events, the following summarizes the contaminants identified at the Site:

- Elevated concentrations of contaminants exist in on-Site soils, especially PCBs and lead in select areas on the FOA. The area adjacent to and including the Southeast Finger contains elevated PCB and lead concentrations in the surface and subsurface soils and sediments. Based on the existing data PCB and lead impacts extend to contiguous portions of the adjacent residential lots where the FOA overlaps onto the neighboring parcels.

- Sediment samples from the Back River and Embayment showed detections of PAHs, metals including lead, and pesticides. PCBs were not detected in the Back River or Embayment. Sediment samples from the wetlands (Southeast Finger, Southwest Finger, and Pond wetlands) showed detections of metals and PCBs. Elevated lead and PCBs were detected in the wetlands, with the Southeast Finger having the most elevated results. Sediment samples from the Pond showed detections of metals including lead. PCBs were not detected in the Pond.

- Near shore surface water samples were collected in the Back River in 2007; the results showed detections of lead and several PCB congeners. In 2014, surface water samples were taken in the Pond; the results indicated several metals and PCB congener detections. Both filtered and unfiltered samples were analyzed. In general, the filtered results are below the unfiltered sample results.

- Historic sampling of groundwater has reported that groundwater has generally low concentrations of chemicals. Specifically, the groundwater does not contain VOCs, SVOCs, and Pesticides above the applicable human health groundwater standards. Arsenic was present in several wells above the drinking water criterion. No PCB Aroclors were detected in the on-Site groundwater, but low levels of PCB congeners were detected.

As discussed in the RI Report, using a groundwater seepage velocity range of 0.014 to 0.54 ft/day, a flux rate across the Site into the Back River can be made. Using a saturated thickness of 5 feet for the overburden aquifer, and a length of discharge of 150 ft, a total of 750 sq ft of cross-sectional area is available for the flux to the Back River. Multiplying the seepage velocity by the cross-sectional area results in a value of range of 11 to 405 cu.ft/day or approximately 82 to 3,030 gallons/day. GHD believes, based on our experience, the flux rate would be on the lower end of this range. of 4.62E-04 ft/day, a flux rate across the Site into the Back River, Southeast Finger and Southwest Finger was calculated. Using a saturated thickness of 10 feet for the overburden aquifer, and a length of discharge of 200 ft, a total of 2,000 sq ft of cross-sectional area is available for the flux. Multiplying the seepage velocity by the cross-sectional area results in a value of 0.92 cu.ft/day or approximately 6.9 gallons/day. For purposes of comparison, Malcolm Pirnie's 2007 RI estimated 3.1 gallons/day. The current estimate does not account for gradient reversals or changes due to tidal fluctuations. Based on the results of the ENSAT 2001 report, other than MW-4 (1.5 foot fluctuation), the remaining wells showed little to no tidal effect. It can be expected that the gradients established by both Malcolm Pirnie and this report (both 0.003) are a reasonable estimate for use in a net total flux.

In conjunction with the low observed contaminant concentrations in groundwater, a minimal groundwater discharge rate suggests that mass flux of contaminants from groundwater to the surface water are negligible. A screening of the 2014 groundwater results to ESVs indicated a number of SVOCs, metals, PCBs (reported as total congeners) and cyanide exceedances, with the majority of the metals exceedances in the upgradient background well (MW-6). Groundwater will also be evaluated against BTAG screening levels for surface water in order to complete the migration and exposure screening for the conceptual site model.

3.2 Contaminant Fate and Transport

Based on available data and previous assessments described above, PCBs and lead are the primary contaminants at the Site. This also consistent with conclusions from past historical investigations (USEPA, 2005). The following presents a description of their fate, transport, and ecotoxicity.

3.2.1 Fate and Transport of PCBs

PCBs are a group of persistent fat-soluble chemicals that vary in both number of chlorines and placement of those chlorines on the biphenyl molecular backbone. PCBs with the same number of chlorines are called homologues. There are 10 different PCB homologues ranging from monochlorobiphenyl, with only one chlorine, to decachlrorobiphenyl, with chlorines at all 10 available locations. PCBs are also differentiated in terms of the placement of chlorine on the biphenyl molecule. There are 209 different combinations of numbers and placement of chlorines, and these different species are termed PCB congeners.

PCB congeners are odorless, tasteless, clear to pale-yellow, viscous liquids. The fate and transport characteristics of PCBs are roughly correlated with the number of chlorine molecules around the two benzene rings. In general, more chlorinated-PCBs are more hydrophobic, less volatile, less soluble, more persistent, and more bioaccumulative than less chlorinated PCBs. Different PCB congeners and mixtures of congeners can vary in their environmental persistence and toxicity.

In the U.S., PCBs were generally sold as commercial mixtures of PCB congeners termed Aroclors. Aroclors were mixtures of different PCB congeners, but each mixture type had, on average, about the same level of chlorination, which was denoted by the name. Thus, Aroclors 1242, 1248, 1254, and 1260 were about 42%, 48%, 54%, and 60% chlorine by mass, respectively. These average proportions correspond to PCB homologues with 3, 4, 5, and 6 chlorines, respectively; that is, Aroclor 1242 is made up of congeners that, on average, have three chlorines. However, Aroclors are mixtures of different homologues. Thus, while Aroclor 1248 is predominantly tetrachlorobiphenyls, it also contains trichlorobiphenyls and pentachlorobiphyenyls with lesser amounts of the other homologues.

PCBs are only marginally soluble in water and exhibit a strong affinity for sorption with organic matter. It is unlikely, however, that dissolved concentrations would achieve this low solubility limit even in highly contaminated systems because of its affinity for organic matter. Although sorption and subsequent sedimentation immobilizes the bulk of PCBs in an aquatic system, PCBs stored in sediments may nevertheless enter the aquatic food web and accumulate in biota.

In terrestrial ecosystems, PCBs also tend to sorb strongly to soils. In both soils and sediments, the degree of sorption increases with the organic content of the soil. Transport is generally associated with particle transport (e.g., overland water flows) of the "carrier" material, and PCBs do not generally contaminate groundwater nor move readily with groundwater. As in aquatic environments, PCBs in terrestrial systems, if bioavailable, will bioaccumulate and biomagnify in food chains, and are not readily biodegraded in either soil or sediments. Some biodegradation does occur, however, but this process tends to be slow. In soils, the primary loss processes are volatilization, but this also tends to be slow for PCBs below the top layers of soil (i.e., the surficial few inches).

3.2.2   Fate and Transport of Lead

Lead is ubiquitous in the environment and occurs as a trace constituent in rocks, soils, water, plants, animals, and air (Eisler, 1988). The most common form of lead in nature occurs as Pb^{++}, but the divalent metal may be oxidized under strong oxidizing conditions to form Pb^{+++}. Metallic lead is sparingly soluble in hard, basic waters to 30 micrograms per liter (μg/L), although solubility is higher, (up to 500 μg/L in soft, acidic waters). However, even when soluble, lead is a particle reactive metal that tends to sorb strongly to soils and sediments. The adsorption capacity of soil/sediments for lead increases with increasing pH, cation exchange capacity, organic carbon content, soil/water redox potential, and phosphate levels. Only a small percentage of total lead is leachable in soils, with the majority of total lead remaining in solid form or adsorbed onto soil particles. Consequently, the main transport mechanism for lead is through erosion of lead-impacted soil and subsequent sediment transport via surface runoff to nearby aquatic areas. Groundwater is not generally a significant pathway for lead migration under neutral pH conditions, although solubility and transport of lead in groundwater can be enhanced at low pH and high pH in soil.

Given lead's low solubility and high affinity for particles, lead tends to precipitate from aquatic systems and concentrate in the sediments. If the sediments become anoxic, lead will bind tightly to sulfides, after which the lead is insoluble, immobile, and largely non-toxic.

Flow rates of moving waters have a strong influence on the migration and speciation of lead. More turbid waters tend to result in increased concentrations of particulate and unstable lead forms and a decrease in bound lead. Conversely, in low stream flow, lead tends to be removed from the water column, resulting in sedimentation in streambeds (Eisler, 1988).

3.3   Ecotoxicity and Potential Receptors

3.3.1   PCB Ecotoxicity

Because of their hydrophobicity and low biodegradability/excretability, PCBs will readily bioaccumulate and even biomagnify in food chains. Biomagnification is the process whereby concentrations tend to increase with each step in the food chain. Thus, high-trophic-level wildlife and fish may have highest exposure levels to PCBs.

PCBs can elicit a broad range of toxic effects in laboratory vertebrates. Adverse reproductive effects (e.g., litter size, offspring survival) appear to be the most sensitive endpoints of PCB toxicity (Golub et al., 1991; Rice and O'Keefe, 1995; Hoffman et al., 1996). Therefore, although PCBs can cause acute toxicity from direct high-level exposure, the primary ecological concerns associated with PCBs are the potential reproductive effects in higher-trophic-level wildlife resulting from chronic low-level dietary exposure. Reproductive success in fish, mammals, and birds can be affected directly by toxic action on the differentiated reproductive tract or indirectly on systems that regulate reproduction (e.g., endocrine and central nervous systems). In laboratory studies, PCBs have been reported to elicit a broad range of direct and indirect effects that could conceivably lead to decreased reproductive function. For example, the liver is one of the primary targets of PCB toxicity and changes in the activity of liver enzymes can result in modulation of steroid hormone levels, suggesting a mechanism by which PCBs could alter reproductive function. PCBs have also been implicated in the modulation of other systems important for reproduction, such as the central nervous system, adrenal gland, and thyroid hormone levels. Direct effects on the male and female reproductive organs have also been reported (Fuller and Hobson, 1986; Peakall et al., 1986; Barron et al., 1995).

