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INTERIM FINAL BASELINE HUMAN HEALTH RISK ASSESSMENT

VOLUME I OF III

Centredale Manor Restoration Project Superfund Site North Providence, Rhode Island

SDMS DocID 242136

BASELINE HUMAN HEALTH RISK ASSESSMENT – INTERIM FINAL

CENTREDALE MANOR RESTORATION PROJECT SUPERFUND SITE
NORTH PROVIDENCE, RHODE ISLAND

Submitted to:

Department of the Army
U.S. Army Corps of Engineers
New England Division

Contract No. DACW33-01-D-0004
Delivery Order No. 01

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BASELINE HUMAN HEALTH RISK ASSESSMENT – INTERIM FINAL

CENTREDALE MANOR RESTORATION PROJECT SUPERFUND SITE
NORTH PROVIDENCE, RHODE ISLAND

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EXECUTIVE SUMMARY

A Baseline Human Health Risk Assessment (BHHRA) has been conducted to evaluate current and potential future human health risks in support of the Remedial Investigation for the Centredale Manor Restoration Project Superfund Site in North Providence, Rhode Island. This BHHRA has been conducted in accordance with the United States Environmental Protection Agency (USEPA) Risk Assessment Guidance for Superfund (RAGS) Parts A, D, and E (USEPA, 1989, 2001c, 2001b), as well as USEPA Region I risk assessment guidance contained in Risk Updates (USEPA, 1994, 1995, 1999).

The main area of the Site, consisting of approximately 9.04 acres, is located in North Providence, Rhode Island, just south of Route 44 on the eastern bank of the Woonasquatucket River. The main area of the Site is known as 2072 and 2074 Smith Street. The remaining portions of the Site consist of reaches, man-made ponds, and wetlands associated with the Woonasquatucket River. The main area of the Site is improved with two apartment buildings for the elderly. These are Centredale Manor Apartments and Brook Village Apartments. Two caps have been constructed at areas known as 2072 and 2074 Smith Street (source area). These caps were constructed to eliminate direct contact soil exposures and to eliminate migration of soil during storm events. In addition, the former tailrace area has been capped and storm water has been re-routed around that area.

The Woonasquatucket River is not used as a source of drinking water and it is designated as a Class B-1 waterbody, suitable for primary and secondary contact recreation and fish and wildlife habitat. The land-use for the northern portion of the Site is expected to remain multi-family residential. The land-use for the eastern shore of Allendale and Lyman Mill reaches is primarily residential with some commercial and industrial activity. A power transmission line is located along the western shore of Allendale Pond and Lyman Mill Pond reaches. It is possible that in the future the western shore of these reaches could be used as a bicycle path. There are residential, commercial, and industrial land uses approximately 200 feet or more from the western shore of Allendale Pond and Lyman Mill Pond. In many locations along those reaches, the river bank is steep and heavily vegetated, making access to the river difficult.

Risk Assessment Overview

The overall BHHRA goals are as follows:

1. to evaluate current and potential future risks to human health associated with the consumption of fish present in the portion of the Woonasquatucket River that constitutes the Site; and
2. to evaluate current and potential future risks to human health associated with human contact with surface water, sediment, and bank soil present in the portion of the Woonasquatucket River that constitutes the Site.

Contaminants that are present in surface water and aquatic sediments may have bioaccumulated in fish and other biota present in the Woonasquatucket River. These fish may be consumed by individuals that catch and/or consume biota from the river. Both neighborhood residents and visitors to the Site could contact surface water, sediment, and bank soils during angling activities or other recreational activities at or adjacent to the river.

The baseline human health risk assessment analyzes potential adverse human health effects for both current and future conditions caused by hazardous substance releases from a site in the absence of any actions to control or mitigate these releases (i.e., under an assumption of no action or in the absence of the Rhode Island Department of Health (RI DOH) health advisories on fish consumption for this particular site). Currently, there is a fish consumption advisory issued by the State of Rhode Island that is not believed to be a sufficient barrier to precluding exposure to biota from the Woonasquatucket River in the short-term. Current and potential future exposure to fish and other biota (including high lipid content biota such as eels) may occur at the river.

Figure ES-1 presents the Conceptual Site Model for the Site. The former chemical manufacturing and industrial activities associated with the Site took place at 2072 and 2074 Smith Street, which are located on the eastern shore of the Woonasquatucket River. These properties are referred to as the source area. Available information indicates that hazardous substances, including dioxins, polychlorinated biphenyls (PCBs), and hexachloroxanthene (HCX) were buried or released to the ground at the source areas and/or were placed in or migrated to the Woonasquatucket River sediments and surface water. Elevated levels of dioxin have also been found in fish taken from the Woonasquatucket River in the vicinity of the Site. Other contaminants detected in Site soils and sediments include chlorinated and aromatic volatile organic compounds, polycyclic aromatic hydrocarbons (PAHs), phthalates, and metals. It is believed the breach of the Allendale Dam in 1991 increased the downstream migration of contaminants and contaminated sediments to Lyman Mill Pond, Manton Pond, and Dyerville Pond. Periodic flooding of the river has resulted in deposition of contaminants and contaminated sediments on bank soils and floodplain soils. Residents living along the river and visiting recreational anglers are potentially exposed via incidental ingestion and dermal contact to contaminants in bank soils, sediments, and surface water, and by consumption of fish and other biota to contaminants that have accumulated in those fish and biota. Dioxins and PCBs are known to accumulate in aquatic biota when they are present in aquatic sediments.

Several actions have been taken to stabilize the source area and eliminate exposures to hazardous substances at the source area and in the immediate area of the source area. These actions include: removal of drums and contaminated soil during construction of Brook Village Apartments and the Centredale Manor Apartments; installation of access-restricting fencing at the tailrace area on the eastern portion of the source area, construction of two protective soil caps, capping of the sluiceway area and re-routing of stormwater around that area, and removals of soil and sediments along the eastern shore of Allendale and Lyman Mill Ponds. Risk assessment activities for the source area (including Brook Village Apartments and Centredale Manor Apartments) and immediately surrounding area (including residential properties on the eastern shore of Allendale Pond and Lyman Mill Pond) were completed in conjunction with these remedial activities. Those risk assessment activities are not repeated in this BHHRA.

The objective of the baseline human health risk assessment is to analyze potential adverse human health effects for both current and future conditions caused by hazardous substance releases from the site in the absence of any actions to control or mitigate these releases (i.e., under an assumption of no action or in the absence of the RI DOH health advisories on fish consumption for this particular site). Currently, there is a fish consumption advisory issued by the State of Rhode Island that is not believed to be a sufficient barrier to precluding exposure to biota from the Woonasquatucket River in the short-term. Current and potential future exposure to fish and

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other biota (including high lipid content biota such as eels) may occur at the river. EPA's goal is the return of a fishable and swimmable condition to the Woonasquatucket River and associated reaches and impoundments.

HAZARD IDENTIFICATION

The purpose of the hazard identification section is to present a compilation of the available sampling data for the hazardous substances present at the site, to identify data sets suitable for use in a quantitative risk evaluation, and to identify contaminants of potential concern in biota, sediment, surface water, and bank soil upon which the quantitative assessment of risk will be based. The BHHRA is based on data collected from several site investigations in soil, surface water, sediment as well as in biota found in and adjacent to the Woonasquatucket River. Data were collected from the Centredale Manor Site source area, four reaches of the Woonasquatucket River adjacent to and downstream of the Site (Allendale Pond, Lyman Mill Pond, Manton Pond, and Dyerville Pond), and at an upstream background area (Greystone Mill Pond, which is upstream on the Woonasquatucket River) and a reference area (Assapumpset Brook, which is a tributary that flows into the Woonasquatucket River at Lyman Mill Pond), both of which are believed to be unimpacted by the Site).

The data evaluation report indicates the analytical data collected at the Site have undergone data validation procedures consistent with USEPA guidelines (MACTEC, 2003). The data validation activities determined that overall, the data that have been collected meet the data quality objectives for the risk assessment activities. The available data were reviewed to identify those data that were representative of current and potential future site conditions and uses and that are therefore suitable for evaluating current and potential future human health risks.

COPC Selection for Biota, Sediment, Surface Water, and Bank Soil.

Using the data collected in soil, surface water, sediment, and biota (American eel, largemouth bass, and white sucker), chemicals were initially identified as chemicals of potential concern (COPCs) by media for the site and the reference/background areas. COPCs require further evaluation in the risk assessment if the chemical concentrations are above risk-based screening concentrations.

Consistent with USEPA Region I guidance, COPCs were selected based on frequency of detection and comparison of detected concentrations to risk-based screening criteria. USEPA Region 9 residential soil Preliminary Remediation Goals (PRGs) were used in the selection of COPCs for bank soil and sediments. Region 9 PRGs for tap water were utilized in the selection of COPCs for surface water and USEPA Region III Risk-Based Concentrations (RBCs) for fish tissue were used in the selection of COPCs for fish tissue consumption.

For the site sediments, COPCs include semivolatile organic compounds (SVOCs), particularly polycyclic aromatic hydrocarbons (PAHs), pesticides, metals and inorganics, polychlorinated biphenyls (PCBs), and dioxins and furans (and hexachloroxanthene (HCX)). In fish tissue, COPCs also include semivolatile organic compounds (SVOCs), particularly polycyclic aromatic hydrocarbons (PAHs), pesticides, metals and inorganics, polychlorinated biphenyls (PCBs), and dioxins and furans. In surface water, COPCs include one volatile organic compound (VOC), one

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SVOC, one pesticide, one PCB (Aroclor-1254), ten metals or inorganics, HCX, and dioxins and furans. In bank soil, 2,3,7,8-TCDD was selected as the COPC.

Dioxins and furans (particularly 2,3,7,8-TCDD), HCX, Aroclor-1254 and possibly PCB-77 appear to be the primary chemical parameters that are detected in environmental media with frequency of detection and concentrations that are indicative of Site-related impacts. In other words, these parameters have clearly elevated concentrations in fish tissue and sediments in the Site-related exposure areas (Allendale Pond, Lyman Mill Pond, Manton Pond, and Dyerville Pond) compared to the Greystone Mill Pond upstream background area and at the Assapumpset Pond and Brook reference area.

The list of COPCs for the background area and reference area is very similar to that for the Site. For the background area and reference area sediment COPCs include semivolatile organic compounds (SVOCs), particularly polycyclic aromatic hydrocarbons (PAHs), pesticides, metals and inorganics, polychlorinated biphenyls (PCBs), and dioxins and furans (and hexachloroxanthene (HCX)). In fish tissue, COPCs also include semivolatile organic compounds (SVOCs), particularly polycyclic aromatic hydrocarbons (PAHs), pesticides, metals and inorganics, polychlorinated biphenyls (PCBs), and dioxins and furans. HCX was not detected and therefore not selected as a COPC for white sucker, but HCX was selected as a COPC for American eel and largemouth bass. In surface water, COPCs include no volatile organic compounds (VOCs), no SVOCs, no pesticides, three metals or inorganics and dioxins and furans.

TOXICITY ASSESSMENT

The purpose of the toxicity assessment is to characterize the relationship between the dose of COPC administered or received and the incidence of adverse health effects in the exposed population. From this quantitative dose-response relationship, toxicity values (e.g., slope factors, reference dose values, or reference concentrations) are derived that can be used to estimate the likelihood of adverse effects as a function of human exposure to an agent. These toxicity values are used in the risk characterization process to estimate the potential for adverse effects occurring in humans at different exposure levels.

The dose-response information may be divided into two major categories:

- toxicity information associated with threshold (non-carcinogenic) health effects.
- toxicity information concerning carcinogenicity, either from human epidemiologic data or from laboratory studies.

All the chemicals selected as COPCs are evaluated for potential non-carcinogenic health effects. In addition, any substance considered to be a known, probable, or possible human carcinogen is also evaluated for its potential carcinogenic effects. The classification of a chemical as a carcinogen does not preclude an evaluation of that same chemical for potential non-carcinogenic health risks, as all potentially carcinogenic chemicals may also exert non-carcinogenic health effects.

Toxicity values were obtained from USEPA recommended sources, including the USEPA's Integrated Risk Information System (IRIS), USEPA Region III Risk-Based Concentration Table,

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the USEPA Region 9 Preliminary Remediation Goals (PRGs) Table, and the USEPA's National Center for Environmental Assessment publications, and various USEPA reports. Published, peer-reviewed toxicity values for HCX were not available. Due to the uncertainty in the toxicity of HCX, that compound has been evaluated separately in Appendix H.

EXPOSURE ASSESSMENT

The objective of the exposure assessment is to estimate the type and magnitude of receptors' exposures to COPCs at or migrating from the site. The exposure assessment is conducted to: 1) characterize the populations of humans potentially exposed via consumption of biota from the Woonasquatucket River and direct contact with surface water, sediment and bank soil at and adjacent to the river; 2) identify the mechanisms by which receptors may be exposed; and 3) identify the intake, or dose, of COPCs that receptors may receive through the identified exposure pathways.

Identification of Potentially Exposed Human Populations

The potentially exposed human populations identified for evaluation in the BHHRA include: Residents Living Along the Woonasquatucket River; Visiting Recreational Anglers who do not live in the immediate vicinity of the Site, but who would visit the Site for recreational angling activities; and employees of The Fogarty Center at 220 Woonasquatucket Avenue. In addition to these receptors, Subsistence Anglers were identified as potential receptors. However, due to the uncertainty in the Subsistence Angler scenario, this receptor has been evaluated in an Appendix to the BHHRA.

Identification of Exposure Points

The exposure points correspond to the exposure areas identified above as follows:

- EA1 is referred to as the Allendale Pond exposure point;
- EA2 is referred to as the Lyman Mill Pond exposure point;
- EA3 is referred to as the Manton Pond exposure point;
- EA4 is referred to as the Dyerville Pond exposure point;
- The Fogarty Center is a specific exposure point on the eastern shore of Lyman Mill Pond;
- The upstream background area is referred to as the Greystone Mill Pond area exposure point; and
- The reference area is referred to as the Assapumpset Brook and Pond exposure point.

Exposure Scenarios and Routes of Exposure

Based on the current and likely future land use of the Site, the BHHRA evaluated the following exposure scenarios. Residents Living Along the Woonasquatucket River (child, older child, and adult) and Visiting Recreational Anglers (child, older child and adult) are assumed to consume a combined fish diet consisting of fish caught at the exposure points and to contact (incidental ingestion and skin contact) surface water and sediment within the Woonasquatucket River, and to contact (incidental ingestion and skin contact) bank soil (Greystone Mill Pond and Allendale Pond exposure points only). For the Visiting Recreational Angler, the child is assumed to consume fish caught by other family members, but it is assumed the young child does not visit the Site for recreational angling and is therefore not exposed to surface water, sediment, or bank soil.