Inherent toxicity of a specific PCB congener depends primarily on the number and position of chlorines. PCB congeners with chlorines substituted at both meta and para positions can exert dioxin-like toxicity, which is the primary mode of vertebrate toxicity for PCB mixtures. An example of this is early-life stage mortality in fish associated with dioxins, furans, and dioxin-like PCBs. As described in the scientific literature (e.g., Elonen et. al., 1998), elevated concentrations of these dioxin-like compounds in fish eggs and larvae can produce significant lethality in some species of cold-water fish, such as salmonids and most especially lake trout (*Salvelinus namaycush*). By comparison, warm water fish eggs and larvae are much less sensitive (Elonen et al., 1998). The Back River serves as habitat for these relatively less sensitive warm-water fish. The critical issue in terms of exposure is the mother's bioaccumulation potential and subsequently maternal transfer of the dioxin-like chemicals to the eggs and larvae. It should also be noted that early-life stage mortality due to PCBs is not well established, even for the most-sensitive and highly exposed lake trout,

Smith (1998) reports a natural experiment -- Canadice Lake in New York -- where lake trout with elevated total PCB concentrations (above 10 mg/kg) nonetheless showed sustained robust reproduction. As these PCB body burdens are well above those body burdens found in aquatic systems (e.g., Lakes Michigan and Huron) where lake trout recruitment is failing, the Canadice Lake data suggest either that the myriad other anthropogenic impacts are critical (Smith, 1994). Alternately, the Canadice Lake examples illustrates that total PCBs in lake trout is a poor indicator of potential toxicity. More likely both factors are critical.

In general, more chlorinated Aroclors tend to be more toxic than less chlorinated Aroclors, although this is not always true. For example, Aroclor 1254 is generally more toxic to mink than Aroclor 1242 which, in turn, is more toxic than Aroclor 1016 (Leonards et al., 1995). Similarly, for rats, Aroclors 1260 and 1254 were much more toxic than Aroclor 1016. However, this general rule may not always hold. For example, Aroclor 1254 tends to contain more of the most dioxin-like toxic congeners, so it could pose more risk to wildlife than more chlorinated mixtures such as Aroclor 1260 (Leonards et al., 1995). Similarly, Aroclor 1248 and 1254 tend to have similar levels of the most toxic congeners (Leonards et al., 1995). Here again, a less chlorinated Aroclor could potentially be as toxic as Aroclor 1254.

Given the importance of dioxin-like toxicity, some have advocated that PCB toxicity be considered in terms of total dioxin-equivalents (TEQ). In cases in which the dioxin-like toxicity of PCBs was significantly reduced by anaerobic degradation, basing risk on TEQ could lead to greater precision in risk estimates (NRC, 2001).

In general, PCBs are much less toxic to invertebrates than to vertebrates, probably because the former lack the Ah receptor that mediates dioxin toxicity (NRC, 2001). PCB toxicity to invertebrates is probably mediated by narcosis (NRC, 2001). Consequently, PCBs in sediments should be only moderately toxic to benthic invertebrates, similar in toxicity to the very high weight PAH compounds such as dibenzo(a,h)anthracene and indeno(1,2,3-cd)pyrene. These PAHs also exert toxicity primarily through narcosis and have similar chemical-physical properties as Aroclor 1248 and 1254. According to recent guidance (USEPA, 2003a), these very high weight PAH compounds, and by extrapolation PCBs, will not be toxic in carbon rich sediments (i.e., > 5% organic carbon) until concentrations approach 100 milligram per kilogram (mg/kg) total PCBs. By comparison, widely used co-occurrence benchmarks for PCBs suggest, erroneously, effects on benthic invertebrates at concentrations as low as 0.04 mg/kg (MacDonald et al., 2000). The co-occurrence benchmarks are, at best, based on an inherently weak, pseudoscientific method – co-incidence with toxicity and, at worst, are totally based on artifacts of complicated, poorly understood calculation methods. Consequently, these benchmarks are not credible predictors of direct toxicity from PCBs (NRC, 2001; Smith and Jones, 2006, 2012). Nonetheless, PCBs in sediments can still pose risk, after bioaccumulation by aquatic life and consumption by vertebrates higher in the food chain, which are very sensitive to PCB toxicity.

Plants also lack the Ah receptor that mediates dioxin toxicity. Thus, PCBs do not typically pose risks to plants.

3.3.2   Lead Ecotoxicity

Lead can be taken up by organisms via inhalation or ingestion. Toxicological effects of lead on aquatic organisms vary depending on different biological and abiotic variables (Wong et al., 1978).

Uptake of lead as part of the diet in animals results in several noted adverse effects on weight, survival, behavior, litter size, and skeletal development (Tsuchiya, 1979). Studies of fish exposed to high concentrations of lead over a long term period have indicated several signs of lead poisoning including spinal curvature; anemia; darkening of the dorsal tail region; spleen, liver, and renal tissue damage; elevated lead concentrations in the blood, bone, gill, liver and kidney; muscular atrophy; paralysis and growth inhibition (Eisler, 1988).

Lead does not readily bioaccumulate or biomagnify in either terrestrial or aquatic food chains, although it will bioaccumulate somewhat in worms and soil biota. Thus, lead tends to be most toxic to lower levels in the food chains.

Lead toxicity to aquatic biota depends on the local conditions of the water body, including pH, hardness, alkalinity, dissolved oxygen, dissolved solids and the presence of other metals. A negative correlation has been identified between lead toxicity and water hardness (Perwak et al., 1981).

3.4   Complete Exposure Pathways

Figure 3.5 presents a Preliminary Conceptual Site Model.

Ecological receptors are not significantly exposed to contaminants in subsurface soils or groundwater. However, aquatic and semi-aquatic biota may be exposed to contaminants in shallow groundwater discharging to surface water and surface sediments. There may also be microorganisms exposed to upwelling groundwater in the hyporheic zone below surface sediments.

Biota could also be exposed directly to contaminants in surface soils, surface sediments, and surface water, and/or indirect exposure via food chain bioaccumulation. Potential ecological receptors include the plants and animals found at the Site, adjacent wetlands and immediately adjacent areas of the Back River.

Soils adjacent to the Southeast Finger and some sediments in the Southwest Finger wetlands are in contact with the waste mass. Site soils are presently stabilized by dense vegetation. At the request of EPA, the "hot spot" area was stabilized by plastic sheeting and continues to be monitored as required by EPA. Prior to the establishment of dense vegetation (*Phragmites*), the migration of contaminants to the Back River potentially occurred. Since the 1990s, this potential migration of contaminants into the Back River has been reduced by the dense, thick, marsh *Phragmites* root systems in the wetlands. Currently, multiple rows of coconut fiber coir logs are also in place along the Back River shoreline to further inhibit soil erosion and potential contaminant migration. The coir logs were installed in February 2006, as directed by EPA's Administrative Order for Removal Response Action (2005 Order) pursuant to Section 106 of CERCLA. The coir logs were upgraded in 2012 and have since become established and are currently functioning as intended.

Near-shore sediment samples in the Back River yielded non-detect concentrations of PCB Aroclors, low to only moderately elevated lead concentrations (10 to 130 mg/kg), and other metals, PAHs and Pesticides (MDE, 2005). Analysis of Back River sediments using USEPA method 1668 identified low levels of PCB congeners, which decreased significantly away from the mouth of the Southeast and Southwest Fingers (Malcolm Pirnie, 2007). With the exception of some elevated metal concentrations, the 2014 sediment samples from the Back River were largely non-detect for SVOCs, Pesticides and PCBs congeners, and non-detect for PCBs Aroclors. Historically, other sources have contributed PCBs to the Back River. The Total Maximum Daily Load (TMDL) decision document,

prepared by MDE in 2011 and approved by USEPA, documents that the Sauer Dump Site represented only 0.09% of the baseline load of PCBs to the Back River.

Based on the above, the potentially complete exposure pathways include the following:

- Exposure of aquatic and semi-aquatic biota to contaminants in shallow groundwater discharging to surface water, surface sediments, and in the hyporheic zone.

- Biota's direct exposure and/or indirect exposure via food chain bioaccumulation to contaminants in surface soils, surface sediments, and surface water.

3.5 Preliminary Assessment and Measurement Endpoints

Assessment endpoints are the specific ecological values that should be protected from Site-related chemicals and guide risk management decisions.

Based on the USEPA guidance (USEPA, 1997 and 1998), habitat and land use (current and future) at the Site, fate and transport, and toxicological aspects of the individual contaminants, the following have been selected as the primary assessment endpoints:

- Survival, growth and reproduction of aquatic organisms inhabiting the water column and aquatic sediments, which are potentially exposed to compounds in discharging groundwater, sediments, and surface water;

- Survival and growth of small mammals and birds ingesting sediment macroinvertebrates, fish and aquatic plants that have potentially bioaccumulated chemicals of potential environmental concern (COPECs); and

- Survival and growth of small mammals and birds ingesting soil macroinvertebrates and terrestrial plants that have potentially bioaccumulated COPECs.

- Survival and growth of larger mammals and birds ingesting small vertebrate prey and terrestrial plants that have potentially bioaccumulated COPECs.

- Survival and growth of terrestrial plants and soil invertebrates exposed to COPECs in surface soils.

Herpetiles (amphibians and reptiles) may also be exposed to Site-related contaminants. Thus, ecological risk assessments typically assume that birds, mammals, and fish serve as surrogates when assessing potential risk to herpetiles. The exception is exposure to amphibians from chemicals in the water column because some water quality criteria and water column ESVs often include toxicity data for amphibians. These assessment endpoints are based on the preliminary Conceptual Site Model, and could change with modifications to the CSM based on new data.

3.6 Preferred Toxicity Data

The preferred toxicity data for the ERA was presented in the Data Usability Assessment and approved by USEPA on December 3, 2014. The 2007 and 2014 sediment data from both the 0 to 2 inch depth interval and the 0 to 6 inch depth interval will be screened. However, risk characterization will focus on the surficial 2 inches, as this is generally considered to be the biologically active zone for macrobenthos. As a first choice, sediment is screened to the USEPA Region III Freshwater

Sediment Benchmarks. Surface water (2007 and 2014) and groundwater data (2007, 20014 and 2015) is also compared to USEPA Region III BTAG values for freshwater.

Surface soil from depths of 0 to 2 feet (2007, 2014 and 2015) is evaluated for purposes of the ERA. Sub-surface soil (depths greater than 2 feet) is not evaluated as part of the ERA. Surface soil is screened to the Ecological Soil Screening Levels (EcoSSLs) and USEPA Region III BTAG Screening Levels.

Thus, preferred USEPA Region III sources of toxicity data are used as a first choice in the initial screening. If the preferred sources lack toxicity data for detected compounds, other reliable sources will be used to provide ecological screening values (ESVs).

3.7   Step 1 Summary

Based on previous investigations, the primary Site contaminants are PCBs and lead, although as typical of salvage yards, the Site also has elevated concentrations of a number of SVOCs and heavy metals. All of these, especially the PCBs and lead, are particle-reactive substances (i.e., they sorb strongly to soil and sediments). Thus, none of these contaminants move readily in groundwater; all primarily move via erosion of surface soils and transport via overland surface flows. If transported to aquatic systems, these contaminants will tend to settle and accumulate in the sediments with the particles to which they are adsorbed.

Current conditions at the Site will greatly reduce transport of the Site's contaminants. The Site's terrestrial area is currently densely vegetated, slopes are typically gentle, and slope-lengths are very short. During multiple Site-visits and sampling events, no erosion furrows or gullies were observed. These factors will encourage the infiltration of precipitation and, thus, suppress both erosion and subsequent transport across the land surface. All runoff will flow to the dense marshes in the wetland fingers or fringe marsh between the Site and Back River, where eroded particles will tend to be trapped by the dense vegetation. Recent modifications – covering of the PCB hot spot and addition of coir logs to prevent erosion of the marsh along the Back River shoreline in 2012, further reduce off-site transport.

The extensive sampling database results corroborate this general model of contaminant fate and transport. While contaminant concentrations of primary Site COPECs are quite high in localized areas of sediments and soils, these concentrations typically decline to near background levels over very short distances, sometimes tens of feet, in apparent flow paths before reaching the Back River. This notable lack of significant efflux from the Site to the Back River was, up to recent sampling events, based largely on surface sediment samples. According to BTAG during RI sample planning, these surface sediment samples might reflect only more recent fluxes of contaminants from the Site. To address historical releases from the Site, BTAG identified an embayment about 600 feet to the west-southwest of the Site, as the area likely to capture and retain historical releases. Sampling of surficial and deep sediments in this embayment was completed to address BTAGs concerns regarding historical loading from the Site.

Samples were taken from the embayment, which was selected by BTAG as an area unlikely to have been eroded over the last several decades and, thus, reflect any potential historical loading from the Site. As with samples taken in the Back River immediately adjacent to the Site, embayment samples also showed no significant levels of PCBs.

Exposure pathways to be assessed in the ERA include the following:

- Exposure of aquatic and semi-aquatic biota to contaminants in shallow groundwater discharging to surface water, surface sediments, and the hyporheic zone.

- Direct exposure of biota to contaminants in surface soils, surface sediments, and surface water, and/or indirect exposure via food chain bioaccumulation.

The potential ecological receptors include the plants and animals found at the Site and in adjacent wetlands and immediately adjacent areas of the Back River.

The area of potentially impacted habitat on the Site and in the Back River is small. The FOA consists of approximately 2.5 acres of ruderal area/old field, 0.52 acre of wetlands, and 0.24 acre of scrub-shrub uplands that transition to maintained lawn further east onto the adjacent residential parcels. Sampling in the adjacent Back River has demonstrated that contaminant efflux from the Site is limited.

4. Screening Level Exposure Estimate (Step 2)

Summary tables of the RI data collected in 2007, 2014 and 2015 are provided in Appendix C. Initially, exposure estimates in the SLERA for all receptors are based on maximum concentrations. For ecological receptors directly exposed to surface water, groundwater, surface soil, and sediment, the maximum concentrations are compared to the ESVs identified for specific receptor groups. The following describe the data that were screened.

4.1 Exposure Parameters

The screening assessment relies on the quotient method, in which an ecological screening quotient (SQ) for each chemical is estimated as:

$$SQ = \frac{EEC}{ESV}$$

where EEC is the estimated exposure concentration and ESV is the ecological screening value, which is also a concentration. In the following analyses, the EEC and SQ values are based on both the maximum and mean concentrations of each chemical. However, as per USEPA guidance, the first screening and selection of initial COPECs will be based on the maximum detected concentration.

In the screening assessment, values of SQ > 1 were flagged to identify chemicals as potentially problematic, and these will be retained as COPECs. For the screening process, duplicate samples and USEPA-split samples were treated as individual samples. A constituent was identified as a preliminary COPEC if:

- The SQ, defined as the maximum concentration divided by its ESV was greater than or equal to 1.0 (i.e., the maximum concentration was greater than its ESV); or

- The constituent was detected, but an ESV for the constituent was not identified.

4.1.1 Surface Water

Surface water samples were collected from the Back River in January 2007 and the Pond in May 2014. These sample results were considered in the ERA screening process. Both filtered and unfiltered 2007 samples were analyzed for lead, PCB Aroclors and PCB homologues. The unfiltered 2014 samples were analyzed for SVOCs, metals, PCBs (Aroclors and congeners), pesticides, and cyanide. The filtered 2014 samples was analyzed for metals, PCBs (Aroclors and congeners), and cyanide. Preliminary screening of surface water was conducted on both the filtered (dissolved) and unfiltered (total) surface water samples. Because sorbed compounds pose minimal exposure to receptors, risk-based decisions were based on the screening of the filtered (dissolved) surface water samples only. It is noted, however, that contaminants can become desorbed based on changing conditions. Additionally, a comparison of total and dissolved concentrations was conducted and there was no significant difference, further supporting consideration of dissolved concentrations in risk-based decisions. A parameter was identified as a COPEC if the SQ for maximum dissolved concentration was greater than or equal to 1.0.

4.1.2 Groundwater

Groundwater samples, both filtered and unfiltered samples, were collected from the onsite monitoring wells in February 2007, July 2014, and January and April 2015. These sample results were considered in the ERA screening process. Both the filtered and unfiltered 2007 samples were analyzed for lead, PCB Aroclors, and PCB homologues. Both the filtered and unfiltered 2014 samples were analyzed for VOCs, SVOCs, PCBs (Aroclors and congeners), pesticides, and metals. The January and April 2015 samples were analyzed for VOCs, SVOCs, metals (filtered and unfiltered), PCB Aroclors and congeners (filtered and unfiltered), and pesticides.

As with surface water, the preliminary screening was conducted on both the filtered (dissolved) and unfiltered (total) groundwater samples. However, sorbed chemicals have minimal potential to move in groundwater and also pose minimal exposure. Thus, as with surface water, risk-based decisions were based on the screening of the filtered (dissolved) groundwater samples only. Again, a comparison of total and dissolved concentrations was conducted and there was no significant difference, further supporting the consideration of dissolved concentrations in risk-based decisions. A parameter was identified as an initial COPEC if the SQ for the maximum dissolved concentration was greater than or equal to 1.0.

4.1.3 Sediment

Sediment samples were collected in 2007 and 2014 from the 0 to 2 inch depth interval and the 0 to 6 inch depth interval. These samples were screened to the USEPA Region III Freshwater Sediment Benchmarks.

Samples were analyzed for VOCs, SVOCs, PCBs, pesticides, and metals. Not all samples were analyzed for all constituent groups.

As with other media, the maximum detected concentration of a constituent was compared to its ESV. For PCBs, the selected benchmarks are based on total concentrations. The concentrations of the individual Aroclors detected in a sample were summed to produce a concentration for total Aroclors. For samples with PCB congener or homologues analyses, total PCBs were calculated by summing the individual congeners or homologues. A parameter was identified as an initial COPEC if the SQ for the maximum concentration was greater than or equal to 1.0.

4.1.4   Surface Soil

Surface soil samples were collected in January and December 2007, May and June 2014 and October 2015. Surface soil is considered to be soil collected from the depth interval of 0 to 2 feet below ground surface (bags), and represents the primary zone of biological activity. Only samples of surface soil from depths of 0 to 2 feet were evaluated for purposes of the ERA. Further, while 0 to 2 feet is evaluated as surface soil for ecological purposes, 0 to 6 inches is typically evaluated separately (or the O and A soil horizons) from the 6 to 24 inch interval as most biological activity occurs within the upper interval. Although sub-surface soil (from depths greater than 2 feet) samples were collected, these were not evaluated as part of the ERA.

Samples were analyzed for VOCs, SVOCs, PCB Aroclors and congeners, pesticides, and metals. Not all samples were analyzed for all constituent groups. All of the 2015 samples were analyzed for lead and several were also analyzed for PCBs. A parameter was identified as an initial COPEC if the SQ for the maximum dissolved concentration was greater than or equal to 1.0.

To address agency comments regarding the presence of hexavalent chromium, sampling was conducted in February 2020 at the same 2014 sample locations and depths as the highest total chromium detections in surface soil (0 to 2 feet) on each residential parcel.

4.1.5   Treatment of Total PCB Congeners, Total Aroclors, and Dioxin-like PCBs in Media

A limited number of PCB congener analyses were conducted in all media. Dioxin toxicity equivalence (TEQ) was estimated by multiplying the latest TEF values for mammals times the concentrations of detected dioxin-like PCB congeners. The TEF values for dioxin-like PCB congeners for mammals were based on those recently proposed by the World Health Organization (Van den Berg et al., 2005). The resulting TEQ values for these samples were then screened against Region III screening levels for 2,3,7,8-TCDD in soil, sediments, and surface water.

5.   Screening-Level Risk Calculation (Step 2)

5.1   Selection of ESVs

5.1.1   ESVs for Surface Water and Groundwater

Observed concentrations in surface water (Pond and Back River) and groundwater were screened against water ESVs protective of aquatic life for freshwater systems. ESVs from Region III BTAG were used as a first choice. These ESVs have the primary advantage of being protective of sensitive

species and accepted by local regulatory agencies. If no Region III ESVs were available for a compound, USEPA chronic WQC for aquatic life were used as ESVs (USEPA, 2006).