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For the Resident Living Along the River and the Visiting Recreational Angler, exposures were evaluated at a total of six exposure points, including the Allendale Pond reach, Lyman Mill Pond reach, Manton Pond reach, and Dyerville Pond reach as well as the Greystone Mill Pond area (upstream background) and the Assapumpset Brook and Pond (reference area). For the employee of the Fogarty Center, incidental ingestion and skin contact with surface soil have been evaluated.

Exposure Point Concentrations

The Resident Living Along the River and the Visiting recreational Angler have been assumed to each have a favorite fishing spot at one of the identified exposure areas. For each contaminant selected as a COPC, at each exposure point, representative concentrations in fish (American eel, largemouth bass, and white sucker), submerged sediment, surface water, and bank soil were identified as the basis for the exposure assessments. Representative concentrations of COPCs in surface soil were identified for the employee of the Fogarty Center. The representative concentrations (exposure point concentrations or EPCs) were calculated based on the 95% upper confidence limit (UCL) on the mean concentration of the data. The procedures used to identify the 95% UCL and the EPC were selected based on the size of the data set and the distribution type for the concentration data.

Identification of Exposure Models and Parameters

Chemical-specific intakes were calculated in a manner consistent with USEPA guidance for risk assessment. Average daily doses (ADDs) of COPCs were calculated as the measure of exposure. The ADDs are expressed as milligrams of contaminant per kilogram of bodyweight per day (mg/kg/day). For non-cancer health effects calculations, the ADD was averaged over the duration of exposure. For cancer risk calculations, the ADD was averaged over a 70-year lifetime (a lifetime average daily dose or LADD). The following exposure parameters are included in the dose calculations:

- Concentrations in fish tissue, sediment, surface water, and bank soil (C)
- Consumption rate (IR)
- Exposure frequency (EF)
- Fraction ingested from contaminated source (FI)
- Exposure duration (ED)
- Body weight (BW)
- Averaging time (AT) – cancer and non-cancer
- Skin surface area exposed (SA)

Consistent with USEPA guidance (USEPA, 1989), exposures were assessed for both Reasonable Maximum Exposures (RME), expressed as the highest estimate of exposure that is likely to occur and Central Tendency (CT) exposure, which represents typical or average exposure conditions. The two scenarios are assessed to place some boundaries on the estimates of exposure, since the exposures are not actually measured and there is variability among people who might be present at the Site with respect to frequency and duration of exposure, the contact rates and consumption rates, and the locations where they are present now and in the future.

Due to the uncertainties associated with potential dermal exposures to dioxins and furans in surface water, the dermal exposure and associated risks have been evaluated in an Appendix to the BHHRA.

RISK CHARACTERIZATION

Using USEPA-approved toxicity values, potential risks associated with current and future exposure for residents living along the river, visiting recreational anglers, and visiting subsistence anglers were evaluated based on fish consumption, exposures to surface water and sediment at four exposure points (Allendale Pond, Lyman Mill Pond, Manton Pond, and Dyerville Pond) within the Woonasquatucket River, at the Greystone Mill Pond (background) and Assapumpset Brook and Pond (reference area), and exposure to bank soil within Allendale Pond, Lyman Mill Pond, and the Greystone Mill Pond area. Risks to adult workers associated with direct contact with surface soils at the Fogarty Center have also been evaluated. Risks have been calculated using both RME and CT exposure scenarios.

Chemical-Specific Excess Lifetime Cancer Risk was calculated using the following equations:

$$Risk_i = CDI_i \times CSF_i$$

where:

$Risk_i$	=	unitless probability of an individual developing cancer as the result of exposure to a chemical i
CDI_i	=	chronic daily intake of chemical i averaged over 70 years (mg/kg-day)
CSF_i	=	USEPA cancer slope factor for chemical i $(mg/kg\text{-}day)^{-1}$

According to RAGS Part A, p. 8-6 and 8-11, if the estimated risk is equal to or greater than 0.01, an alternative approach (one-hit equation for high carcinogenic risk levels) for calculating cancer risk should be used:

$$Risk_i = 1 - e^{(-CDIi \times CSFi)}$$

where:

$Risk_i$	=	unitless probability of an individual developing cancer as the result of exposure to a chemical i
CDI_i	=	chronic daily intake of chemical i averaged over 70 years (mg/kg-day)
CSF_i	=	USEPA cancer slope factor for chemical i $(mg/kg\text{-}day)^{-1}$
e	=	exponent (value of 2.1817)

Pathway-Specific Incremental Lifetime Cancer Risk:

$$Risk_T = \Sigma\ Risk_i$$

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where:

$Risk_T$	=	unitless probability of an individual developing cancer as the result of multiple chemical exposures
$Risk_i$	=	unitless cancer risk estimate for a single chemical associated with biota consumption

The results from the carcinogenic risk assessment are compared to acceptable risk ranges established by the USEPA. The USEPA's guidelines, established in the National Hazardous Substances and Pollution Contingency Plan (NCP) identify acceptable exposure levels as those concentration levels "that represent an excess upper bound lifetime cancer risk to an individual of between 10^{-4} [one in ten thousand] and 10^{-6} [one in one million] using information on the relationship between dose and response" (USEPA 1990). Where the cumulative RME site risk to an individual exceeds the upper end of this range, action is generally warranted at a site. Where the cumulative RME site risk to an individual is less than 10^{-4}, action is generally not warranted. However, EPA may also decide that a lower level of risk is unacceptable and that action is warranted, if there are extenuating circumstances, such as uncertainties in the risk assessment.

Following are the equations used to determine the Hazard Quotient (HQ)s and HIs.

The following equation is used to determine the hazard quotient:

$$HQ_i = \frac{I_i}{RfD_i}$$

where:

HQi	=	hazard quotient of chemical i
Ii	=	intake of chemical i averaged over the exposure period (mg/kg-day)
RfDi	=	reference dose for chemical i corresponding to the same exposure duration as the intake (mg/kg-day)

The following equation is used to determine the hazard index:

$$HI = \Sigma\, HQ_i$$

where:

HI	=	potential for noncarcinogenic effects from multiple chemical exposures
HQi	=	hazard quotient for each chemical associated with biota consumption

An HI of less than 1 indicates that noncarcinogenic toxic effects are unlikely. An HI greater than 1 indicates a greater possibility of a noncarcinogenic toxic effect occurring, but the circumstances must be evaluated on a case-by-case basis. EPA typically considers action if the Hazard Index is greater than one.

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The incremental cancer and non-cancer risks (the difference between the risks at the Site and the upstream background area, Greystone Mill Pond) have been identified for each receptor at each exposure point. The incremental risks (Site-related risks) have been compared to the Superfund cancer risk range of 10^{-6} to 10^{-4} and to a non-cancer Hazard Index value of 1.

RISK SUMMARY

The cancer and non-cancer risk estimates have been developed for both RME and CT exposure scenarios. The exposure assessment has produced RME and CT estimates of daily contaminant intakes that were based on the exposure models and exposure parameters identified in Section 3. Section 3 contains detailed discussion of the models and exposure parameters. The estimates of cancer risk and non-cancer hazards associated with fish consumption and direct contact with sediments, surface water, and bank soil for Residents Living Along the River and for Visiting Recreational Anglers have been developed using the equations identified above.

Cancer and non-cancer risks have been calculated for each receptor at each of the Site-related exposure points (Allendale Pond, Lyman Mill Pond, Manton Pond, and Dyerville Pond, and the Fogarty Center) as well as at the upstream background location Greystone Pond and at the reference location Assapumpset Pond and Brook. Table ES-1 and Table ES-2 present the summary of RME and CT non-cancer and cancer risks respectively. Overall, cancer and non-cancer risks are greater for fish consumption, surface water contact, and sediment contact at the Site-related exposure points (Allendale Pond, Lyman Mill Pond, Manton Pond, and Dyerville Pond) than at either the background area (Greystone Mill Pond) or the reference area (Assapumpset Pond and Brook).

In order to identify the risks that are related to the Site, the incremental risks have been calculated for the Resident living Along the River and the Visiting Recreational Angler at Allendale Pond, Lyman Mill Pond, Manton Pond, and Dyerville Pond. Incremental risks are defined as the difference in risk between the Site exposure point and the risks at the background location and therefore, reflect the portion of the risk estimate that is Site-specific. These incremental cancer and non-cancer risks (both RME and CT) were subsequently been compared to USEPA Superfund risk management criteria.

Table ES-1 and Table ES-2 present the risk calculated for each receptor at each of the exposure points, the upstream background area, and the reference area, and also the incremental risks associated with each receptor at Allendale Pond, Lyman Mill Pond, Manton Pond, and Dyerville Pond. As shown in Table ES-1, for both RME and CT scenarios for both the Resident Living Along the River and the Visiting Recreational Angler, the fish consumption exposure pathway is by far the predominant contributor to non-cancer risk among the exposure pathways at EA1 through EA4. As shown in Table ES-2, fish consumption and exposure to sediment are the predominant contributors, among the exposure pathways, to the RME and CT cancer risk for the Resident Living Along the River at EA1 through EA4. For the Visiting Recreational Angler, fish consumption is the predominant contributor to the RME and CT cancer risk at EA1 through EA4. For the Resident Living Along the River and the Visiting recreational Angler, fish consumption represents the greatest contributor to cancer risk, based on the fish tissue dioxin equivalent concentrations. Overall, Aroclor-1254 contributes the greatest portion of the non-cancer risk for the Resident Living Along the River and the Visiting Recreational Angler at the exposure points adjacent to and downstream of the Site source area. Dioxin equivalents represent the largest

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chemical contributor to RME and CT cancer risk for the Resident Living Along the River and the Visiting Recreational Angler at the exposure points adjacent to and downstream of the Site source area.

Figure ES-2 presents graphically the summary of RME and CT incremental cancer and non-cancer risks for the Resident Living Along the River. Figure ES-3 presents graphically the summary of RME and CT Incremental cancer and non-cancer risks for the Visiting Recreational Angler. As can be seen in Tables ES-1 and ES-2, no Incremental risks have been identified for the Worker at the Fogarty Center. Incremental risks could not be calculated for the Fogarty Center because there was no true surface soil exposure scenario evaluated at the upstream background area.

Relationship Between Risk Estimates and the EPA Risk Range

Incremental non-cancer risks are compared to the USEPA Superfund benchmark which is a Hazard Index of one. A Hazard Index of less than one indicates that noncarcinogenic toxic effects are unlikely. An HI greater than one indicates a greater possibility of a noncarcinogenic toxic effect occurring, but the circumstances must be evaluated on a case-by-case basis. EPA typically considers action if the Hazard Index is greater than one.

Incremental cancer risks were compared to acceptable risk ranges established by the USEPA. The USEPA's guidelines, established in the National Hazardous Substances and Pollution Contingency Plan (NCP) identify acceptable exposure levels as those concentration levels "that represent an excess upper bound lifetime cancer risk to an individual of between 10^{-4} [one in ten thousand] and 10^{-6} [one in one million] using information on the relationship between dose and response" (USEPA 1990). Where the cumulative RME site risk to an individual exceeds the upper end of this range, action is generally warranted at a site. Where the cumulative RME site risk to an individual is less than 10^{-4}, action is generally not warranted. However, EPA may also decide that a lower level of risk is unacceptable and that action is warranted, if there are extenuating circumstances, such as uncertainties in the risk assessment.

As shown in Table ES-1, both RME and CT Incremental non-cancer risks associated with fish consumption are considerably higher than the non-cancer Hazard Index benchmark of one for the Resident Living Along the River and the Visiting Recreational Angler at Allendale Pond and Lyman Mill Pond. Also as shown in Table ES-2, the Incremental cancer risks associated with fish consumption and exposure to sediment are also considerably higher than the upper end of the Superfund risk range for both receptor groups. Figures ES-2 and ES-3 provide a graphical summary of the Incremental non-cancer and cancer risks. The RME and CT cancer and non-cancer risks are within the Superfund cancer risk range and below a Hazard Index benchmark value of one for adult worker surface soil exposures at the Fogarty Center.

UNCERTAINTY ANALYSIS

The actual fish consumption rates for current and potential future human receptors are uncertain. Reasonably conservative recreational angler consumption rates have been estimated from literature sources and have been used in the BHHRA. It is not clear that subsistence angling is currently taking place or will take place in the future and the potential fish consumption rates for subsistence angling are also uncertain. Therefore, the subsistence angler scenario has been evaluated in an appendix to the BHHRA.

There is currently a fish consumption advisory, issued by the Rhode Island Department of Health (DOH), that may not be completely effective in preventing consumption of fish from Allendale Pond, Lyman Mill Pond, Manton Pond, and Dyerville Pond. However, the advisory may be effective in reducing fish consumption rates in the area. The Rhode Island DOH and the USEPA have also published advice to the public in 1999 to avoid swimming, wading, and bathing in the river and to avoid drinking water from the river. This advice was reinforced with a press release from USEPA and Rhode Island DEM in May and August of 2003. The BHHRA has been conducted to evaluate baseline conditions, in the absence of any measures to eliminate or minimize potential exposures. In that context, the BHHRA may overestimate current exposures associated with the river.

One of the contaminants in fish tissue and sediments, HCX, does not have published, peer-reviewed toxicity values. An appendix to the BHHRA evaluates the potential toxicity and risks associated with HCX exposures, and discusses the potential impacts of HCX on the BHHRA results and conclusions. It appears that cancer risks might be slightly underestimated due to the absence of published, peer-reviewed toxicity information for HCX.

The concentrations in fish tissue of the predominant risk contributors are directly related to corresponding sediment concentrations. Although direct contact exposures to sediments are not the largest risks at the Site, the sediments appear to be associated with the largest risks (those associated with fish consumption) at the Site.

The dermal exposures associated with dioxins and furans in surface water for the current and potential future human receptors are uncertain. Therefore, the dermal exposures and risks associated with dioxins and furans in surface water are evaluated in an Appendix to the BHHRA.