For non-polar organic compounds still lacking ESVs, Final Chronic Values (FCVs) for narcotic toxicity were used. The FCVs were developed by independent researchers (e.g., DiToro et al., 2000; DiToro and McGrath, 2000) and then proposed by USEPA as de facto water quality criteria (USEPA, 2003b, 2008) after review by its Science Advisory Board. The narcosis theory suggests that most non-polar organics (e.g., BTEX, PAH, chlorinated and non-chlorinated VOCs) exert toxicity via a common mechanism – narcosis. Consequently, the limited toxicity dataset for any one narcotic chemical (e.g., naphthalene or TCE) on one or a small number of species could be combined with the more extensive data of other narcotic chemicals on other species. By this method, USEPA amassed an extensive toxicity database for many non-polar organics on many species. This dataset included considerable toxicity data for mollusks as well as finfish, crustaceans, insects, and worms. This summed toxicity information on all narcotics and all species was then used to generate safe water column concentrations, or final chronic values (FCVs), for individual PAHs (USEPA, 2003b) and VOCs (USEPA, 2008).

The FCVs for narcotics were generated in a manner similar to aquatic life water quality criteria. That is, the FCVs are intended to be protective of more sensitive species. When available, FCV values for specific chemicals were taken from USEPA (2008) or USEPA (2003b). When FCV values were not available, values for specific chemicals were based on equations presented in USEPA (2008) and log Kow values in USEPA (2005) or from DiToro and McGrath (2001).

Several factors should be considered when assessing the implications of narcosis based screening criteria:

- The different narcotic chemicals exert toxicity through a common mechanism and pose additive toxicity. Thus, quotients of exposure concentrations to FCVs (or equivalent sediment benchmarks) must be summed across all of the non-polar organics. As with typical screening analyses, values of the summed quotient, hereafter called the Total Narcotic SQ, less than or only nominally above 1.0 indicate little potential for toxicity. Total Narcotic SQ values above 1.0 suggest the potential for toxic effects, although such toxicity is not implied.

- The FCVs for the narcotics are intentionally conservative/protective. They are based on the most sensitive species from a list of species that included invertebrates and vertebrates and infaunal, epibenthic/epifaunal (organisms living on the top of the sediment), and water column species. The USEPA dataset included toxicity data for 10 different mollusk taxa (snails and bivalves). These mollusk taxa were generally less sensitive to narcotic toxicity than other species.

For other compounds, WQC were obtained from Michigan DEQ. Michigan derives water quality criteria using the same methods as USEPA, and will, upon request, provide the technical basis for their WQC. Thus, these WQC are generally technically robust and transparent.

Lastly, a few detected compounds (acetophenone, caprolactam, and dimethyl phthalate) still had no ESVs, so these were based on a literature search. Acetophenone is a widely used food additive and, thus, was considered non-toxic. This assessment was confirmed by a search of EPA's EcoTox database, which reported acute bioassays with three species, Daphnia, a ciliate, and three results for fathead minnow. These LC50 values were 162,000 ug/L, 882.000 ug/l, and average of 180,000 ug/L respectively. A chronic ESV was estimated at 16,200 ug/l, based on a 10-fold acute-to-chronic

uncertainty factor applied to the lowest LC50. Gross and Zapp (1984) reviewed the toxicity of caprolactam, and they report essentially non-toxic acute LC50s for three fish species and Daphnia ranging from 4,880,000 μg/L to 1,400,000 μg/L. Inspection of EPA's EcoTox database yielded another acute bioassay, for the golden orfe fish, which had an LC50 of 7,000,000 ug/L. For perspective, these equal about concentrations of 0.7% to 0.1%. Thus, a water column ESV of 140,000 μg/L was developed based on a 10-fold acute-to-chronic uncertainty factor of the lowest LC50. A chronic screening value, 1100 ug/L, for dimethyl phthalate was obtained from EPA Region 4 guidance (USEPA Region 4, 2018).

5.1.2   ESVs for Sediment

For most compounds, sediment screening values were obtained from USEPA Region III when available. As stated above, these ESVs are protective and acceptable to local regulatory agencies.

In general, Region III ESVs are threshold effect levels (TEC) values, which were used to screen chemicals in aquatic sediments. The TEC values are based on a variety of other sediment quality guidelines, such as the National Oceanic and Atmospheric Administration (NOAA) ER-L (Environmental Effects Range – Low) values and Ontario's LEL (Lowest Effects Level). The TEC values are based on larger databases and more varied habitats than any of the individual sources.

TEC values have the advantage of being widely used and widely accepted. They are also protective, so that there is little chance that toxic sediments will be dismissed as non-toxic. However, in the author's opinion, this means TEC values are useful only in screening for lack of toxicity; exceedances of TEC values does not imply that impacts are expected. Moreover, because of the data from which they were derived, TEC values are sometimes lower than naturally occurring background values, making them inefficient as screening values.

TEC values are based on the co-incidence of sediment toxicity/observed impacts with observed concentration of a compound. Consequently, much of the information used to develop a TEC value was potentially due to toxicity from other compounds. Recent analyses indicate that the TEC values are actually simply background values (Smith and Jones, 2005), rather than toxicologically based endpoints or even values that co-occur with toxicity. These problems are manifest for many chemicals that really cannot cause or even contribute to toxicity. Region III has no value for Total PAHs, so the TEC value, 1.61 mg/kg, was used to screen total PAHs.

If no Region III ESVs were available, sediment quality values generated by the Netherlands (Crommentuijn et al. 1997) were used. These values were derived with similar methods as the Dutch soil criteria and have the same advantages. If no TEC or Dutch values were available, ESVs presented in USEPA Region V or NOAA (2008) were used. Calcium, magnesium, potassium and sodium are macronutrients and thus were considered non-toxic. No valid ESVs exist for aluminum in sediments; therefore, the EcoSSL was applied to sediment toxicity. The EcoSSL assumes that aluminum will not be toxic, except at very low pH; thus, aluminum was assumed to be non-toxic.

Some compounds still lacked ESVs. For non-polar organics, the narcosis methods described above were used to develop equivalent sediment benchmarks, as discussed in the references noted above. An ESB for di-n-butyl phthalate was obtained from USEPA (2008).

Benzaldehyde was detected, infrequently at low concentrations in sediments (and soil) at the Site but no benchmark could be found. Potential toxicity of benzaldehyde was assumed to be minimal for

several reasons. First, the compound is a naturally occurring compound (from almonds), although it is also synthesized. It is widely used as an additive to foods and cosmetics; thus, is has very low toxicity to vertebrates (Ohio Department of Health 2013). Second, it will not persist in bioavailable forms in the environment. It is quite soluble, somewhat volatile, and very biodegradable. It is also readily degradable by photolysis with an estimated half-life of less than 10 hours (UNEP 1996). Lastly, while the compound shows slight acute toxicity to aquatic life (PAN 2016), with lowest acute and sub-chronic LC50 values of about 1000 ug/L, these concentrations are not expected to actually occur in water or sediment pore due to the compound's high degradability. Thus, the compound was assumed to be relatively non-toxic in environmental exposures.

Notwithstanding this reasoning, a sediment ESV for benzaldehyde was obtained from Region 4 guidance (2018) and, adjusted, as per that guidance, to Site-specific organic carbon level of 10%. This yielded a final sediment ESV of 590 ug/kg.

5.1.2.1 ESVs for Divalent Metals using AVS/SEM

Several divalent metals (copper, iron, lead, mercury, nickel, and zinc) and chromium were detected at concentrations above the co-occurrence-based sediment quality guidelines (CoSQGs). Because of the problems with CoSQGs and assessments of toxicity based on bulk sediment chemistry, sediments were also analyzed for AVS/SEM (Acid Volatiles Sulfide/Simultaneously Extractable Metals). As described in USEPA (2005), the AVS/SEM method assesses the bioavailability of metals and is therefore a more defensible methods to screen metals' toxicity to benthic invertebrates (USEPA, 2005; Burgess et al., 2013). In contrast to Co-SQGs, the AVS/SEM method specifically estimates differences in bioavailability across different sediments. In sediments, a number of ligands or binding substances bind metals and render them insoluble and/or non-toxic. In sediments, the primary binding agents for divalent metals are sulfides and organic carbon. Sulfides readily combine with most divalent metals (e.g., cadmium, copper, lead, mercury, nickel, silver, zinc) to form insoluble metal-sulfide complexes. The stability constants for most of these metal-sulfide associations are very high; consequently, minimal releases occur from insoluble metal sulfides to soluble and toxic metals in sediment pore water.

It has been suggested that normalizing concentrations of metals in sediment to acid volatile sulfide (AVS) might provide a universal explanation of metal availability from sediments (DiToro et al., 1990; Hansen et al., 1996; USEPA 2005). In turn, the equilibrium partitioning theory for sediment toxicity assumes that low pore water concentrations of metal will translate into limited bioavailability and, by extension, low toxicity. Given this mechanistic understanding, early versions of the AVS/SEM methodology focused on comparing molar concentrations of AVS to molar concentrations of simultaneously extractable metals (SEM). If the molar concentrations of the former were higher than the latter, excess sulfides were assumed to be present in the sediments and all divalent metals were assumed to be bound, rendering them non-bioavailable. Thus, sediments in which molar concentrations of AVS exceeded molar concentrations of SEM were assumed to be non-toxic. In general practice, the decision rule was expressed as a ratio, such that ratios of SEM/AVS less than 1.0 were assumed to be non-toxic.

Since this early work on AVS/SEM, understanding of sediment and water column geochemistry has progressed further (NAS 2003; USEPA 2005). Currently, theory and empirical data suggest that factors in addition to AVS will influence the concentrations of metals in pore waters. Notably, in

sediments, freely dissolved, and potentially toxic metals will distribute among organic ligands, iron oxides, manganese oxides, and perhaps clay surfaces. Typically, organic carbon is the most important of these other binding agents, so most recent AVS/SEM guidance (USEPA 2005) has focused on the additional binding capacity of organic carbon. This newer approach assumes that the potential toxicity of divalent metals that are not bound by AVS and, thus, potentially dissolved in sediment pore water will by further attenuated by binding to organic carbon. To predict potential toxicity from divalent metals, USEPA (2005) focuses on what is called the carbon-normalized excess SEM. This is the simply the difference between SEM and AVS divided by the concentration of organic carbon. Based on both empirical and mechanistic considerations, USEPA (2005) concludes that

> *"Any sediments in which (SEM – AVS) / foc < 130 umols / g_{oc} should pose low risk of adverse ecological effects due to cadmium, copper, lead, nickel, and zinc."*

Thus, the AVS/SEM method generates screening numbers that predict when sediments will likely be non-toxic. Based on empirical results, USEPA suggests that toxicity is uncertain for sediments with carbon normalized excess SEM values between 130 umols / g_{oc} and 3000 umols / g_{oc}. Above 3000 umol/g, sediment toxicity become more likely than not, although some sediments were non-toxic even above this threshold.