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MAJOR FINDINGS

The major findings of the BHHRA include the following:

- Incremental RME and CT cancer risks from consumption of fish for the current and future Resident Living Along the River and the Visiting Recreational Angler are above the USEPA Superfund risk range at Allendale Pond, Lyman Mill Pond, Manton Pond, and Dyerville Pond.
- Incremental cancer risks (risks above background) associated with dioxin (2,3,7,8-TCDD specifically) from consumption of fish for the current and future Resident Living Along the River and the Visiting Recreational Angler are each above the USEPA Superfund risk range at Allendale Pond, Lyman Mill Pond, Manton Pond, and Dyerville Pond.
- Incremental RME and CT cancer risks (risks above background) for exposure to sediment or bank soil for the current and future Resident Living Along the River and the Visiting Recreational Angler are not (with two exceptions) above the USEPA Superfund risk range at Allendale Pond, Lyman Mill Pond, Manton Pond, and Dyerville Pond. The Incremental RME cancer risks for exposure to sediment for the current and future Resident Living Along the River are above the USEPA Superfund risk range at Allendale Pond and Lyman Mill Pond only.
- Incremental RME and CT cancer risks for exposure to surface water for the current and future Resident living Along the River and the Visiting Recreational Angler are not above the USEPA Superfund risk range at Allendale pond, Lyman Mill Pond, Manton Pond. and Dyerville Pond.
- Cancer risks from exposure to surface soil at the Fogarty Center are not above the USEPA Superfund risk range.
- Incremental non-cancer risks from consumption of fish for the current and future Resident Living Along the River and the Visiting Recreational Angler are equal to or above (by factors between 1 and 27) the USEPA Superfund benchmark Hazard Index of one at Allendale Pond, Lyman Mill Pond, Manton Pond (CT only and Resident Living Along the River only), and Dyerville Pond.
- Incremental non-cancer risks associated with Aroclor-1254 from consumption of fish for the current and future Resident Living Along the River and the Visiting Recreational Angler are equal to or above the USEPA Superfund benchmark Hazard Index of one at Allendale Pond, Lyman Mill Pond, Manton Pond (CT only and Resident Living Along the River only), and Dyerville Pond.
- Incremental non-cancer risks (Hazard Index) from exposure to surface water and sediment for the current and future Resident Living Along the River and the Visiting Recreational Angler are below the USEPA Superfund benchmark Hazard Index of one at Allendale Pond, Lyman Mill Pond, Manton Pond, and Dyerville Pond.
- The non-cancer Hazard Index from exposure to surface soil at the Fogarty Center is below the USEPA Superfund benchmark Hazard index of one.

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NEXT STEPS

The Remedial Investigation (RI) has recently been completed and the Feasibility Study (FS) will soon be completed. The RI determined and summarized the sources, nature and extent of contamination at the Site, characterized the fate and transport of contaminants, and evaluated potential human health and ecological risks resulting from exposure to Site-related contaminants. The FS will evaluate risk management strategies and alternatives for remediating contamination that is found to pose an unacceptable risk to human health or the environment. The FS will also evaluate the long-term effectiveness of the short-term removal actions and determine whether additional action is required to affect a permanent remedy.

In support of the FS, Preliminary Remediation Goals (PRGs) for the fish consumption pathway will be estimated for Chemicals of Concern or COCs (those chemicals that are associated with an excess lifetime cancer risk greater than one-in-one-million and/or a non-cancer hazard quotient greater than one in any medium) by a two-step process. In the first step, risk-based biota PRGs (tissue concentrations) for the COCs will be identified for various fish consumption risk levels (cancer risk of 10^{-6}, 10^{-5}, 10^{-4}, and hazard quotients of 0.1, 1, and 10). In the second step, sediment concentrations corresponding to the biota PRGs will be calculated using the concept of steady-state conditions between lipophilic substances in sediment and fish tissue. Sediment targets associated with direct sediment contact will also be a component of sediment PRG development.

Development of the PRGs will be discussed in further detail in a separate document. The calculated risks for the reference area and background area obviously will be a consideration in the derivation of PRGs and the selection of remedial objectives. At the reference and background areas, the concentrations of some risk contributors in biota are associated with human health risks that are themselves above typical Superfund risk management criteria.

The results of the RI and FS will be used to formulate a Proposed Plan for the Site. The Proposed Plan will recommend remedial actions that will result in overall protection of human health and the environment, fulfill Superfund requirements, be acceptable to stakeholders, and satisfy USEPA remedial guidelines.

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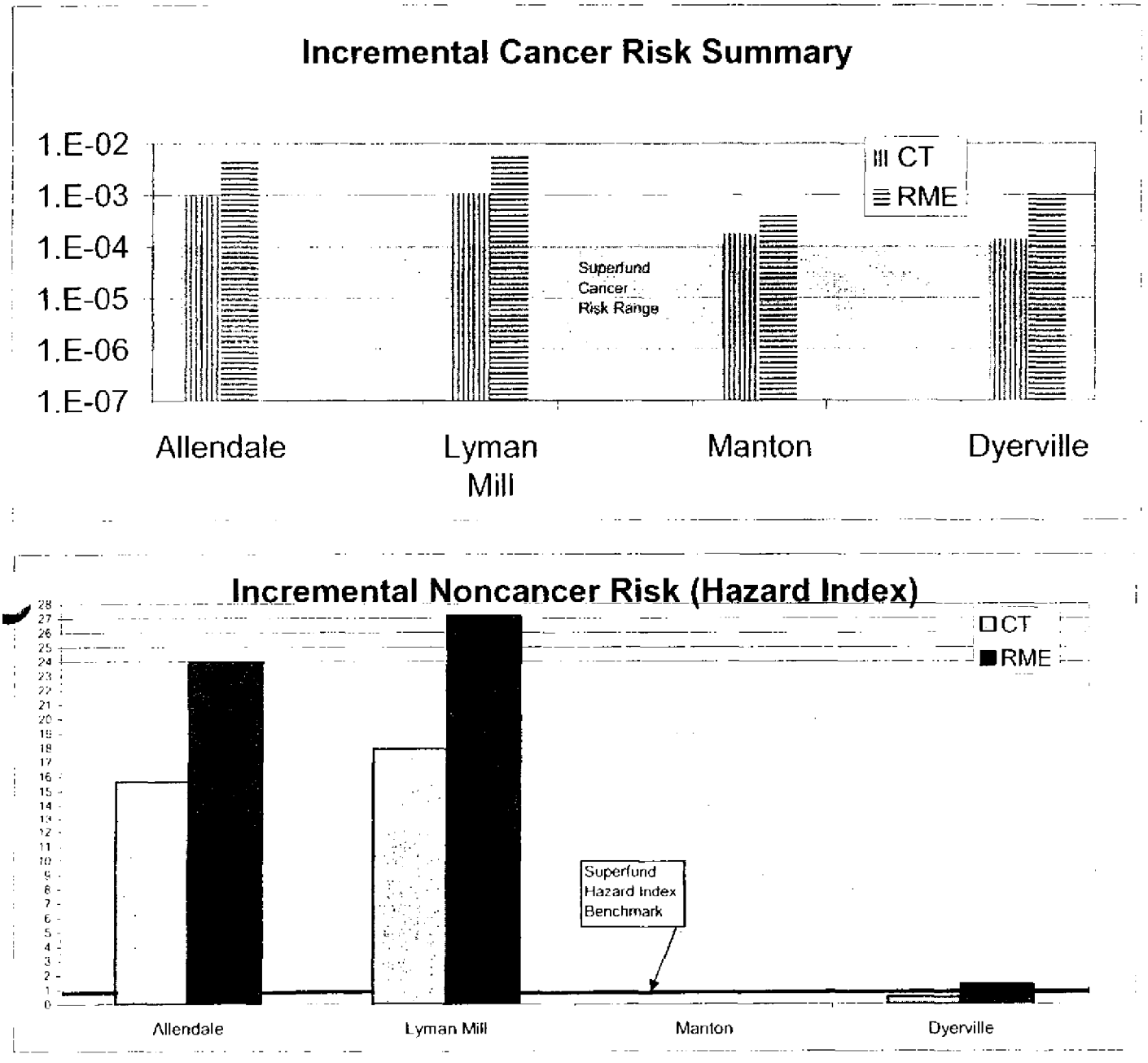
Figure ES-2. Summary of Incremental Cancer and Non-Cancer Risks -
Resident Living Along the River

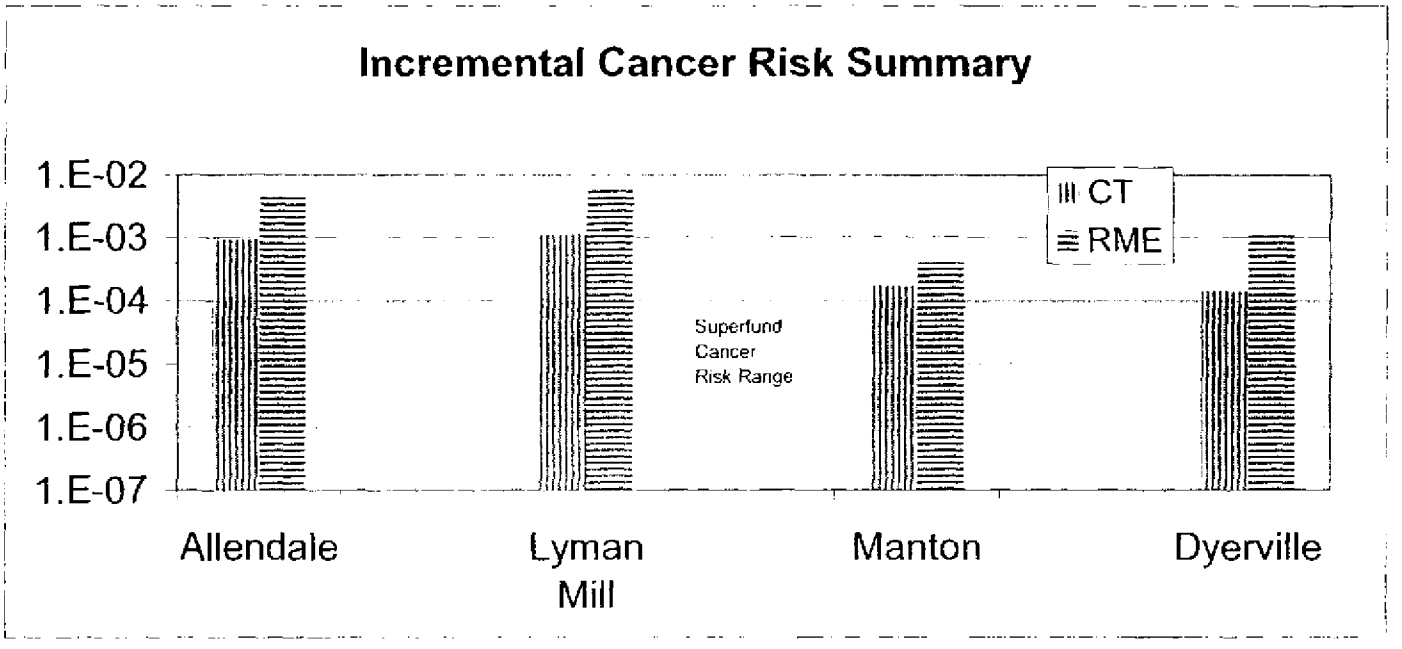
Baseline Human Health Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island

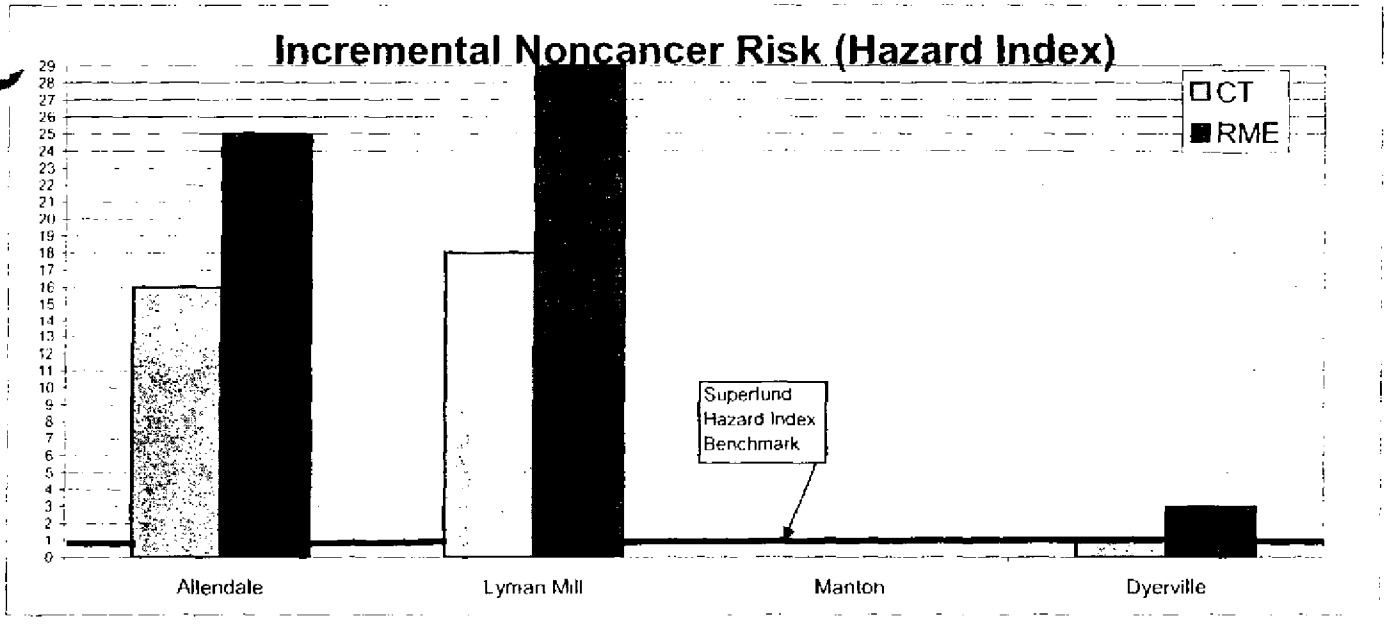


CANCER RISK REPRESENTS THE SUM OF RISKS FOR ALL AGE GROUPS COMBINED
HAZARD INDEX REPORTED FOR YOUNG CHILD
INCREMENTAL HAZARD INDEX AT MANTON IS ZERO FOR RME AND CT SCENARIOS

Figure ES-3. Summary of Incremental Cancer and Non-Cancer Risks -
Visiting Recreational Angler

Baseline Human Health Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island





CANCER RISK REPRESENTS THE SUM OF RISKS FOR ALL AGE GROUPS COMBINED
HAZARD INDEX REPORTED FOR YOUNG CHILD
INCREMENTAL HAZARD INDEX AT MANTON IS ZERO FOR RME AND CT SCENARIOS

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Tab
Summary of Non-Cancer Risks