Because only trivalent chromium, which is essentially non-toxic, occurs in anoxic sediments, USEPA guidance also concludes that chromium toxicity will not occur when AVS concentrations are detectable. Given their very high binding to AVS, silver and mercury toxicity are also dismissible when AVS concentrations are detectable.

AVS concentrations can vary seasonally. AVS is generated by sulfate-reducing bacteria under sub-oxic conditions, while AVS is oxidized to sulfate when dissolved oxygen is abundant. Consequently, sediment concentrations of AVS will peak in late summer/early fall, when oxygen levels are minimal after sustained periods of high water temperature and microbiological activity. In contrast, sediment concentrations of AVS will be lowest in late winter/early spring under sustained conditions of more abundant oxygen and restrained microbiological activity. USEPA guidance (2005) recommends that AVS/SEM samples be collected when AVS "might be expected to be present at low concentrations." More specifically, the guidance recommends sampling from "November to early May". The AVS/SEM samples described herein were sampled during this optimal window, early May.

5.1.3   ESVs for Soil

If available, ESVs for chemicals in soil were taken from Ecological Soil Screening Levels (EcoSSLs) produced by the USEPA (2005a). These benchmarks have the following advantages:

1. They are intended to be protective of all ecological receptors through both direct toxicity (e.g., toxicity to plants and soil invertebrates) and indirect toxicity by bioaccumulation (e.g., toxicity after bioaccumulation to herbivores and predators); and

2. Their derivation is described in detail in the public domain.

However, for several of the metals (e.g., antimony, cadmium, lead, manganese, and vanadium), the EcoSSLs have the disadvantage of being lower, sometimes considerably lower than naturally occurring background concentrations. ESVs below background concentrations are ineffective,

because they fail to screen out naturally occurring chemicals within background concentrations. For example, soil at virtually all sites in the United States will exceed EcoSSLs for antimony and vanadium. ESVs below background imply that toxicity occurs to wide-ranging species from naturally occurring, wide-raging soil concentrations, making them ineffective in screening out non-problematic chemicals that are present at background concentrations.

The EcoSSLs documents for aluminum and iron are narrative descriptions and do not provide specific screening values. According to these narratives, aluminum is assumed to be non-toxic to ecological receptors except at low pH (< 5.5). Iron is also considered to be a non-problematic constituent in soils at circumneutral pH values. However, pH was not measured, thus, aluminum and iron were retained as potential COPECs.

If no EcoSSLs for an inorganic were available, then USEPA Region III soil ESVs were used. These ESVs have the primary advantage of being conservative and accepted by local regulatory agencies. As in sediments, calcium, magnesium, potassium and sodium are macronutrients and thus were considered non-toxic.

If a detected compound still lacked an ESV, Netherlands MPCs (Maximum Permissible Concentrations) from Crommentuijn et al. (1997) were used as ESVs. The MPCs are estimated as the maximum amount of a metal that could be added to background concentrations without causing toxicity to most species. The derivation of the Netherlands MPCs is transparent, and it is well described in the source document (Crommentuijn et al., 1997). The Dutch values also specifically incorporate background concentrations. On the other hand, these values are primarily based on direct toxicity, although potential toxicity via bioaccumulation pathways is a minor part of the derivation.

If none of the above sources had an ESV, then USEPA (2003) Region V ecological screening levels (ESL) were used. The Region V ESLs consider toxicity by both direct toxicity and bioaccumulation, and they are derived for a wide variety of organic and inorganic compounds.

5.2   Consideration of Bioaccumulative Chemicals in Sediment

The EcoSSLs and many Region V ESVs for soil include potential food chain effects as well as direct toxicity. The same is true of the water column ESVs, which generally consider the most stringent exposure pathway, either via direct toxicity or bioaccumulation for compounds that readily bioaccumulate from water. However, the sediment ESVs are typically based on direct toxicity to the benthos, organisms that live in or on the sediments. Some COPECs may also pose risk to predators, via the food chain, after bioaccumulation by organisms low in the food chain. Thus, the bioaccumulative COPECs in sediments may pose risks to predators of the benthos, if significant bioaccumulation occurs, in addition to posing direct toxicity to the benthos themselves. Potential predators of benthos include birds, such as ducks, and raccoons that feed on the aquatic benthos. After emergence, adults of the larvae benthic insects present a complete exposure pathway from chemicals in sediments to bats, swallows, and other insectivorous birds such as redwing blackbirds. If a COPEC bioaccumulates readily in these benthic invertebrates, these COPECs may be passed up from the sediments through the benthos to the benthos-eating wildlife. Potential bioaccumulative substances are based on Region III BTAG guidance, which consider bioaccumulation potential from sediments to biota (USEPA 2000).

This ERA considers potential toxicity of COPECs through bioaccumulation pathways as well as direct toxicity. Specifically, bats and swallows, which may feed on aquatic insects emerging from on-site sediments, will represent consumers of aquatic benthos.

To be conservative, the potential food chain exposure to COPECs was modeled using worst-case assumptions. That is, these receptors were assumed to eat only contaminated food from the Site for their entire lives. Thus, the aerial insectivores were assumed to eat only aquatic insects emerging from the Site's aquatic environments, despite the fact that their foraging areas are very much larger than the Site. The total exposure for each species was modeled as:

*Total Dose = [food] * consumption rate * absorption efficiency + [soil] * incidental soil consumption rate * absorption efficiency + [water] * drinking rate * absorption efficiency + [air] * inhalation * absorption efficiency + [soil] * dermal absorption rate + [airborne dust] * dust inhalation * absorption efficiency*

All bracketed terms (e.g., [food]) refer to the concentration of the chemical in that medium; other values are self-explanatory. Based on the conservative methodology recommended by the USEPA (1997), relative absorption efficiency was assumed to be 100% for all pathways. On the other hand, the last three terms (exposure via air, dermal absorption, and airborne dust) can be assumed to be insignificant. Water concentrations are similarly very low, so exposure via drinking water is also insignificant. Consequently, the equation collapses to:

*Total Dose = [food] * consumption rate + [soil] * incidental soil consumption rate*

Species-specific ingestion rates were taken from data supplied in USEPA Region 4 (2019). If specific ingestion rates were not available, rates were estimated from consumption-body mass (allometric) models as per USEPA (1993a). Body weights and ingestion rates used for the ERA's measurement receptors are as presented in Table 5.1.

5.2.1   Sources of Toxicity Reference Values (TRVs)

Once exposure is estimated with the food chain models described above, the estimated exposure is then compared to a toxicity reference value (TRV). As recommended by USEPA (1997), TRVs used in the SLERA are conservative NOAELs (no observed adverse effects levels). NOAELs are doses that did not cause significant ecological effects. As described in EPA guidance (1997), "Ecological effects of most concern are those that can impact populations … effects on development, reproduction, and survivorship."

As a first choice, the NOAEL TRVs identified for EcoSSLs were used. This includes the NOAELs for low molecular weight (LMW) and high molecular weight (HMW) PAHs, which are presented in the PAH EcoSSL document but not formally adopted for developing an avian EcoSSL due to insufficient data[2]. As with the EcoSSLs themselves, these ESVs are based on transparent process and received some external review. In general, these TRVs were based on more extensive datasets and quality control that other widely used sources. In many cases, the final TRV was based on the median of many studies, lending confidence in the TRV. However, in many instances, there were low outlier TRVs, and to be conservative, the final TRV was based on this single, outlying estimate.

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[2] More specifically, the NOAEL for HMW PAHs, 2 mg/kg-day, and LMW PAHs, 1653, mg/kg-day were each based on only one study apiece.

This method frequently produced EcoSSLs that were in the range of naturally occurring background for several metals, or for the PAHs, lower than very widely dispersed concentrations.

If no EcoSSL and its TRVs was developed for a COPEC, TRVs were taken from reliable sources. In general, TRVs were taken from Sample et al. (1996), a widely used source. However, the PCB TRV was based on the extensive analysis of James Chapman of USEPA Region V. The mercury NOAELS were taken from multigeneration LOAELs found in Sample et al. (1996).

TRVs are doses of a chemical shown to have no ecological effects on an organism. When the estimated exposure is divided by the TRV, it produces an SQ. As before, SQs below 1.0 indicate that that chemical is unlikely to cause impacts. SQ values above 1.0 indicate that the potential for risk cannot be dismissed with the current analysis and data. Other sources were employed when no values were available from Sample et al. (1996). The TRVs and their sources are listed in Table 5.2. These TRV values were employed without modification for body size or metabolic rate.

5.2.2   Estimation of Bioaccumulative Chemicals in Aquatic Benthos

As a first choice, concentrations of COPECs in aquatic insects were predicted from regression models produced by ORNL (Bechtel Jacobs, 1998a). Models based on all data (depurated and non-depurated) from Table 3 of the ORNL reference were used to predict concentrations. The ORNL models predict dry-weight concentrations. These were converted to wet-weight benthos concentrations by assuming that benthos were 75% water. Estimates of COPECs in benthos were based on sediment samples from the top 6 inches (Table 5.3) and also the more limited data set due to the top 2 inch sediment samples (Table 5.4).

No reliable methods are available to predict benthos concentrations of selenium, and a search of the literature did not yield empirical studies. Therefore, the ORNL models for earthworms and soil were used to estimate dry-weight concentrations in aquatic benthos. Similarly, no good methods are available to estimate benthos concentrations of silver. Therefore, benthos were assumed to have the same dry weight concentrations as the sediments. As above, the estimated dry weight concentrations in benthos were converted to wet weight benthos concentrations assuming that benthos are 75% water.