Baseline Human Health Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island

Non-Carcinogenic Risk			Combined Fish Diet CT	Combined Fish Diet RME	Sediment CT	Sediment RME	Surface Water CT	Surface Water RME	Bank Soil CT	Bank Soil RME	Hazard Index CT	Hazard Index RME	Incremental Hazard Index CT	Incremental Hazard Index RME
Resident Living Along River														
Current & Future	Assapumpset													
		Child	0.5	2	0.02	0.2	0.003	0.009	NA	NA	0.5	2	--	--
		Older Child	0.3	1	0.002	0.02	0.002	0.005	NA	NA	0.3	1	--	--
		Adult	0.3	1	0.001	0.01	0.002	0.004	NA	NA	0.3	1	--	--
	Greystone													
		Child	3	5	0.07	0.5	0.003	0.008	0.2	1	4	7	--	--
		Older Child	2	4	0.005	0.07	0.002	0.004	0.03	0.2	2	4	--	--
		Adult	2	3	0.003	0.04	0.002	0.003	0.02	0.1	2	4	--	--
	Allendale													
		Child	19	30	0.2	0.8	0.03	0.1	NC	NC	19	31	**16**	**24**
		Older Child	13	20	0.02	0.1	0.02	0.06	NC	NC	13	20	**10**	**16**
		Adult	12	19	0.009	0.06	0.02	0.05	NC	NC	12	19	**10**	**16**
	Lyman Mill													
		Child	21	34	0.08	0.4	0.01	0.04	NA	NA	22	34	**18**	**27**
		Older Child	14	22	0.007	0.05	0.01	0.02	NA	NA	14	22	**12**	**18**
		Adult	14	21	0.004	0.03	0.01	0.01	NA	NA	14	22	**11**	**18**
	Manton													
		Child	4	5	0.005	0.03	0.01	0.04	NA	NA	4	6	0	0
		Older Child	1	4	0.0004	0.004	0.01	0.02	NA	NA	1	4	0	0
		Adult	1	3	0.0003	0.002	0.01	0.01	NA	NA	1	4	0	0
	Dyerville													
		Child	4	8	0.05	0.5	0.01	0.04	NA	NA	4	8	0.5	1
		Older Child	3	5	0.004	0.07	0.01	0.02	NA	NA	3	5	0.4	1
		Adult	3	5	0.003	0.04	0.01	0.01	NA	NA	3	5	0.4	1
Visiting Recreational Angler														
Current & Future	Assapumpset													
		Child	0.5	2	NA	NA	NA	NA	NA	NA	0.5	2	--	--
		Older Child	0.3	1	0.0004	0.004	0.0003	0.0009	NA	NA	0.3	1	--	--
		Adult	0.3	1	0.0003	0.002	0.0003	0.0008	NA	NA	0.3	1	--	--
	Greystone													
		Child	3	5	NA	NA	NA	NA	NA	NA	3	5	--	--
		Older Child	2	4	0.001	0.01	0.0003	0.0008	0.002	0.02	2	4	--	--
		Adult	2	3	0.0008	0.007	0.0003	0.0007	0.001	0.01	2	3	--	--
	Allendale													
		Child	19	30	NA	NA	NA	NA	NA	NA	19	30	**16**	**25**
		Older Child	13	20	0.004	0.03	0.003	0.008	NC	NC	13	20	**10**	**16**
		Adult	12	19	0.002	0.01	0.003	0.007	NC	NC	12	19	**10**	**16**
	Lyman Mill													
		Child	21	34	NA	NA	NA	NA	NA	NA	21	34	**18**	**28**
		Older Child	14	22	0.002	0.01	0.001	0.003	NA	NA	14	22	**12**	**19**
		Adult	14	21	0.001	0.005	0.001	0.002	NA	NA	14	22	**11**	**18**
	Manton													
		Child	2	5	NA	NA	NA	NA	NA	NA	2	5	0	0
		Older Child	1	4	0.0001	0.0007	0.001	0.003	NA	NA	1	4	0	0
		Adult	1	3	0.00005	0.0004	0.001	0.002	NA	NA	1	3	0	0
	Dyerville													
		Child	4	8	NA	NA	NA	NA	NA	NA	4	8	0.7	2
		Older Child	3	5	0.001	0.01	0.001	0.003	NA	NA	3	5	0.4	2
		Adult	3	5	0.0006	0.007	0.001	0.002	NA	NA	3	5	0.4	2
Commercial/Industrial Employee														
Current & Future	Fogarty Center [1]		NA	NA	NA	NA	NA	NA	0.01	0.07	0.01	0.07	NB	NB

[1] Surface Soil
CT = Central Tendency
RME = Reasonable Maximum Exposure
NC = Not Calculated - Only dioxin data available for bank soil. No Hazard Index calculated
NB = No background information for surface soil. Incremental Risk not calculated
NA = Not Applicable - Medium not assessed for this exposure area
Incremental Receptor Risk = Difference in risk between the exposure point and the background exposure point
-- = Incremental risk is not calculated for background or reference areas
BOLDED incremental risk are above the Superfund Noncancer Hazard Index benchmark of 1

Prepared by: RAR
Checked by: KJA

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Table
Summary of Cancer Risks

Baseline Human Health Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island

Carcinogenic Risk		Combined Fish Diet		Sediment		Surface Water		Bank Soil		Receptor Total		Incremental Receptor Risk	
		CT	RME	CT	RME	CT	RME	CT	RME	CT	RME	CT	RME
Resident Living Along River (all age groups)													
Current & Future	Assapumpset	6E-06	5E-05	2E-07	6E-06	1E-08	1E-07	NA	NA	6E-06	6E-05	--	--
	Greystone	2E-05	8E-05	1E-06	2E-05	1E-08	1E-07	2E-06	4E-05	2E-05	1E-04	--	--
	Allendale	1E-03	5E-03	2E-05	2E-04	5E-07	6E-06	1E-06	2E-05	1E-03	5E-03	**1E-03**	**5E-03**
	Lyman Mill	1E-03	6E-03	6E-06	3E-04	3E-07	1E-05	NA	NA	1E-03	6E-03	**1E-03**	**6E-03**
	Manton	2E-04	5E-04	1E-06	4E-05	3E-07	1E-05	NA	NA	2E-04	5E-04	**2E-04**	**4E-04**
	Dyerville	2E-04	1E-03	7E-07	2E-05	3E-07	1E-05	NA	NA	2E-04	1E-03	**1E-04**	**1E-03**
Visiting Recreational Angler (all age groups)													
Current & Future	Assapumpset	6E-06	5E-05	1E-08	4E-07	3E-10	3E-09	NA	NA	6E-06	5E-05	--	--
	Greystone	2E-05	8E-05	5E-08	2E-06	3E-10	3E-09	6E-08	2E-06	2E-05	9E-05	--	--
	Allendale	1E-03	5E-03	8E-07	1E-05	1E-08	2E-07	3E-08	7E-07	1E-03	5E-03	**9E-04**	**5E-03**
	Lyman Mill	1E-03	6E-03	3E-07	2E-05	2E-08	3E-07	NA	NA	1E-03	6E-03	**1E-03**	**6E-03**
	Manton	2E-04	5E-04	6E-08	2E-06	2E-08	3E-07	NA	NA	2E-04	5E-04	**2E-04**	**4E-04**
	Dyerville	2E-04	1E-03	3E-08	1E-06	2E-08	3E-07	NA	NA	2E-04	1E-03	**1E-04**	**1E-03**
Commercial/Industrial Employee													
Current & Future	Fogarty Center [1]	NA	NA	NA	NA	NA	NA	8E-07	2E-05	8E-07	2E-05	NB	NB

(1) Surface Soil
CT = Central Tendency
RME = Reasonable Maximum Exposure
NB = No background information for surface soil. Incremental Risk not calculated
NA = Not Applicable - Medium not assessed for this exposure area
Incremental Receptor Risk = Difference in risk between the exposure point and the background exposure point
-- = Incremental risk is not calculated for background on reference areas
BOLDED incremental risk are above the high end of the Superfund Cancer risk Range (1E-04 to 1E-06)

Prepared by: KJA
Checked by: RAR

1.0 INTRODUCTION

The U.S. Environmental Protection Agency (USEPA) Region I and U.S. Army Corps of Engineers (USACE), New England District are conducting a Baseline Human Health Risk Assessment (BHHRA) and a Baseline Ecological Risk Assessment (BERA) for the Centredale Manor Restoration Superfund Site located in North Providence, Rhode Island (hereafter referred to as "the Site"). A site location map is provided as Figure 1. A layout of the Site, including important Site features and the surrounding areas is provided as Figure 2.

1.1 BACKGROUND AND INVESTIGATION HISTORY

Harding ESE previously identified project goals, objectives, and data requirements to support human health and ecological risk assessments for the Site (Harding ESE, 2000). In the Work Plan prepared to support the human health and ecological risk assessments (Harding ESE, 2001a), Harding ESE evaluated the existing data and documents (as of March 2001) provided by USEPA and the USACE. Available information was reviewed to determine the need for additional data for successfully accomplishing the planned biota consumption risk assessment (BCRA) and ecological risk assessment. During the data review, the number of samples collected for each medium, chemical analyses performed for samples collected during particular sampling events, and the location of samples collected were evaluated. It was determined that insufficient historical biota, soil, sediment, and surface water data were available to perform the BCRA and BERA and that additional samples needed to be collected from these media to fill the data gaps. Based on this information, a Field Sampling Plan (FSP), Harding ESE, 2001b) was prepared to identify the specific data that would be required for each medium in each exposure area in order to evaluate the human health biota consumption pathway and ecological exposure pathways.

The BCRA/BERA 2001 field collection activities began in May 2001. Tissue samples collected in the Allendale and Lyman Mill reaches and in upstream areas included fish, crayfish, emerging insects, and earthworms. Media sampled in those areas included floodplain soil, aquatic sediment, and surface water. In the Assapumpset reference area, fish, crayfish, aquatic sediment, and surface water samples were also collected. Fish sampling was conducted in Manton and Dyerville Ponds. No other sampling occurred in those reaches. BCRA/BERA field activities were completed in late July 2001. The abiotic media data were collected to support the bioassessment data. A variety of bioassessment studies were identified as necessary to conduct the ecological risk assessment (e.g., floodplain invertebrate study, aquatic macroinvertebrates study). The additional sediment, soil, and surface water samples were necessary to interpret the additional biological data.

The samples collected during the BCRA/BERA field sampling program were analyzed or tested for chemical, physical, and biological parameters by laboratories at several organizations, including:

PARAMETERS	LABORATORY
• Dioxin/Furan and PCB Congener	Battelle, Columbus, OH
• PCB Aroclor and Chlorinated Pesticides	Battelle, Duxbury, MA
• Polycyclic aromatic hydrocarbons	Battelle, Duxbury, MA
• Metals, AVS/SEM	Battelle, Sequim, WA

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• SVOCs and VOCs	Severn Trent Laboratories, Baltimore, MD
• BOD	Cassell Testing, Hunt Valley, MD
• Nutrients	Chesapeake Biological Laboratory, Solomons, MD
• Sediment Toxicity	Great Lakes Environmental Center, Traverse City, MD
• Taxonomy	Normandeau Associates, Bedford, NH

Additional field efforts and analyses were performed by other parties in 2001. USGS collected tree swallow samples in May and June of 2000 and 2001. USACE collected fish data to support an early life stage (ELS) bioassay conducted by Battelle. TetraTech collected floodplain soil samples in June and July, sediment samples in July, and groundwater samples in August. The scope of work for the BCRA was subsequently expanded to include additional media and exposure pathways, including potential exposures to surface water, sediment, bank soil, and surface soil. Because the scope of the baseline risk assessment was no longer limited to biota consumption only, it was referred to as the baseline human health risk assessment or BHHRA.

The overall BHHRA goals are as follows:

1. to evaluate current and potential future risks to human health associated with the consumption of fish present in the portion of the Woonasquatucket River that constitutes the Site; and
2. to evaluate current and potential future risks to human health associated with human contact with surface water, sediment, and bank soil present in the portion of the Woonasquatucket River that constitutes the Site.

Contaminants that are present in surface water and aquatic sediments may have bioaccumulated in fish and other biota present in the Woonasquatucket River. These fish may be consumed by individuals that catch and/or consume biota from the river. Both neighborhood residents and visitors to the Site could contact surface water, sediment, and bank soils during angling activities or other recreational activities at or adjacent to the river.

The baseline human health risk assessment analyzes potential adverse human health effects for both current and future conditions caused by hazardous substance releases from the site in the absence of any actions to control or mitigate these releases (i.e., under an assumption of no action or in the absence of the Rhode Island Department of Health (RI DOH) health advisories on fish consumption for this particular site). Currently, there is a fish consumption advisory issued by the State of Rhode Island that is not believed to be a sufficient barrier to precluding exposure to biota from the Woonasquatucket River in the short-term. Current and potential future exposure to fish and other biota (including high lipid content biota such as eels) may occur at the river.

Organization of Report

This BHHRA report is divided into seven sections: an introduction is provided in Section 1.0; a hazard identification is presented in Section 2.0; exposure assessment including receptor identification, development of exposure profiles and exposure point concentrations is presented in Section 3.0; the toxicity assessment is presented in Section 4.0, the risk characterization is contained in Section 5.0, the uncertainty analysis is discussed in Section 6.0, and the Conclusions and Recommendations are presented in Section 7.0, and the development of Preliminary Remediation Goals is discussed in Section 8.0. Appendix A addresses sample selection considerations for the risk assessment. Appendix B contains the photographs of site features to

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support the exposure assessment. The documentation of the biota consumption rates used in this assessment is presented in Appendix C. The toxicity assessment supporting information is presented in Appendix D. Additional risk characterization supporting information is presented in Appendix E. The risk characterization results for the potential visiting subsistence angler scenario are presented in Appendix F. The "high-end" fish consumption assessment for the Visiting Recreational Angler and the Resident Living Along the River is presented in Appendix G. Appendix H contains the human health risk assessment associated with the compound HCX detected in biota, sediment, and surface water. This compound has been evaluated in that appendix due to the uncertainties in the toxicity of the compound. In addition, Appendix I presents an evaluation of the potential exposures and associated risks for dioxins and furans in surface water.

The table numbering in this report is consistent with the numbering of Tables in the USEPA's *Risk Assessment Guidance for Superfund (RAGS): Volume I - Human Health Evaluation Manual (Part D, Standardized Planning, Reporting and Review of Superfund Risk Assessments) Final* (USEPA, 2001c). That guidance includes standardized tables (with a specific numbering scheme) for reporting risk assessment activities. For each group of tables (such as the Table 2s that present the selection of chemicals of potential concern), the tables are numbered consecutively.