PAH, PCB, Total DDTs (the sum of DDT, DDE, and DDD), Total chlordane (the sum of chlordane isomers), and heptachlor epoxide concentrations in aquatic benthos were predicted from empirical results of USEPA research (USEPA, 1996a), which was peer reviewed and published in Tracey and Hansen (1996). Based on extensive review of available data, Tracey and Hansen present empirical biota-to-sediment accumulation factors (BSAF) values normalized to organic carbon (OC) in the sediments and lipids in the benthos. Median BSAFs for PAHs, PCBs, and pesticides for benthos and benthically-attached fish were 0.29, 1.1 and 1.8, all on a gram lipid per gram OC basis. The sediments at the Site are very rich in organic carbon, averaging slightly more than 10% in the top 6 inches and about 11% in the top 2 inches. There was also some black carbon, albeit very low concentrations, that would further reduce bioavailability. Aquatic invertebrates have lipid levels of about 2.0% (Oliver and Niimi, 1988). Thus the Site-Specific BSAF values will be about 1/5th the values quoted above, or 0.058, 0.22, and 0.36 times sediment concentrations for PAHs, PCBs, and pesticides.

For assessment of dioxin-like risks from PCBs in sediments, the dioxin-like PCBs were assumed to bioaccumulate as readily as the other PCBs. Thus, for estimating risks to mammal predators of benthos, PCB TEQ in benthos was estimated as 0.22 times the PCB TEQ in sediments, with the latter estimated with the mammalian TEFs. For assessing risks to bird predators of benthos, the PCB TEQ in benthos will be 0.22 times the TEQ in sediments, with the latter estimated with avian TEFs.

A last bioaccumulative substance, according to Region III BTAG, detected in sediments is pentachlorophenol. Although pentachlorophenol is quite hydrophobic (log Kow = to 5.1), the compound is rapidly metabolized (e.g., see USEPA, 1996b, 2000) such that it, like the PAHs, does not readily bioaccumulate in food chains. Rather, both biodilute in food chains. However, no reliable BSAF values could be located for pentachlorophenol. Thus, a conservative BSAF of 1.0 was assumed, as recommended in USEPA (1996a). When applied to the Site-specific carbon (10%) and fish lipid levels (2%), this BSAF equates to a Site-specific accumulation factor of 0.2.

Methylmercury was not analyzed in sediments but it, like the pesticides and PCBs, readily bioaccumulates in food chains. However, the bioavailability of methylmercury (and mercury) will be very low in these sediments, as both decline with higher TOC and higher AVS (Chen et al., 2009). To be conservative, however, it was assumed that the benthos did bioaccumulate methylmercury. Based on the observation that methylmercury in benthos is typically about 25% of total mercury (e.g., see Becker and Bingham 1995, Chen et al. 2009), methylmercury in benthos was assumed to be 1/4th the total mercury concentrations estimated with the ORNL model.

5.2.3 Estimation of Bioaccumulative Chemicals in Fish

Piscivorous wildlife, such as herons and mink, may also be indirectly exposed to sediment contaminants, via contaminants bioaccumulated by their forage-fish prey. However, bioaccumulation of by forage fish will be limited at the Site because little of the sediment associated with the Site occur fish habitat. The pond is likely ephemeral and sub-oxic and reportedly fishless. None were noted in several GHD visits to the Site. Most of the Phragmites marsh has little to no standing water, even at high tide. The nearshore Back River is deep enough for fish, but Site-related chemical concentrations are only moderately elevated for some metals and quite low for total PCBs. Thus, the potential for fish to bioaccumulate Site-related chemicals is very limited.

This multi-step bioaccumulation pathway will limit potential exposure to most of the sediment COPECs. Most of these COPECs, notably the metals (except methylmercury), PAHs, and pentachlorophenol, are readily excreted by vertebrates and some invertebrates. Thus, these chemicals biodilute, i.e., they become less concentrated in upper levels of food chains (e.g., see Thomann and Komlos 1999). Of the sediment COPECs, only PCBs and the pesticides have potential to biomagnify in food chains (i.e., increase in concentration with passage up the food chain from microbenthic prey to forage fish).

Therefore, concentrations of pesticides and PCBs in fish were estimated with empirical BSAF estimated by Tracey and Hansen (1996). As noted above, these authors determined median BSAF values, for benthos and fish consuming benthic invertebrates, to be about 1.1 for PCBs and 1.8 for pesticides. Both values are normalized to body lipids and sediment organic carbon. The dominant fish that would forage in this marsh would be mummichogs, which have average lipid levels of 2.5% (USEPA BSAF database). Based on the previous assumption that sediments are 10% organic

carbon, these parameters produce BSAF values of 0.28 for PCBs and 0.45 for pesticides. Predicted fish concentrations are provided in Table 5.5.

For assessment of dioxin-like risks from PCBs in sediments, the dioxin-like PCBs were assumed to bioaccumulate as readily as the other PCBs. Thus, for estimating risks to mammal predators of fish, PCB TEQ in benthos was estimated as 0.22 times the PCB TEQ in sediments, with the latter estimated with the mammalian TEFs. For assessing risks to bird predators of fish, the PCB TEQ in benthos will be 0.22 times the TEQ in sediments, with the latter estimated with avian TEFs (Van den Berg et al. 1998). Because TEQ for both mammals or birds were essentially the same in the top 2 inch samples as the top 6 inch samples (which includes the former samples), fish concentrations were based on TEQ values for the top 6 inches only, to eliminate essentially duplicative analyses.

Unlike other metals and divalent mercury, methylmercury also biomagnifies in food chains. Its concentrations were conservatively estimated with the biomagnification factor, 6.2 times, for forage fish from zooplankton (USEPA 1997). Because of the low bioavailability of mercury in this carbon-rich and AVS-rich ecosystem (Chen et al., 2009), this is likely a conservative assumption.

6. Results of Preliminary Screening and Initial COPEC Selection

The results of screening the analytical data versus the ESVs are discussed below and are summarized in Tables 6.1 through 6.8. Information presented in these tables includes the maximum detected concentration, the minimum detected concentration, number of samples, frequency of detection (FOD), frequency of exceedance (FOE), Ecological Screening Value (ESV), source of the ESV, SQs for the maximum concentration (MaxSQ). SQ values for the mean concentration (MeanSQ) are also provided. These are calculated two ways: with less than detection concentrations set equal to zero and also set to ½ the detection limit. Initial COPEC selection is based on the MaxSQ. The rationale for retaining or eliminating a constituent as an initial COPEC is also provided.

It should be noted that the mean concentration, FOD, and FOE are most valid when the sampling is representative, i.e., not biased to areas of contamination or lack thereof. For many Superfund sites, sampling tends to be focused specifically on areas of suspected high contamination or known hot spots. Such focused sampling represents a conservative bias, i.e., one tending to bias high the mean, FOD, and FOE. Review of sample locations for this analysis suggest that soil and sediment samples appear to be regularly spaced throughout the study (Figure 3.4). Thus, the mean, FOD, and FOE are not expected to have significant conservative biases due to biased sampling locations.

6.1 Surface Water

Tables 6.1 and 6.2 summarize the surface water screening which included samples from the Back River and the Pond. The screening was conducted on both the unfiltered (total) surface water samples (Table 6.1) and filtered (dissolved) concentrations. However, the selection of initial COPECs is usually based on the screening of the filtered (dissolved) surface water samples only (Table 6.2). This distinction does not matter because the same chemicals, with the exception of arsenic, exceeded ESVs for both total and dissolved samples, although SQ values were generally

much higher with total water samples than with dissolved. Arsenic in unfiltered surface water, 5.7 ug/L, just exceeded the ESV of 5 ug/L but was just under that ESV in filtered water.

VOCs

Surface water samples were not analyzed for VOCs.

SVOCs

No SVOCs were detected in surface water from the River or Pond; therefore, no SVOCs were retained as initial COPECs.

Metals

Aluminum, barium, iron, lead and manganese were retained as initial COPECs in the Pond since their MaxSQ values were greater than 1.0. Lead was the only metal analyzed in the Back River samples (2007), but was not detected above its ESV.

PCB Congeners and Aroclors

Total PCB congeners were well above the screening level for total PCBs in the Pond only, and this contaminant was retained as an initial COPEC. No individual PCB Aroclors were detected in surface water from the Back River or Pond; therefore, none were retained as initial COPECs.

Pesticides

No pesticides were detected in surface water from the Back River or Pond; therefore, none were retained as initial COPECs.

6.2   Groundwater

Tables 6.3 and 6.4 summarize the groundwater screening. The screening was conducted on both the filtered (dissolved) and unfiltered (total) groundwater samples (Table 6.3). However, the selection of initial COPECs is based on the screening of the filtered (dissolved) groundwater samples (Table 6.4). For completeness, BTAG also requested that screening of total groundwater samples should also be discussed.

6.2.1   Total Groundwater

VOCs

Maximum concentrations of carbon disulfide, chlorobenzene, isopropyl benzene, and toluene were above the ESVs in unfiltered groundwater.

S*VOCs*

Maximum concentrations of phenol and several PAHs (2-methylnaphthalene, anthracene, fluoranthene, naphthalene, and phenanthrene) had maximum concentrations above their ESVs in unfiltered groundwater.

Metals

The maximum concentrations of most total metals exceeded their ESVs: aluminum, arsenic, barium, calcium, copper, iron, lead, magnesium, manganese, mercury, selenium, and thallium.

PCB Congeners and Aroclors

In unfiltered groundwater, maximum concentrations of total PCB congeners, total PCB TEQ, and total PCB Aroclors all exceeded their ESVs.

Pesticides and Other Parameters

DDD was the only pesticide detected in unfiltered groundwater just above, about 10% above, its ESV. Cyanide was also detected above its ESV.

6.2.2   Filtered Groundwater

VOCs

Carbon disulfide and isopropyl benzene were the only VOCs retained as initial COPECs.

SVOCs

Naphthalene and phenol had maximum concentrations slightly above, about twice their ESVs. Both were retained as initial COPECs since their MaxSQ is greater than 1.0.