1.2 DESCRIPTION OF SITE

The main area of the Site, consisting of approximately 9.04 acres, is located in North Providence, Rhode Island, just south of Route 44 on the eastern bank of the Woonasquatucket River. The main area of the Site is known as 2072 and 2074 Smith Street. The remaining portions of the Site consist of reaches, man-made ponds, and wetlands associated with the Woonasquatucket River.

As shown in Figure 2, currently, two high rise buildings (Centredale Manor and Brook Village, both federally-subsidized, senior housing complexes) are located in the main area of the Site, which is zoned for residential occupancy. In addition to the buildings, the main area is covered by roadway and parking lots. On the eastern portion of the main area is a drainage swale that begins near the northern portion of the main area and extends south, then curves to the west and discharges south of the main area into the Woonasquatucket River and Allendale Pond. Stormwater has been redirected around this area and a cap has been installed over the drainage swale area. This area has also been referred to as the "sluiceway" and the "tailrace" area. This cap is referred to in this document as the two-foot cap in the sluiceway area.

The Site was used for disposal of wastes containing hazardous substances. Between approximately 1940 and 1970, USEPA has reason to believe that the main area of the Site was the location of a chemical manufacturing facility (which produced hexachlorophene, among other chemicals) and an incineration-based drum recycling facility. The first housing complex was built in approximately 1976. The second high rise was built in 1982. Construction records show that hazardous substances were removed from the Site during the construction of the second complex. Soil sampling and analysis at the Site have detected elevated levels of dioxin (particularly 2,3,7,8-TCDD), PCBs, VOCs, SVOCs, and selected metals (Battelle, 2003b). Aquatic sediment sampling and analysis has revealed that dioxin, PCBs (particularly Aroclor-1254), PAHs, other SVOCs, pesticides, hexachloroxanthene, and metals were present in sediments. Elevated levels of dioxins were generally limited to the top two feet of sediments, with the highest 2,3,7,8-TCDD concentrations reported for depositional wetland areas of

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Allendale Pond and a decrease in concentrations with distance downstream of the source area (Battelle 2003b). Manton Pond and Dyerville Pond sediment 2,3,7,8-TCDD concentrations are reported to be lower than in Allendale Pond and Lyman Mill Pond. It is reported that Aroclor-1254 is the predominant PCB in aquatic sediments and that Aroclor-1254 concentrations were above background concentrations in approximately 50% of the aquatic sediment samples collected from the study area (Battelle, 2003b). The compound, hexachloroxanthene (or HCX) has been detected in Site soils and in aquatic sediments within the study area. It is reported that hexachloroxanthene is a constituent of commercial hexachlorophene (WHO, 1989), with a reported concentration of 100 mg/kg HCX in the hexachlorophene. It is also reported that hexachlorophene is manufactured from 2,4,5-trichlorophenol (WHO, 1989) and that 2,3,7,8-TCDD was reportedly formed during the manufacture of 2,4,5-trichlorophenol from 1,2,4,5-tetrachlorobenzene (WHO, 1989). Figure 2 provides an overview of the study area.

Site Investigations and Actions

From 1970 to 1986, the Rhode Island Department of Environmental Management (RIDEM) oversaw several investigations at the property. In November 1981, RIDEM issued a Notice of Violation and Order requiring the property owners to immediately identify all hazardous materials on site and properly dispose of all hazardous wastes. No information regarding compliance actions is provided in available file information. In February 1982, approximately 400 drums were excavated from the property and disposed of off site. Approximately 30 of the drums contained chemical residues. The residues were sampled and at least eight drums containing hazardous materials were manifested off site. In 1982, RIDEM required that soil samples be collected in the footprint of the Centredale Manor building prior to its construction. Following sampling and analysis, approximately 6,000 cubic yards of soil from the building footprint were excavated and disposed of as non-hazardous waste.

In January 1999, EPA began a time-critical removal action at the property. The removal action involved sampling; clearing; installation of fencing; a Flood Evaluation Study of the Site and surrounding area; the design and implementation of interim soil caps for specific areas of the Site; and community outreach. The time-critical removal action was completed in the spring of 2000.

Since January 1999, EPA has conducted several PRP search activities, including interviewing persons familiar with former operations at the property, performing title searches, file reviews, and issuing information request letters.

In August 1999, EPA started initial Remedial Investigation (RI) activities at the site. A pilot tree swallow study was conducted during the spring of 2000. Additional Site evaluations were performed for an ecological risk assessment and human fish consumption, and included biota testing and additional tree swallow studies.

In February 2000, EPA began a non-time critical removal action. The action was initiated when EPA signed an Approval Memorandum for the performance of two Engineering/Cost Analyses (EE/CAs). The two EECAs were later combined into one. In September 2000, an EE/CA report was issued. The EE/CA report recommended that the Allendale Dam be restored and that floodplain sediment from residential and recreational-use properties be removed.

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In the spring of 2000, EPA issued a Unilateral Administrative Order (UAO) to five potentially responsible parties, ordering them to complete time-critical removal activities. All five parties complied with the Order, and their Completion of Work report was approved by EPA in September 2000.

In the fall of 2000, EPA began a source area investigation. As part of the investigation, monitoring wells were installed, and soil, groundwater, and surface and downhole geophysical data were collected. The source area investigation was completed in the summer of 2001.

On January 18, 2001, EPA issued an Action Memorandum approving the non-time critical removal action recommended in the September 2000 EE/CA report.

In March 2001, EPA issued a second UAO to the same five potentially responsible parties that were issued a UAO in 2000. The five parties were ordered to implement the non-time critical removal action approved by EPA on January 18, 2001.

In the summer of 2001, the non-time critical removal action activities were started. These activities included the reconstruction of the Allendale Dam and the restoration of the Allendale Pond. The removal action activities were completed in February 2002.

In March 2002, EPA notified 11 additional parties of their potential responsibility with regard to the site.

In the spring and summer of 2002, additional soil samples were collected from residential and recreational-use properties along the Woonasquatucket River. Removal of contaminated soil from these properties was completed in the winter of 2002 and final restoration activities were completed in the fall of 2003.

In the summer and fall of 2002, EPA performed additional groundwater and soil testing and conducted geomorphic and geophysical studies of the Woonasquatucket River. In the spring of 2003, EPA conducted a sediment core investigation, involving dioxin analyses and age-dating of the cores. The data is currently being evaluated.

In September 2002, EPA entered into a third Administrative Order on Consent with ten PRPs who agreed to implement and finance another time-critical removal action within the former tailrace. In October 2002, EPA issued a UAO to two additional companies, ordering them to participate in the removal action.

In the fall of 2003, EPA began implementing the time-critical removal action, which involved designing and building a third soil cap as part of the reconstructed tailrace. This removal action has been completed.

Therefore, three caps have been constructed at areas known as 2072 and 2074 Smith Street (source area). These caps were constructed to eliminate direct contact soil exposures and to eliminate migration of soil during storm events. The caps are referred to as cap #1 and cap #2 and the two-foot cap over the sluiceway area.

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1.3 EXPOSURE AREAS

As discussed in the Work Plan and FSP (Harding ESE, 2001a,b), the study area was segregated into general exposure areas (EAs) as defined by the four historical impoundments in the Woonasquatucket River (see Figure 2). EA1 consists of the Centredale Reach portion of the Woonasquatucket River, the Allendale Pond floodplains, and the Allendale Pond channel. The Lyman Mill reach of the Woonasquatucket River and the Lyman Mill Pond represent EA2. Manton Pond is EA3 and the Dyerville reach of the Woonasquatucket River and Dyerville Pond constitute EA4. The Fogarty Center is located at 220 Woonasquatucket Avenue, which is on the eastern shore of Lyman Mill pond. The Fogarty center is a private, non-profit organization that provides services to adults with developmental disabilities. The background sample collection area include the Woonasquatucket River upstream of the Site (from Route 44 to and including Greystone Mill Pond). The background area is referred to as Greystone Mill Pond or the background area throughout this report. A reference sample collection area remote from the Woonasquatucket River includes Assapumpset Pond and Brook which are located to the west of the Woonasquatucket River and which flow into Lyman Mill Pond. This reference area is referred to as Assapumpset Pond and Brook or the reference area throughout this report.

The background location was identified based on the following criteria and considerations: the background location is not impacted by the Superfund Site under study or any other Superfund Site; the background location has the same basic physical and habitat characteristics as the study area; and the location should reflect any upstream impacts that may be affecting the study area. In this case, the background area includes the area of the Woonasquatucket River upstream of the site, from Route 44 north, to and including Greystone Mill Pond. There were no identified migration pathways linking site contaminants to that area. Greystone Mill Pond is likely affected by the discharge of the Smithfield Wastewater Treatment Plant and the impacts of that wastewater treatment plant likely extend into at least some portion of the site. In the BHHRA, Greystone Mill Pond is considered the most appropriate comparison location for the purposes of determining site-related incremental risk because Greystone is a riverine environment directly upstream of the study area. Differences between the background area and the Site would generally be expected to be associated with Site-related activities.

The reference area, Assapumpset Pond and Brook was selected for characterization because the Pond and Brook are tributaries to Lyman Mill Pond. The pond and brook are upstream of the lower portion of the Site (Lyman Mill, Manton, and Dyerville Ponds). The pond and brook carry considerable flow from an area of open space upgradient and west of the site. The reference area was characterized in order to assess possible sources of contamination to Lyman Mill Pond and downstream areas in the event that Lyman Mill Pond and downstream areas contaminant characteristics that differed from those of Allendale Pond, which is located adjacent to and immediately downstream of the source area. The site investigation and the risk assessment did not identify any likely input of contaminants from Assapumpset Pond to the Site. In the BHHRA, human health risks have been characterized to provide additional context for the Site and background risks. However, the incremental risks for the Site have been identified as the difference between Site risks and the risks at the background area. The reference area human health risks have not been considered in the calculation of Site-related Incremental Risk above background.

In this BHHRA, the term "exposure point" has been used to identify locations or areas of exposure. The exposure points correspond to the exposure areas identified above as follows:

- EA1 is referred to as the Allendale Pond exposure point;
- EA2 is referred to as the Lyman Mill Pond exposure point;
- EA3 is referred to as the Manton Pond exposure point;
- EA4 is referred to as the Dyerville Pond exposure point;
- The Fogarty Center is a specific exposure point on the eastern shore of Lyman Mill Pond;
- The upstream background area is referred to as the Greystone Mill Pond area exposure point; and
- The reference area is referred to as the Assapumpset Brook and Pond exposure point.

1.4 CURRENT AND REASONABLY FORESEEABLE USES OF THE SITE

The northern portion of the Site is occupied by the Brook Village and Centredale Manor apartment complexes. These parcels are currently occupied and covered by buildings, pavement, or landscaping. The remaining portions of the Site consist of reaches and wetlands associated with the Woonasquatucket River. The river is not used as a source of drinking water. Per the Rhode Island Department of Environmental Management (RIDEM) Water Quality Regulations EVM 112-88.97-1 (June 23, 2000), the Woonasquatucket River (from Esmond Mill Drive in Smithfield to the CSO Outfall at Glenbridge Avenue in Providence) is classified as a Class B1 water body. Class B1 water bodies are:

> "designated for primary and secondary contact recreational activities and fish and wildlife habitat. They shall be suitable for compatible industrial processes and cooling, hydropower, aqua cultural uses, navigation, and irrigation and other agricultural uses. These waters shall have good aesthetic value. Primary contact recreational activities may be impacted due to pathogens from approved wastewater discharges."

The land-use for the northern portion of the Site is expected to remain multi-family residential. The land-use for the eastern shore of Allendale and Lyman Mill reaches is primarily residential with some commercial and industrial activity. There is a power transmission line that runs along the western shore of Allendale and Lyman Mill reaches. That area has potential future use as a bike path. The bank is steep and high in most areas along the western shore of Allendale Pond and Lyman Mill Pond. There are residential, commercial, and industrial land uses approximately 200 feet or more from the western shore. These areas are shown in aerial photographs in Figure 2.

The Fogarty Center at 220 Woonasquatucket Avenue provides services to adults with developmental disabilities. The maintained lawn area between the single-story building and the Woonasquatucket River may be subjected to flooding during periods of high water. The area is surrounded by a chain link fence and the lawn is maintained. The lawn area is used for leisure activities, including cook-outs and other activities.

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1.5 SUMMARY OF ENVIRONMENTAL CONDITIONS

Sampling activities conducted by USEPA and RIDEM revealed elevated polychlorinated dibenzo dioxins and furans (dioxins and furans) in soils and sediments as well as in fish taken from Woonasquatucket River. As mentioned above, other contaminants detected onsite include: Polychlorinated biphenyls (PCBs), chlorinated and aromatic volatile organic compounds (VOCs), polycyclic aromatic hydrocarbons (PAHs), phthalates, and metals. The Site was added to the National Priorities List on February 4, 2000. For the Woonasquatucket River, there is currently a fish consumption advisory in place that recommends that people not eat fish, eels, turtles, or plants from the river downstream of the Smithfield Treatment Plant. Greystone Mill Pond, Allendale Pond, Lyman Mill Pond, Manton Pond, and Dyerville Pond are within the area subject to the advisory. Assapumpset Pond and brook are not within the area of the advisory.

Approximately 400 drums and 6,000 cubic yards of soil were removed from the property during construction of the apartment complexes. However, the exact locations of these remediation activities are not known. Temporary caps were installed over heavily contaminated areas near residences. Two soil caps were placed in the source area in July 2000.

Temporary fencing was erected around areas of contaminated surface soil in January 1999. The temporary fencing was replaced with chain-link fence between May and September 1999 to prevent access to contaminated areas.

Allendale Dam was partially breached in 1991, allowing the water level in Allendale Pond to recede. As a result, most of the pond bottoms adjacent to residential properties along the eastern bank of Allendale Pond were exposed and much of the area became vegetated. USEPA planned to reconstruct Allendale Dam during the summer of 2001, thus restoring Allendale Pond. In May 2001, prior to any reconstruction efforts, two successive breaches occurred at Allendale Dam. This likely resulted in the migration of contaminated sediment to Lyman Mill Pond and a substantial lowering of Allendale Pond water. Since then, a new dam has been installed and water levels in Allendale Pond are back to levels that existed prior to the 1991 breach.