Metals

Many metals exceeded their ESVs and were retained as initial COPECs in groundwater: aluminum, arsenic, barium, calcium, copper, iron, magnesium, manganese, mercury, and selenium. These metals were retained as initial COPECs since their MaxSQs are greater than 1.0.

PCB Congeners and Aroclors

In filtered groundwater, total PCB congeners exceeded the ESV for total PCBs. It was retained as an initial COPEC since its MaxSQ was greater than 1.0. No individual PCB Aroclors were detected in filtered groundwater.

Pesticides

No pesticides were detected in filtered groundwater, so none were retained as initial COPECs.

6.2.3   Pore Water

BTAG had suggested an evaluation of pore water be conducted for groundwater discharging to surface water for parameters exceeding the surface water screening values (ESVs). As noted above, some of the groundwater results exceeded the ESVs for surface water. An evaluation focusing on near-shore monitoring wells MW-1R and MW-2 was therefore conducted and, as stated in the USEPA September 24, 2018 response, the contaminants present in groundwater are not expected to pose an unacceptable ecological risk via the groundwater to the surface water pathway based on the groundwater data presented. Further, USEPA concluded that an evaluation of the hyporheic zone is not warranted at this time. Appendix D presents an Addendum to the SLERA

Report documenting the pore water evaluation and EPA's correspondence with the above determinations. In an email correspondence dated November 13, 2018, USEPA approved the Group's pore water response.

6.3 Sediment

6.3.1 Screening of Sediment Samples from Top 6 inches

Tables 6.5 summarizes the sediment screening. The initial screening and selection of COPECs was conducted on all samples taken from the top 6 inches, which includes the 0 to 6 inch samples and the 0 to 2 inch samples.

SVOCs

Most of the PAHs exceeded ESVs, as did bis(2-ethylhexyl)phthalate. Following conservative screening methods, these were retained as initial COPECs, as their MaxSQs exceeded 1.0.

Metals

Based on the conservative ESVs, most of the non-divalent metals were retained as initial COPECs: antimony, arsenic, barium, beryllium, cobalt, iron, lead, manganese, selenium and vanadium. Most of the AVS/SEM related metals also exceeded conservative ESVs for bulk sediment chemistry, but these were not retained as initial COPECs. All but one AVS/SEM sample had a carbon-normalized excess SEM value below the conservative threshold[3], and the one exception had a negligible (10%) exceedance of the carbon normalized excess SEM threshold. In deference to the more scientifically-defensible AVS/SEM method, the common divalent metals (cadmium, copper, lead, nickel, and zinc) were dismissed as COPECs for toxicity to benthos. Chromium, silver, and mercury will not be at toxic levels in the presence of AVS (USEPA 2005), and they were also dismissed as potential COPECs.

PCB Congeners and PCB Aroclors

Total PCB congeners and PCB Aroclors exceeded their ESVs. These were retained as an initial COPECs since their MaxSQs were greater than 1.0.

Pesticides

All of the detected pesticides, except DDE and DDD, exceeded their ESVs. These were retained as initial COPECs since their MaxSQs were greater than 1.0.

6.3.2 Screening of 0-2 inch Sediment Samples

An additional screening was also completed on the 0 to 2 inch samples only (Table 6.6). Of the initial COPECs identified in Table 6.5, total PCB Congeners, total PCB Aroclors, and metals were also analyzed in the surficial 2 inches, which USEPA (2015) determined typically represents the

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[3] The approach taken is that recommended in EPA guidance (USEPA, 2005). That is, AVS, SEM, and organic carbon for a specific sample were taken to estimate that sample's carbon normalized excess SEM.

biologically active zone. Although concentrations of most COPECs were lower in the surficial 2 inches compared to the surficial 6 inches of sediments, these reductions did not significantly reduce the number of compounds with MaxSQs greater than 1.0.

Two important issues with respect to the biologically active zone in the upper about two inches are the biota involved and stability of the sediments. USEPA's analysis of the biologically active zone focused on macroinvertebrates. However, deeper sediments may include other taxa, such as fungi and bacteria. The second issue concerns the stability of the sediments in the future: erosion of upper layers would make the heretofore deep sediments the upper biologically active zone. Formal sediment stability analyses were not conducted.

6.3.3   Screening Results of Bioaccumulative Chemicals in Sediment

The screening results for bioaccumulative chemicals are presented in Tables 6.7 through 6.9. Potential effects of bioaccumulating chemicals were assessed for samples taken within the surficial 6 inches (including 0 to 2 and 0 to 6 inch samples) and also for the surficial 2 inches only. As shown in the tables, several COPECs potentially pose risk to aerial insectivores (bats and swallows). Based on the maximum exposure assumptions and the maximum concentrations, arsenic, copper, selenium, zinc and Total PCB Aroclors were retained as bioaccumulative COPECs in sediments. For most receptors, total PCB TEQ usually did not pose risks, but following EPA directive, PCB TEQ was retained as a COPEC because total PCB Aroclors were retained.

Risks to fish-eating wildlife from bioaccumulative chemicals in sediments were also considered (Table 6.9). Based on the most conservative exposure scenario, only Total PCB Aroclors were estimated to pose potential risk to fish-eating birds and mammals. Total PCB Aroclors were already retained as an initial COPEC based on potential risks to aerial insectivores. As above, total PCB TEQ did not pose risks to fish eating wildlife. However, following EPA directive, PCB TEQ was retained as a COPEC because total PCB Aroclors were retained.

6.4   Surface Soil

6.4.1   Screening of Soil Samples from Surficial 6 inches

Table 6.10 summarizes the screening for initial COPECs in surface soil for samples collected from the 0 to 6 inch depth interval.

SVOCs

Several PAHs, a few phthalates and phenol exceeded their ESVs. These were retained as initial COPECs since their MaxSQs were greater than 1.0.

Metals

Most of the metals exceeded their ESVs. Antimony, barium, cadmium, chromium, cobalt, copper, lead, manganese, mercury, nickel, selenium, thallium, tin, vanadium and zinc were retained as initial COPECs since their MaxSQs were greater than 1.0.

Of the five (5) hexavalent chromium samples collected in February 2020 from areas of highest known total chromium on the residential parcels surrounding the Sauer Dump, three (3) samples

were collected within the 0 to 6 inch depth interval. Hexavalent chromium was not detected above the MDL in these samples.

PCB Congeners and PCB Aroclors

Both measures of PCBs exceeded their ESVs. Both were retained as initial COPECs since their MaxSQs were greater than 1.0.

Pesticides

Most of the detected pesticides exceeded their ESVs. These were retained as initial COPECs since their MaxSQs were greater than 1.0.

6.4.2   Screening of Soil Samples from 6 to 24 inches

Table 6.11 summarizes the screening for initial COPECs in surface soil for samples collected from the 6 to 24 inch depth interval.

SVOCs

A few PAHs, phthalates and phenols exceeded their ESVs. These were retained as initial COPECs since their MaxSQs were greater than 1.0.

Metals

Most of the metals exceeded their ESVs. Antimony, cadmium, chromium, copper, lead, manganese, mercury, nickel, selenium, thallium, tin, vanadium and zinc were retained as initial COPECs since their MaxSQs were greater than 1.0.

Of the five (5) hexavalent chromium samples collected in February 2020 from areas of highest known total chromium on the residential parcels surrounding the Sauer Dump, two (2) samples were collected within the 6 to 24 inch depth interval. Hexavalent chromium was detected in these samples at concentrations of 0.85 J and 2.0 J mg/kg, which are well below the EcoSSL of 130 mg/kg.

PCB Congeners and PCB Aroclors

Both measures of PCBs exceeded their ESVs. Both were retained as initial COPECs since their MaxSQs were greater than 1.0.

Pesticides

DDD, DDE and DDT exceeded their ESVs. These were retained as initial COPECs since their MaxSQs were greater than 1.0.

Table 6.12 presents background soil results. Several exceedances were observed in background soil demonstrating the conservativeness of the ESVs.

7. Uncertainties

Evaluation of risk to ecological receptors is typically associated with numerous sources of uncertainty. Assumptions must be made regarding exposure concentrations and responses of ecological receptors to COPECs. To avoid incorrectly dismissing actual risks, exposure concentrations and other assumptions are conservatively biased toward identifying risk. As a result of this bias, it can be concluded with a high level of certainty that constituents with MaxSQs below 1.0 do not pose any unacceptable risk to ecological receptors. At the same time, given the highly conservative nature of the screening, a MaxSQ greater than 1.0 does not necessarily demonstrate that the ecological risks actually exist. It only demonstrates that current information is insufficient to dismiss these risks. Specific sources of uncertainty associated with the ERA are discussed below.

For avian and mammalian receptors, a number of assumptions regarding contaminant uptake, bioaccumulation, ingestion rates, and exposure concentrations were made. To account for uncertainty, assumptions were invariably biased conservative. That is these assumptions will, individually, tend to exaggerate risks and, when compounded together, exaggerate risks significantly. For example, chemicals were assumed to be 100% bioavailable, exposures were based on maximum concentrations, ESVs and TRVs were based on no effect levels, and birds and mammals were assumed to eat only most contaminated food only from the Site.

Given the wide variety of animal and plant species and limited toxicological data, there is always some potential that risks for some unusually sensitive species were underestimated. However, because many of the conservative assumptions described above compound and, when multiplied together, produce very large safety factors, there is little uncertainty about conclusions of no risk for compounds not selected as initial COPECs. In contrast, given the high degree of conservatism, there is great uncertainty whether any of the initial COPECs can cause real impacts, even when SQ values are high.

The conservativeness of the screening assessment is well illustrated by the following. Except for the AVS/SEM analyses and the food chain analyses for organics in the sediments, no screening analyses considered the very high binding capacity of the sediments or the moderately high binding capacity to the Site's soil. Illustrating the conservativeness of the screening, the exceptions where Site-specific bioavailability was considered produced far fewer initial COPECs than the other analyses, where the conservative default assumption of high bioavailability was applied.

Another uncertainty pertains to future risks. The sampling data suggest that migration of Site contaminants to the Back River has been minimal, in the case of PCBs, and ecologically negligible in the case of metals. Since the Site is now more vegetated than during dumping operations, the evidence showing no significant recent releases argues that current and future effluxes will be even lower. However, BTAG has expressed concern about factors such as sea level rise associated with global warming. Sea level rise would change the current physical situation and potentially accelerate efflux of COPECs from the Site to the Back River.

While future effects of sea level rise are possible, attendant concerns cannot be addressed quantitatively in the risk assessment since these effects are a function of the following largely unquantifiable unknowns:

1. The likely rate of sea level rise.

2. The concomitant rate of marsh accretion and marsh level rise.

3. The depth in the sediment that PCBs are buried when and if the first exceeds the second.

4. Given 3, the potential for significant erosion of the PCB contaminated sediments.

5. Given 4, the potential consequences if the PCBs were eroded.

While the probabilities and potential effects of these are not quantifiable, the potential risks can be qualitatively assessed by consider just the last factor. For example, the level of sea level rise in the 20th century has been estimated to be about 1.8 mm/yr. (Houghton and Ding, 2001), although more recent rises may be faster, probably 2.6 mm/yr. to 2.9 millimeters (0.11 in/yr.) (National Geographic, 2013). All of these are slower than accretion rates recently estimated for Phragmites marshes in the Chesapeake Bay, 3 to 4 mm/yr. (Rooth et al., 2003). Moreover, a recent study in Nature, states that marsh accretion rates may increase even more with flooding due to "biophysical feedback processes" (Kirwan et al. 2016).

To be conservative, it could be assumed that sea level rise is at the high end of estimated, 3.5 mm/yr., slightly faster than the lower end of marsh accretion, 3 mm/yr. Under this conservative scenario, net sea level rise over the marsh surface is only 0.02 inches per year or a negligible additional inch of water in the marsh over the next half-century. Also over this 50 year period, the marsh accretion will have isolated the currently high PCB concentrations in surface sediment below about 6 inches of new, carbon rich sediments low in PCBs. These calculations suggest that sea level rise will have no real effects on exposure in the future.

Moreover, even if the higher PCB concentrations in the Southeast Finger were, catastrophically, eroded into the Back River, the potential effects on the Back River would also likely be negligible. This prediction is based on the following: 1) the mass of PCBs in the Southeast Finger is small, 2) erosion of PCBs from the Southeast Finger would also erode the very high organic carbon concentrations and the more numerous low PCB concentrations in contiguous areas of the Southeast Finger, and 3) erosion of the Southeast Finger would necessarily co-occur with erosion from many other nearshore marshes in the Back River and in Chesapeake Bay. The net effect of washing a small amount of PCB mass in with relatively enormous masses of carbon-rich sediments from all the coastal marshes in the Back River would not have significant effects on the effective PCB concentrations (i.e., PCB per unit organic carbon) in the Back River.

## 7.1 Fate and Transport Effects of Dioxin-Like Toxicity in PCBs in Different Media

To determine whether fate and transport processes had significantly altered the toxicity of the PCBs, the resulting TEQ estimated for soil, sediment, surface water and groundwater were graphed versus total PCB congeners (see figure at end of section). Total Aroclors were often less than detection, but the limited detections tended to show the same relationship as TEQ vs. total PCB congeners in both soil and sediments.

Across all sediments, the geomean ratio of TEQ, in ng/kg, to total PCB congeners, was about 21. The ratio in soil was about the same but slightly higher, about 27. In contrast to the situation in soil

and sediments, the TEQ per PCB mass is greatly reduced in surface water and especially groundwater. On average, the TEQ in surface water is about 1/10th that shown in for soil and sediments. Thus, the PCBs in the water column, but not necessarily the sediments, in the Pond are much less toxic for equivalent concentrations of Total PCBs. Groundwater is even more depleted in dioxin-like congeners, on average having about 0.05 ppm TEQ per PCBs, 400 times less than the PCBs in the soil. Thus, the groundwater is likely far less toxic than would be expected from its PCB concentrations. In both cases of surface water and groundwater, basing risk assessment on total PCBs in these media will likely exaggerate risks.

In addition, the ESV for surface water, which is applied to groundwater, is based on moderately chlorinated PCBs (mostly penta and hexachlorobiphenyls) that in addition to posing dioxin-like toxicity, also biomagnify significantly in food chains, and are very persistent in the environment.   In contrast, the PCBs congeners in downgradient groundwater wells are about 97% monochlorobiphenlys and dichlorobiphenyls. These chemicals do not readily biomagnify, if at all, are considerably less persistent and degradable, and do not pose any dioxin-like toxicity at all (USEPA, 1995).

Total PCB Congeners (mg/kg) vs. Dioxin TEQ (ng/kg) in Various Media
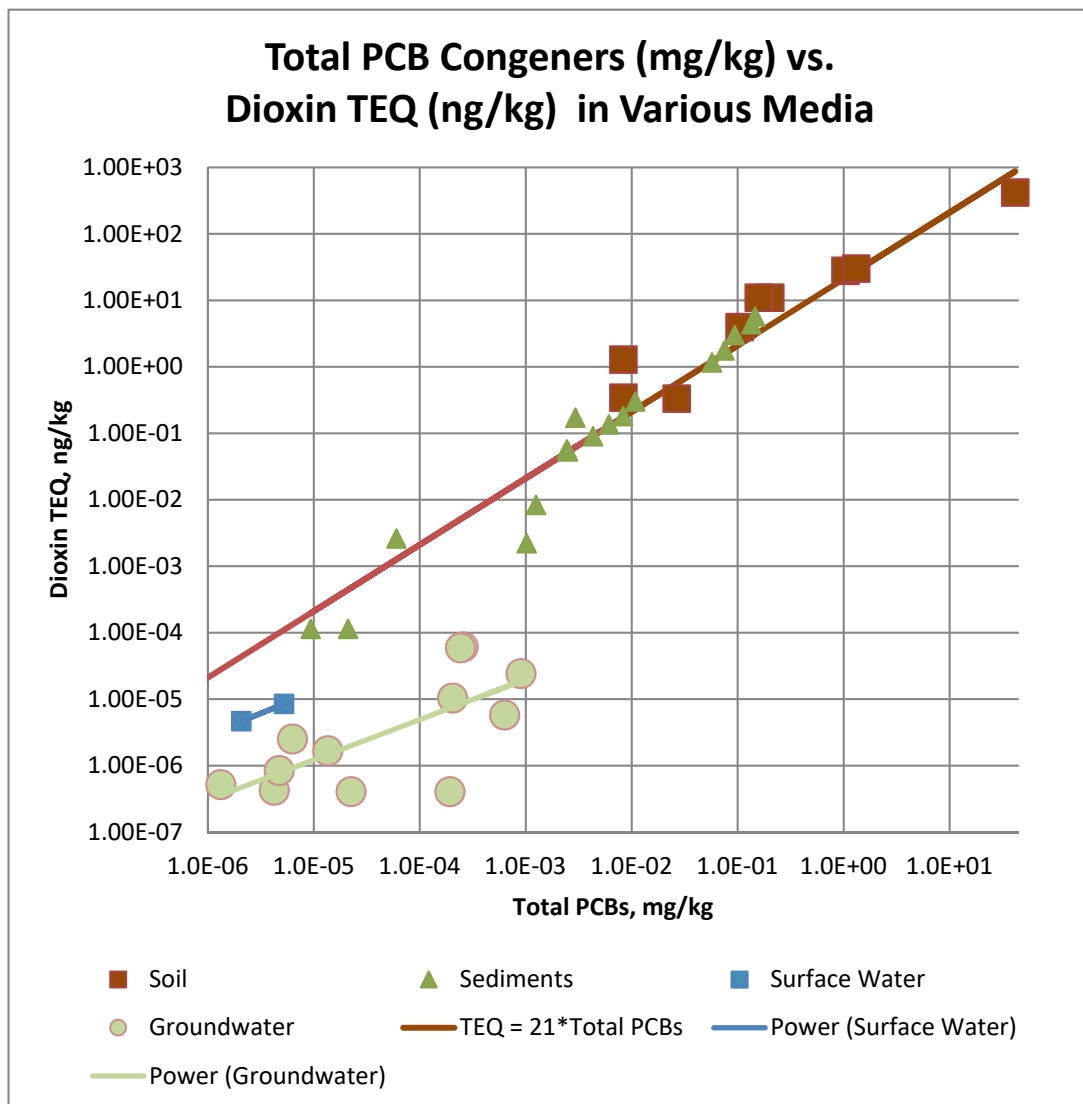
8. Risk Characterization

A large number of compounds were retained as initial COPECs in the ERA. Many of the ESVs for soil and sediments used in the screening are in the range of naturally occurring background concentrations for metals and widely dispersed ambient concentrations for anthropogenic compounds. Identification of initial COPECs was based on maximum concentrations and assumed that receptors were always resident at the Site. Lastly, ESVs also considered sensitive receptors.

USEPA guidance recognizes the conservatism in its screening assessment. This guidance recommends that initial COPECs should be re-evaluated with less conservative assumptions and consideration of Site-specific factors such as bioavailability, habitat quality, and the size of affected area. However, these refinements are recommended to occur after Steps 1 and 2 of the ERA are finished. Refinements are recommended in Step 3, the first Step in a Baseline Ecological Risk Assessment (BERA).

Notwithstanding this guidance, the following presents a qualitative refinement based on the information contained herein. For example, the screening tables also present Mean SQ values based on the mean value (MeanSQ), which is the most accurate estimate of exposure for ecological risk. Even when based on very conservative ESVs, inspection of the screening results shows that MeanSQ values are typically below or only moderately above *de minimis* levels, i.e., SQ values below or only marginally above 1.0.

9. Scientific / Management Decision Point (SMDP)

Following completion of Steps 1 and 2, there are three possible outcomes of the ERA:

1. The information is sufficient to dismiss risks to ecological receptors. If so, neither further ERA analyses nor remediation are warranted.

2. The information is sufficient to demonstrate unacceptable ecological risk. If so, media clean-up levels protective of ecological risk should be determined.

3. The information is not sufficient to make either conclusion. If so, supplemental ERA analyses are necessary to refine the risk.

EPA directed that #2 is most appropriate. However, it is expected that remediation scenarios required for protection of human health will eliminate exposure pathways to ecological receptors.

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