The residential soils and sediments along the eastern shore of Allendale Pond and Lyman Mill Pond have been the subject of previous investigations and the need for remedial activities associated with those soils and sediments has been evaluated as part of the Non-Time-Critical Removal Action (NTCRA) for those areas. The details of the NTCRA objectives were identified in the *Request for Removal Action Centredale Manor Restoration Project, North Providence, Rhode Island-Action Memorandum – Non-Time-Critical Removal Action (NTCRA)* dated January 18, 2001 (USEPA, 2001a). In 2002, excavation of soils and sediment was conducted as part of the NTCRA. Based on the post excavation figures provided by USEPA, several environmental samples are no longer representative of Site conditions and have not been considered in this BHHRA. Table A-1 of Appendix A identifies all environmental samples utilized in the BHHRA. Table A-2 of Appendix A identifies soil and sediment sample locations that were not considered in the BHHRA for various reasons. Those samples removed due to NTCRA excavation activities are identified in Table A-2 as "Excavated" and "NTCRA". All residential soils that were evaluated as part of the NTCRA were also not considered in this BHHRA. Those sample locations are identified as "Not evaluated" and "NTCRA" in Table A-2.

Numerous soil sample locations no longer represent potential soil exposure opportunities for potential receptors at the Site. Soil sample locations within the footprint of the two caps (#1 and #2) and the rip-rap along the eastern bank of the Woonasquatucket River adjacent to and downstream of the source area are no longer representative of soils with current exposure potential. The caps, rip-rap, and existing paved areas will be evaluated as part of the permanent remedy for the source area. These contaminated soil areas and a need for caps were evaluated as part of the EPA Region I Action memorandum, dated May 4, 1999, as amended September 13, 1999 and June 1, 2000, and were addressed under the Time-Critical Removal Action (TCRA) in 2000. Therefore, soil samples from those areas have not been considered in this risk assessment. Soil sample locations from these areas that are not considered in the BHHRA are identified in Table A-2 as "Allendale" and either "Cap #1", "Cap #2", or "Rip-Rap.

In addition, the drainage swale area located to the east of Centredale Manor has been remediated. The storm water that typically flows through this area has been diverted and the area has been covered with a two-foot thick cap. Therefore, samples collected from within this area will not be representative of potential soil, surface water, or sediment exposure in the future and are not considered in this BHHRA. Table A-2 identifies those sample locations as "Allendale" and "2 ft Cap – Sluiceway".

As indicated above, due to the breach of the Allendale Dam (in 1991 and in May 2001), water levels within Allendale reach have fluctuated, and have generally been lower than they were prior to 1991. Many of the soil samples collected from the Allendale Reach were collected at a time when the water levels were uncharacteristically low due to the breach of the Allendale Dam. The Allendale Dam was restored in 2002. As indicated in the Engineering Evaluation/Cost Analysis (EE/CA) (Tetra-Tech, 2000), the water level in Allendale with the dam restored would be 93.5 feet above sea level. Therefore, the typical water level in the Allendale Reach is expected to be 93.5 feet. Numerous soil samples have been identified that were collected from areas that are now, based on a water elevation of 93.5 feet, actually representative of sediments. Figure 3-1 of the EE/CA was used to identify the extent of area within Allendale Reach that is below elevation 93.5. All "soil" samples collected from that area have been treated as "sediment" for the purposes of the BHHRA because the locations of those samples are typically below water. Table A-2 in Appendix A documents those "soil" samples that are now "sediments" because the water level in Allendale has been restored to historical levels.

1.6 CONCEPTUAL SITE MODEL

The conceptual site model (CSM) identifies potential source areas from which oil and/or hazardous materials (OHM) may have been released, the migration pathways through which OHM may have been transported and/or translocated to other environmental media, and where possible exposure may occur. The CSM provides a framework for understanding sources of OHM, migration pathways, identification of potential receptors, and development of exposure profiles. The CSM for the Site was developed as part of the Work Plan and is also presented here. The CSM is presented graphically in Figure 3.

1.6.1 Source Area

Releases of hazardous substances from former industrial operations have occurred at the Site. The source area consists of two parcels located at 2072 and 2074 Smith Street (Lots 200 & 250)

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that cover approximately 9 acres (see Figure 2). Evidence suggests that operations at the former chemical company and drum reconditioning facility resulted in waste disposal onto surface soil and beneath the ground surface. Wastes have also been released directly into the Woonasquatucket River, which runs along the western side of the source area (Tetra Tech NUS Inc., 2000). Dioxins and furans have been detected in soils and sediments as well as in fish tissue collected in 1996 from the Woonasquatucket River. Other contaminants detected in Site media include PCBs, VOCs, PAHs, hexachloroxanthene (HCX), phthalates, and metals.

1.6.2 Migration of OHM

Downstream sediments have been impacted through the transport and deposition of contaminants from the source area. Dioxins and furans have been detected in sediments at Allendale Pond and Allendale Dam, which is approximately 2000 feet from the source area. With the partial breaching of Allendale Dam in 1991 and the more recent breach in 2001, contaminants have migrated downriver to Lyman Mill Pond. Evidence suggests that contaminants originating from the source area have migrated beyond Manton Dam down to Dyerville Pond (Figure 2 shows the location of these features).

1.6.3 Potential Human Receptors

Previous risk assessment activities have identified and evaluated certain current and/or future human receptors. These receptors include residents of 2072 and 2074 Smith Street as well as residents who live along the eastern shore of Allendale Pond and Lyman Mill Pond. As shown in Table 1.1, these receptors and the specified exposure pathways have been evaluated previously and are not included in the BHHRA. In summary, potential exposures to surface and subsurface soil, groundwater, potential vapors, and waste materials for residents of the Centredale Manor Apartments and the Brook Village Apartments were previously evaluated and measures have been taken to mitigate and/or eliminate those exposures. Residential exposures to bank soil, shallow sediments, and surface soil at residential properties on the shore of Allendale Pond and Lyman Mill Pond have also been evaluated and actions have been taken to mitigate and/or eliminate those exposures. The potential receptors for the BHHRA are discussed in detail below.

Table 1.1 indicates which receptors and exposure pathways are to be evaluated per this work plan. Certain other receptors and exposure pathways were also considered, but were not selected for inclusion in this work plan for the BHHRA for the reasons described below:

- Occupants of Centredale Manor and Brook Village Apartments, potential direct contact and inhalation of dust associated with surface soils. This scenario was qualitatively evaluated in the *Draft Health Consultation* (ATSDR, 1999). That evaluation led to the installation of Cap #1 and Cap #2. Most of the impacted soil is beneath one of those caps or the paved roadways and parking lots at the property. It is understood that these caps and the pavement will be included as components of the permanent remedy. Therefore, potential exposure to those soils is not foreseeable. Construction-related exposure to those soils is also not foreseeable, since institutional controls will be part of the permanent remedy. Those controls will maintain the integrity of the barriers to soil exposure.
- Occupants of Centredale Manor and Brook Village Apartments, potential inhalation of VOCs that might potentially migrate from groundwater or subsurface soil into buildings.

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There has been indoor air monitoring conducted by OEME in both apartment buildings. Report indicates that no significant migration of VOCs into buildings was found.

- Residents along the eastern shore of Allendale Reach and Lyman Mill Pond, potential direct contact with surficial soils at residential properties within the floodplain of the river. This pathway has been addressed by the Non-Time-Critical Removal Action (NTCRA) as described in the USEPA NTCRA Action Memorandum dated January 18, 2001.
- Residents along the eastern shore of Allendale Reach and Lyman Mill Pond, potential inhalation of VOCs potentially migrating from groundwater or surface soil into buildings. Investigations have not identified Site-related VOC contamination in groundwater or subsurface soil in that area. This pathway does not require further evaluation.
- Residents of Centredale Manor Apartments and Brookside Village Apartments as well as residents along the eastern shore of Allendale Reach and Lyman Mill Pond, potential ingestion, dermal, and inhalation exposure associated with the use of groundwater. Groundwater in the area is not used for any potable purpose. This pathway does not require further evaluation.
- Commercial/industrial workers along western shore of Allendale and Lyman Mill Reaches, potential direct contact exposure to floodplain soils on commercial/industrial properties. The bank along the western shore is high (substantially more than 10 feet high in most areas) and steep. There is no indication that impacted sediments would have been transported to the commercial/industrial properties to the west of the river. There is no indication this pathway is complete and no further evaluation is required.
- Commercial/industrial workers along western shore of Allendale and Lyman Mill Reaches, potential inhalation of VOCs potentially migrating from groundwater or surface soil into buildings. Investigations have not identified Site-related VOC contamination in groundwater or subsurface soil in that area. This pathway does not require further evaluation.

Consistent with USEPA objectives, the following pathways are evaluated for the BHHRA as summarized in Table 1.2:

1) potential exposure to chemicals of potential concern (COPCs) via ingestion of fish. Contaminants that are present in surface water and aquatic sediments may have bioaccumulated in fish and other biota present in the Woonasquatucket River. These fish may be consumed by individuals that catch and/or consume biota from the river. Child, older child, and adult consumers may be exposed to COPCs via ingestion of fish and other biota. Therefore, recreational anglers/consumers and residents living along the river are evaluated in the BHHRA, focusing on three age groups that include the child (ages 1 through 6), the older child (ages 7 through 18), and the adult (19 through 30 for the resident and 19 through 70 for the visiting recreational angler). In addition, a supplementary evaluation focused on potential visiting subsistence anglers/consumers is presented in Appendix F because of the uncertainty associated with that scenario. 2) potential exposure COPCs via incidental ingestion and dermal contact with surface water, sediment, and bank soil. Neighborhood residents involved in recreational angling as well as wading and swimming may be exposed to these media. In addition, visitors to the Site involved in recreational or subsistence angling could also be exposed to these media while wading. Adult workers (25 year exposure) at the Fogarty Center may also be exposed via incidental ingestion of and dermal contact with surface soils at that property

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that are within the floodplain of the Woonasquatucket River. Table 1.2 identifies the receptors and exposure pathways evaluated in the BHHRA.

1.7 SUMMARY OF DATA

In the Final Data Evaluation Report (DER) (MACTEC, 2003) historical analytical data for biota collected in the Woonasquatucket River, Site sediments, surface water, and soil were compiled and reviewed. The DER also presents and summarizes the biota, sediment, surface water, and soils data collected during the 2001 Spring and Summer BCRA/BERA investigation activities in and adjacent to the Woonasquatucket River and at Assapumpset Brook and Pond.

The historical data (primarily reported in *Final Technical Memorandum, Woonasquatucket River Sediment Investigation, Centredale Manor Restoration Project Site, North Providence, Rhode Island* (Tetra Tech NUS, Inc., 2000a) and the *Final Engineering Evaluation/Cost Analysis, Centredale Manor Restoration Project Site, North Providence, Rhode Island* (Tetra Tech NUS, Inc., September, 2000b), data from the 2001 BCRA/BERA investigation, and data collected to support ecological risk assessment activities have been considered for use in this BHHRA.

The analytical data generated for sediment, surface water, and biota samples collected in 2001 to support the BCRA and BERA are generally consistent with the analytical data generated for these media in previous investigations. The analytical data generated from the 2001 investigations indicate that dioxins and furans (particularly 2,3,7,8-TCDD), HCX, PCBs (particularly Aroclor 1254 and some coplanar PCB congeners), semi-volatile compounds (including PAHs), metals, and pesticides were detected in sediments within Allendale Pond, Lyman Mill Pond, Manton Reach, and Dyerville Reach. The 2001 investigation also identified dioxins and furans (particularly 2,3,7,8-TCDD), HCX, semi-volatile compounds (including PAHs), metals, and pesticides in biota collected from Allendale Pond, Lyman Mill Pond, Manton Reach, and Dyerville Reach. For surface water, the 2001 investigation identified metals and inorganics, semi-volatile compounds (including PAHs), and some pesticides. No Aroclors were reported for 2001 surface water samples. The 2001 surface water samples were not analyzed for dioxins and furans.

The 2001 investigation results are consistent with the results of historical investigations and the reported releases at the Site (described in section 1.2 above) as well as with the information in the more recent investigation reports from 2003 and 2004: *Task 22H Chemistry Data Report, YR2002 Tree Swallow Study, Centredale Manor Restoration Project Superfund Site, North Providence, Rhode Island* (Battelle, 2003c); *Data Summary Report, Interim Data Collection, Remedial Investigation and Feasibility Study, Centredale Manor Restoration Project Superfund Site, North Providence, Rhode Island* (Battelle, 2003d); *Task RI-8, Petroleum Hydrocarbon Assessment of Centredale Sediment Cores, Centredale Manor Restoration Project Superfund Site, North Providence, Rhode Island* (Battelle, 2003e); and *Task 22I Chemistry Data Report, YR2003 Tree Swallow Study, Centredale Manor Restoration Project Superfund Site, North Providence, Rhode Island* (Battelle, 2004).

The BHHRA considered 6 sediment samples for Assapumpset Pond and Brook, 12 sediment samples for Greystone Mill Pond, 164 sediment samples from Allendale Pond reach, 49 sediment samples from Lyman Mill Pond, 3 sediment samples from Manton Pond and 3 sediment samples from Dyerville Pond. For surface water, the BHHRA considered 3 samples from Assapumpset

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Pond and brook, 9 samples from Greystone Mill Pond, 43 samples from Allendale Pond, and 29 samples from Lyman Mill Pond. For bank soil, the BHHRA considered 4 samples from Greystone Mill Pond and 20 samples (only 2,3,7,8-TCDD) from Allendale Pond.

The evaluation of fish consumption exposures and risks is based on the analytical data collected for the following numbers of fish samples:

- Assapumpset Pond and Brook – 3 brown bullhead, 4 largemouth bass (fillet), 6 American eel;
- Greystone Mill Pond – 10 white sucker, 10 largemouth bass (fillet), 10 American eel;
- Allendale Pond – 10 white sucker, 10 American eel;
- Lyman Mill Pond – 10 white sucker, 10 largemouth bass (fillet), 10 American eel;
- Manton Pond - 3 largemouth bass (fillet);
- Dyerville Pond – 3 American eel

PCB congener analysis was conducted for a subset of the fish samples that were collected as shown below. The species and numbers of samples analyzed for PCB congeners were:

- Assapumpset Pond and Brook – no fish tissue samples were analyzed for PCB congeners;
- Greystone Mill Pond – 1 white sucker, 1 largemouth bass (fillet), 1 American eel;
- Allendale Pond – 3 white sucker, 4 American eel;
- Lyman Mill Pond – 3 white sucker, 5 largemouth bass (fillet), 3 American eel;
- Manton Pond – no fish tissue samples were analyzed for PCB congeners;
- Dyerville Pond – 1 American eel

Summaries of analytical data for the environmental media evaluated in the BHHRA are presented in Tables 2.1 through 2.14, which document the selection of chemicals of potential concern. In addition to the data summaries provided in the DER and in Tables 2.1 through 2.14, a graphical presentation of data is useful in identifying potential EAs that exhibit analytical parameters that have elevated frequency of detection and/or elevated concentrations relative to conditions at a background area and a reference area that are expected to be free of potential impacts from the Site. The upstream background area (Greystone Mill Pond) and the reference area (Assapumpset Brook and Pond), have been the subject of recent investigation activities. Five potential EAs have previously been identified for the Site (Allendale Reach, Lyman Mill Reach, Manton Reach, Dyerville Reach and the Fogarty Center). The order of presentation of reference area, the background area, and potential EAs above is consistent with the physical locations of the areas. Assapumpset Brook and Pond are not part of the Woonasquatucket River itself, but is a tributary to the river. The brook and pond are removed from the Site. Greystone Mill Pond is upstream of the Site on the Woonasquatucket River (an upstream location). Sediment and soil samples were collected just north of the source area (north of Route 44). These sample locations are considered to be part of the upstream (Greystone Mill Pond) data set. Greystone Mill Pond is the furthest upstream of the source area and Dyerville Reach is the most downstream area from the source area. Allendale Reach is both adjacent to and immediately downstream of the source area. The graphical presentation of Site data is intended to provide perspective with respect to the nature and location of likely Site-related impacts. This graphical presentation is not used to screen out or to otherwise adjust the list of analytes that are considered in the risk assessment. The selection of COPCs for the risk assessment is based on all of the available environmental data that are representative of current and future conditions.

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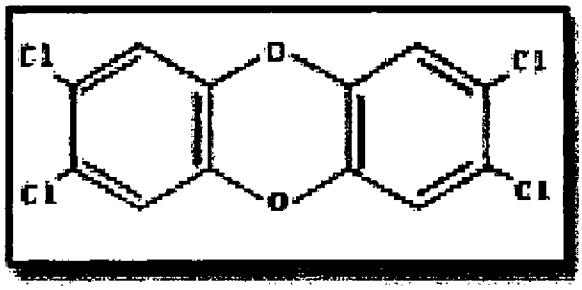
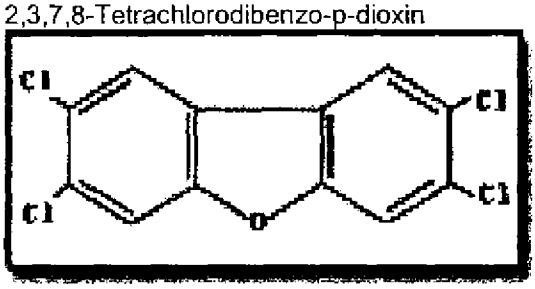
Figures 4 through 12 present histogram summaries of dioxin/furan, HCX, and co-planar PCB equivalent concentrations (TEQs) in sediment, surface water, floodplain and bank soil, potentially edible biota (white sucker (brown bullhead instead of white sucker in Assapumpset Pond only), American eel, and largemouth bass) as well as crayfish, emerging insects, and earthworms. The TEQs are media-specific concentrations that are normalized to the toxicity of the 2,3,7,8-TCDD congener, generally considered to be the most toxic of the dioxin, furan, and dioxin-like compounds. The TEQs are calculated by multiplying the medium-specific concentration of each congener or congener group by a Toxicity Equivalence Factor (TEF) and summing those products. The TEF is a measure of the toxicity of a particular congener or congener group relative to toxicity of 2,3,7,8-TCDD. In simple terms, the dioxins/furans TEQ indicates the concentration of 2,3,7,8-TCDD that would have the same toxicity as the mixture of dioxins and furans being evaluated.

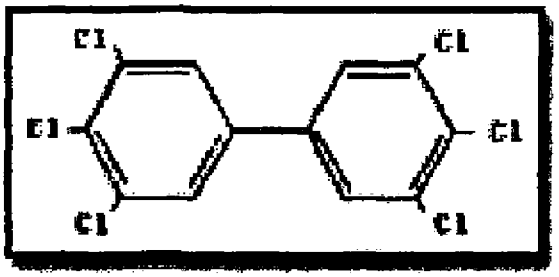


2,3,7,8-Tetrachlorodibenzo-p-dioxin



2,3,7,8-Tetrachlorodibenzofuran

Biphenyl is a dual-ring structure comprised of two six-carbon benzene rings joined by a single carbon-carbon bond. Up to ten chlorine atoms can be substituted for hydrogen atoms in the biphenyl molecule. Each of the carbon atoms in the benzene rings is assigned a location number between 1 and 6. The carbon atoms assigned the location 1 are bonded to each other and are not available for chlorine substitution. Each unique chemical compound within the PCB category is referred to a congener. Therefore, the biphenyl molecule containing two chlorine atoms (each located at the "4" position of one of the benzene rings), would be a PCB congener referred to as 4,4'-Dichlorobiphenyl. A total of 209 PCB congeners have been identified. A biphenyl with one chlorine atom is referred to as a monochlorobiphenyl and a biphenyl with ten chlorine atom is referred to as a decachlorobiphenyl. Homologs are subcategories of PCB congeners having equal numbers of chlorine atoms. For example, there are 12 PCB congeners that have two chlorine atoms. These 12 congeners are included in the dichlorobiphenyl homolog.



3,3',4,4',5,5'-Hexachlorobiphenyl

Commercially produced PCB mixtures were sold under many names. However, the most common naming convention for commercial PCB mixtures was reference to the Aroclor series. Aroclors are mixtures of various chlorinated biphenyls. The last two digits in the Aroclor identifier indicates the percentage of the mixture represented by chlorine. Aroclor-1242 is a mixture of chlorobiphenyls with a chlorine content of 42%. Aroclor-1254 is a mixture of chlorobiphenyls with a chlorine content of approximately 54 %. Typically, the higher the chlorine content, the greater the abundance of the heavier chlorinated biphenyls (such as pentachlorobiphenyls and hexabiphenyls).

During investigations of the Site, analysis of PCBs has been completed by two different analytical approaches. The first, and most frequently applied approach (for the large majority of samples) at the Site is the analysis for Aroclors via Method 8082. In this analysis, the following analytical parameters are typically reported: Aroclor-1016; Aroclor-1221; Aroclor-1232; Aroclor-1242; Aroclor-1248; Aroclor-1254; Aroclor-1260; and Aroclor-1268. The second approach, identification and quantification of individual PCB congeners, was used less frequently, with only a few representative samples per area, at the Site. The identification of individual PCB congeners was accomplished by a modified Method 1668A. Although each of the 209 PCB congeners has a unique chemical name (such as 4,4'-Dichlorobiphenyl), a shorthand means of identifying the individual congeners has been developed. Each of the congeners has been assigned a unique number from 1 to 209 (Ballschmiter, 1992). The numbering scheme assigns lower numbers to lower chlorine content congeners and higher numbers to higher chlorine content congeners. As an example, the PCB congener 3,4',5-trichlorobiphenyl is also referred to as PCB-39.

A total of 68 of the PCB congeners, based on their chemical structure, have been identified as "dioxin-like" or co-planar PCB congeners. These co-planar PCB congeners have been assigned 2,3,7,8-TCDD TEFs in a manner similar to the dioxin and furan congeners (Van den Berg et. al. 1998). A TEQ for all co-planar PCBs has been calculated for each sample of fish tissue and sediment that has been analyzed for PCB congeners. PCB congener analysis was not conducted for any surface water samples or bank soil samples. Among sediment samples used in the risk assessment, PCB congener analysis was conducted for 1 sample in Greystone Mill Pond, 1 sample in Allendale Pond, 3 samples in Lyman Mill Pond, and 1 sample in Dyerville Pond. No sediment samples from Assapumpset Pond and Brook, Manton Pond, or Dyerville Pond that were used in the risk assessment were analyzed for PCB congeners. No fish tissue samples used in the risk assessment from Assapumpset Pond and Brook or Manton Pond were analyzed for PCB congeners. One American eel sample, 1 largemouth bass fillet sample, and 1 white sucker tissue sample used in the risk assessment that were collected from Greystone Mill Pond were analyzed for PCB congeners. Among fish tissue samples from Allendale Pond that used in the risk

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assessment, PCB congener analysis was conducted for 3 American eel samples and 3 white sucker samples. Three American eel samples, five largemouth bass fillet samples and 3 white sucker tissue samples used in the risk assessment that were collected from Lyman Mill Pond were analyzed for PCB congeners. An American Eel sample used in the risk assessment that was collected from Dyerville Pond was analyzed for PCB congeners.

The TEFs used in the development of Figures 4 through 12 are the mammalian TEFs for dioxins, furans, and dioxin-like (coplanar) PCBs as published in Van den Berg et al., 1998. The application of the mammalian TEFs in the histograms is intended to provide a consistent means of presenting the nature and extent of contamination. It is not intended to imply that mammalian TEFs are appropriate for estimating risks for ecological receptors. In the Baseline Ecological Risk Assessment (BERA), TEFs for birds and fish are also employed as appropriate in evaluating risks.

Each of the figures presents, for a single medium, average concentrations of each of these three analyte groups for the upstream background area (Greystone Mill Pond), the reference area (Assapumpset Brook and Pond), and for potential exposure areas EA1 (Allendale Reach), EA2 (Lyman Mill Reach), EA3 (Manton Reach), and EA4 (Dyerville Reach) in that sequence. Each histogram presents a graphical representation of concentrations at remote, upstream, near-source, and downstream locations (from left to right). All of the data in Figures 4 through 12 are presented in picograms per gram (or parts per trillion (ppt) so that data may be compared directly across media. All of the concentration data in Figures 4 through 12 are presented on a logarithmic scale – therefore, values that are separated by a major tick-mark on the y-axis have concentrations that differ by a factor of ten, those separated by two tick-marks would differ by a factor of one hundred. Several conclusions may be reached from Figures 4 through 12:

- In sediment (Figure 4), dioxin/furan 2,3,7,8-TCDD equivalent concentrations and HCX 2,3,7,8-TCDD equivalent concentrations are at least one hundred times higher in Allendale Reach than in the background area and reference area and there appears to be a decreasing trend in concentrations moving downstream from Allendale Reach. This suggests a Site-related impact on dioxins/furans and HCX in sediments in areas adjacent to and downstream of the source area.

- In sediment (Figure 4), no specific spatial trend for co-planar PCB 2,3,7,8-TCDD equivalent concentrations was observed for sediment. This would suggest there is not a Site-related impact on concentrations of co-planar PCBs in sediment in areas adjacent to and downgradient of the source area.

- Dioxin/furan 2,3,7,8-TCDD equivalent concentrations in sediment in all areas tested are typically five-hundred to one-thousand times higher than the corresponding HCX 2,3,7,8-TCDD equivalent concentrations (Figure 4).

- In surface water (Figure 5), dioxin/furan 2,3,7,8-TCDD equivalent concentrations are approximately one hundred times higher in Allendale and approximately ten times higher in Lyman Mill than in the Greystone Mill Pond background area.

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- Dioxin/furan 2,3,7,8-TCDD equivalent concentrations in floodplain and bank soil in Allendale Pond and in Lyman Mill Pond are approximately ten times and eight times higher, respectively, than in Greystone Mill Pond (Figure 6).

- In white sucker (Figure 7), American eel (Figure 8), and largemouth bass (Figure 9) (no data for Allendale for largemouth bass), dioxin/furan 2,3,7,8-TCDD equivalent concentrations are generally ten to one-hundred times higher in Allendale and Lyman Mill Reaches than in the background area and the reference area. These average 2,3,7,8-TCDD equivalent concentrations are approximately 100 pg/g or 100 ppt. HCX was detected in white sucker and American eel in Allendale and Lyman Mill Reaches but not in the background or reference area. The 2,3,7,8-TCDD equivalent concentrations for HCX are equal to or less than 0.01 ppt. Dioxin/furan 2,3,7,8-TCDD equivalent concentrations in white sucker, American eel, and largemouth bass (no data for largemouth bass in Allendale) in Allendale and Lyman Mill Reaches are generally higher than either the HCX 2,3,7,8-TCDD equivalent concentrations or the co-planar PCB 2,3,7,8-TCDD equivalent concentrations in those same reaches. The data suggest a Site-related impact on dioxin/furan and HCX concentrations in fish in Allendale and Lyman Mill Pond.

- The average concentrations of co-planar PCB 2,3,7,8-TCDD equivalent concentrations in white sucker (Figure 7), American eel (Figure 8), largemouth bass (Figure 9), crayfish (Figure 10), and emerging insects (Figure 11) from Allendale and/or Lyman Mill Pond are higher than the corresponding average concentration in Greystone Mill Pond. These data suggest a possible Site-related impact on co-planar PCB 2,3,7,8-TCDD equivalent concentrations in the three fish species, crayfish, and emerging insects from Allendale and/or Lyman Mill Pond.

- In crayfish (Figure 10), average dioxin/furan 2,3,7,8-TCDD equivalent concentrations in Allendale and Lyman Mill Pond are approximately thirty to fifty times higher than the corresponding average in Greystone Mill Pond reference area. Average concentrations of HCX 2,3,7,8-TCDD equivalent concentrations in Allendale and Lyman Mill are approximately five to eight times higher than the corresponding average concentration in Greystone Mill Pond reference area. These data suggest a Site-related impact on dioxins/furans and HCX concentrations in crayfish in Allendale and Lyman Mill Pond.

Figures 13 through 21 present select dioxin and co-planar PCB congener concentrations (raw data, not adjusted to 2,3,7,8-TCDD equivalents) for sediment, surface water, floodplain and bank soil, white sucker, American eel, largemouth bass, crayfish, emerging insects, and earthworms. The purpose of these figures is to compare and contrast the dioxin/furan "signature" and co-planar PCB "signature" among the reference areas and the potential EAs. The "signature" is determined by which congener(s) has the highest concentrations and how the concentrations of specific congeners compare to each other.

For example, in Figure 13, in the Assapumpset reference area, the four dioxin/furan congeners with the highest sediment concentrations are, (in descending order), octa-chlorinated dibenzodioxin, 1,2,3,4,6,7,8-hepta chlorinated dibenzodioxin, octa-chlorinated dibenzofuran, and 1,2,3,4,6,7,8-hepta chlorinated dibenzofuran. The 2,3,7,8-TCDD congener (i.e., most important from a toxicological perspective) concentration is only about 0.01 times the concentration of octa-chlorinated dibenzo dioxin, the congener with the highest concentration. In contrast, for

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Allendale Reach, the 2,3,7,8-TCDD congener has the highest average concentration (approximately 300 times higher than in Assapumpset). Allendale has a similar distribution of the hepta- and octa-chlorinated dibenzodioxins and dibenzofurans as does Assapumpset. The following conclusions may be drawn from Figures 13 through 21.

- There is a pronounced difference in dioxin/furan "signature" between the reference areas and Allendale, Lyman Mill, and Manton (for those media with congener data) for sediment (Figure 13), floodplain and bank soil (Figure 15), white sucker (Figure 16), American eel (Figure 17), largemouth bass (Figure 18), crayfish (Figure 19), emerging insects (Figure 20), and earthworms (Figure 21). In virtually all cases, the average 2,3,7,8-TCDD concentrations are at least 100 times higher in Allendale and Lyman Mill Reaches than in the two reference locations. This suggests a Site-related impact on 2,3,7,8-TCDD concentrations for all of the media evaluated for Allendale, Lyman Mill Pond, and Manton Reach.

- It appears that the relative concentrations of the three co-planar PCBs shown on the histograms and the actual magnitude of those concentrations are reasonably consistent among the reference areas and the potential EAs for sediment (Figure 13), surface water (Figure 14), floodplain and bank soil (Figure 15), and earthworms (Figure 21). However, the concentrations of PCB-77 are approximately 10 to 15 times higher in Allendale and Lyman Mill than in Greystone Mill Pond. This suggests possible Site-related impact on co-planar PCB concentrations in sediment only.

- Among the potentially edible biota, the white sucker exhibited the highest average concentrations of 2,3,7,8-TCDD.

Figures 22 through 26 present the PAH "signature" for the reference areas and potential EAs for sediment, floodplain and bank soil, white sucker, American eel, and largemouth bass. Unlike the previous figures, the concentration scale for these figures is arithmetic or linear, rather than logarithmic scale because there is less variability (generally less than 10-fold differences rather than 100-fold or 1000-fold differences that were observed for dioxin/furan, HCX, and co-planar PCB equivalent concentrations and raw concentrations) among the PAH compounds and among concentrations at the reference locations and the potential EAs. This makes it possible to present all of the data on a single arithmetic scale.

The following conclusions can be drawn from these PAH histograms:

- In sediment (Figure 22), fluoranthene, pyrene, benzo(b)fluoranthene, and chrysene were consistently detected at the highest concentrations compounds among the PAHs. Overall, the relative distribution of compounds (signature) looks very similar for the reference areas and the potential EAs. However, there is variation in the magnitude of reported concentrations, with Assapumpset and Manton showing the lower concentrations. In general, Greystone, Allendale, and Lyman Mill Reaches have similar concentrations of PAHs (concentrations in the reach with the highest concentrations are generally less than twice the concentrations in the reach with the lower concentrations). There does not appear to be any trend that would suggest a Site-related impact on PAHs in sediment in Allendale Reach or downstream reaches.

- In surface water, PAHs were generally not detected. Therefore, no histogram is presented.

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- In floodplain soil and bank soil (Figure 23) (Greystone, Allendale, and Lyman Mill data available), fluoranthene, pyrene, and phenanthrene were consistently detected at the highest concentrations among the PAHs. Overall, the relative distribution of compounds (signature) looks very similar for the reference area and the potential EAs. One exception is the concentration of acenaphthylene in Lyman Mill Reach, which appears to be approximately ten times higher than in Allendale and Greystone Reaches. As can be seen by comparing the PAH signatures for Lyman Mill Sediment (Figure 20) and Lyman Mill floodplain (bank soil not evaluated for that reach) (Figure 23), the PAH signatures are similar and the concentrations are somewhat higher in soils (individual compounds at approximately 12 mg/kg) than in sediment (individual compounds at approximately 2.5 mg/kg).

- In fish (white sucker (Figure 24), American eel (Figure 25), and largemouth bass (Figure 26), phenanthrene, fluoranthene, and 2-methylnaphthalene were consistently detected at the highest concentrations among PAHs. Concentrations of the phenanthrene and fluoranthene vary somewhat across fish species (approximately 8 ug/kg to 22 ug/kg phenanthrene in white sucker, not detected to approximately 16 ug/kg phenanthrene in American eel, and not detected to approximately 9 ug/kg in largemouth bass. Naphthalene and 2-methylnaphthalene concentrations in all three fish species are higher in Assapumpset than in any other reach. The PAH fingerprint in fish tissue is similar to the sediment fingerprint observed, with the heavier molecular weight PAHs comprising a somewhat smaller portion of the total PAH concentration in fish than in sediment. Concentrations of PAHs in fish tissue are approximately 500 to 1000 times lower than corresponding concentrations in sediment.

- There are no specific trends that would suggest a Site-related impact on PAHs distribution or concentrations in surface water, sediment (Figure 22), floodplain and bank soil (Figure 23), white sucker (Figure 24), American eel (Figure 25), or largemouth bass (Figure 26) at or adjacent to the source area within Allendale Reach.

Figures 27 through 34 summarize average concentrations of select pesticides (DDT, DDD, DDE, dieldrin, chlordane, and heptachlor) and PCB Aroclors in sediment, surface water, floodplain and bank soil, white sucker, American eel, largemouth bass, crayfish, and earthworms. These data are presented with a logarithmic scale for concentration. The following conclusions may be drawn from these histograms.

- In sediment (Figure 27), chlordane (except in Dyerville) and the PCB Aroclors 1254 and 1268 were consistently detected at the highest concentrations among pesticides and PCBs. The signature of DDT, DDD, and DDE appears similar in sediment from all reference and potential EAs. The average sediment concentration of dieldrin in Allendale Reach appears to be approximately 4-fold higher than in the other reaches. Heptachlor seems to be unique to Allendale and Lyman Mill Reaches. In Allendale Reach, Aroclors that were not observed in other reaches are present (Aroclors 1016, 1221, 1232, 1242, and 1248). The average concentrations of Aroclor 1254 in Allendale, Lyman Mill and Dyerville reaches are each greater than the corresponding concentrations in the two reference areas. This suggests a Site-related impact on Aroclor 1254 concentrations in sediments in Allendale, Lyman Mill Pond, and Dyerville Reach.

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- For surface water (Figure 28), as would be expected with the low solubility of Aroclors and the persistent pesticides that have been evaluated, these parameters were typically not detected.

- In floodplain soil and bank soil (Figure 29), the signature and concentrations of DDT, DDD, DDE, dieldrin, chlordane, and heptachlor appear similar in sediment from all reference and potential EAs. Aroclors 1254 and 1268 are detected in Greystone and Lyman Mill (with higher concentrations in Greystone). In Allendale reach, Aroclors 1242, 1248, and 1254 are detected, all with average concentrations higher than in any of the other reaches.

- In white sucker (Figure 30), American eel (Figure 31), and largemouth bass (Figure 32), Aroclor 1254, chlordane, and Aroclor 1268 were consistently detected at the highest concentrations among pesticides and PCBs. Overall, concentrations of Aroclor 1254 and chlordane appear to be higher in Allendale and Lyman Mill Reaches than in the reference areas. The signature and concentrations of DDT, DDD, DDE, dieldrin, chlordane, and heptachlor appears similar in sediment from all reference and potential EAs. Concentrations of these pesticides are consistent in all three fish species. Chlordane concentrations are generally 100 ug/kg or above (greater than 1000 ug/kg in Lyman Mill for white sucker and largemouth bass). Aroclor 1254 concentrations are generally 100 ug/kg or above consistently (greater than 1000 ug/kg for white sucker in Allendale and Lyman Mill and for largemouth bass in Lyman Mill). The data suggest a Site-related impact on Aroclor 1254 concentrations in the three fish species in Allendale and Lyman Mill Reaches.

- In crayfish (Figure 33), chlordane and Aroclor 1254 and 1268 are detected consistently and at similar concentrations in Greystone, Allendale, and Lyman Mill Reaches (average concentrations appear lower in Assapumpset Reach). Aroclor 1254 average concentrations appear to be highest in Allendale and Lyman Mill Reaches. Concentrations of chlordane and Aroclor 1254 concentrations are roughly ten-fold lower than in fish.

- In earthworms (Figure 34), chlordane and Aroclor 1254 and 1268 are detected consistently and at similar concentrations in Greystone, Allendale, and Lyman Mill Reaches. Chlordane and Aroclor 1254 average concentrations appear to be highest in Allendale Reach. Concentrations of chlordane and Aroclor 1254 concentrations are roughly ten-fold lower than in fish.

- The predominant pesticide (chlordane) and Aroclors (1254 and 1268) detected in fish (Figures 30-32), earthworms (Figure 32), and crayfish (Figure 31) are consistent with the predominant pesticide (chlordane) and Aroclors (1254 and 1268) detected in sediments (Figure 27). The concentrations of the chlordane and Aroclors 1254 and 1268 are consistent between sediment and the three fish species.

Figures 35 through 42 summarize average concentrations of select inorganics (arsenic, cadmium, chromium, lead, and mercury) in sediment, surface water, floodplain and bank soil, white sucker, American eel, largemouth bass, crayfish, and earthworms. Overall, there is a small amount of variability in inorganics concentrations among the reference and EAs for any given medium. Therefore, the inorganics data are presented using an arithmetic scale. The following conclusions may be drawn from these inorganics histograms.

- In sediment, arsenic, cadmium, and mercury concentrations vary very little among the two reference areas and four potential EAs. Average concentrations of lead vary among the reaches, with the highest average concentrations reported for Allendale and Lyman Mill Reaches. The average concentration of lead in Allendale Reach is approximately double the concentration in Greystone Mill Pond. Chromium concentrations in sediment vary from less than 20 mg/kg in Assapumpset Brook to approximately 150 mg/kg in Dyerville reach, with no apparent spatial trend. There is little indication of a Site-related impact on inorganics (those presented in the histograms) concentrations in sediments adjacent to and downstream of the source area.

- In surface water, average concentrations of arsenic, cadmium, chromium, and mercury are consistently well below 10 ug/liter in the two reference areas and in Allendale and Lyman Mill Reaches. Average concentrations of lead in surface water are all below 5 ug/liter except in Allendale reach (approximately 12 ug/l). There is little indication of a Site-related impact on inorganics (those presented in the histograms) concentrations in surface water adjacent to and downstream of the source area.

- In floodplain soil and bank soil, there is no indication that inorganic concentrations in soils adjacent to and downstream of the source area have been impacted by the Site.

- In white sucker, American eel, and largemouth bass, average concentrations of arsenic, cadmium, chromium, lead, and mercury are all below 0.5 mg/kg in all areas tested. There are no consistent spatial trends that would suggest that fish adjacent to and downstream of the source area have been impacted by a release of the inorganics that are presented in these histograms. For the American eel, average concentrations of lead in Allendale, Lyman Mill, and Dyersville Reaches appear to be somewhat higher than those in the reference areas. However, these concentrations are very low (less than 0.5 mg/kg). No Site-related impact is indicated.

- In crayfish, all average concentrations of inorganics are below 1 mg/kg. There is one apparent spatial trend, with lead concentrations appearing to increase from Greystone to Allendale, to Lyman Mill Reach. However, the difference between the average concentration at Greystone and Lyman Mill Reaches is only approximately 0.3 mg/kg. Therefore, there is no indication of a Site-related impact on inorganics concentrations in crayfish adjacent to or downstream of the source area.

- In earthworms, there are no spatial trends that indicate that inorganic concentrations in earthworms adjacent to or downstream of the source area have been impacted by the Site.

- Overall, average concentrations of chromium and lead in fish and crayfish are less than one percent of the corresponding sediment concentrations.

- Overall, average concentrations of chromium and lead in earthworms are less than 30 percent of the corresponding soil concentrations.

In addition to the histograms discussed above, Figures 43 through 48 present a "plan-view" representation of the concentration distribution of dioxin equivalents, HCX, total PCBs, total

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PAHs, total pesticides, and metals in total toxic units in sediment and surface water. These figures summarize the nature and extent of contamination for these analytical parameters.

In summary, dioxins and furans (particularly 2,3,7,8-TCDD), HCX, Aroclor-1254and possibly PCB-77 appear to be the primary chemical parameters that are detected in environmental media with frequency of detection and concentrations that are indicative of Site-related impacts. Figure 49 indicates that sediment concentrations of 2,3,7,8-TCDD, HCX, and PCB-77 are dramatically higher in Allendale Pond, Lyman Mill Pond, Manton Reach, and Dyerville Reach than in either the upstream background area Greystone Mill Pond or the reference area encompassing Assapumpset Pond and Brook. Sediment concentrations of these three contaminants are highest in the two ponds (Allendale Pond and Lyman Mill Pond) adjacent to and immediately downstream of the source area. Sediment concentrations in Manton and Dyerville reaches are lower than in Allendale Pond and Lyman Mill Pond, but higher than in the upstream background location at Greystone Mill Pond. This pattern strongly indicates that these contaminants are related to the source area. Figure 50 also indicates that concentrations of 2,3,7,8-TCDD in white sucker tissue are dramatically higher in Allendale Pond and Lyman Mill Pond than in the background area greystone Mill Pond and at the Assapumpset Pond reference area. HCX was detected in white sucker tissue from Allendale Pond and Lyman Mill Pond but not at either the Greystone Mill Pond upstream background location or at the Assapumpset Pond reference location. The PCB-77 concentrations in white sucker tissue from Lyman Mill Pond are higher than the corresponding concentrations in white sucker from Allendale Pond and from the Greystone Mill Pond upstream background location. The PCB-77 concentrations in white sucker tissue from Allendale Pond are lower than corresponding concentrations in the Greystone Mill Pond upstream background location. This suggests that PCB-77 sediment concentrations in the areas adjacent to and downstream of the source area appear to be more strongly site-related than are the PCB-77 concentrations in white sucker tissue form the same areas.

There is information that indicates that 2,3,7,8-TCDD and HCX would have been expected to be present at the source area due to the manufacture of hexachlorophene. It has been documented that hexachlorophene was manufactured at the source area and that trichlorophenol was used in the manufacture of hexachlorophene (Deposition of Thomas F. Cleary, February 10, 2003). It has also been reported in the literature that 2,3,7,8-TCDD was a contaminant of trichlorophenols (International Programme on Chemical Safety (IPCS), 1989). Mr. Cleary indicated in his deposition that crude trichlorophenol was treated with chemicals at the source area in order to purify it. Mr. Cleary did not indicate which impurities were removed or where and how the removed impurities were disposed. The World Health Organization (WHO, 1989) indicates that a scientific criteria document for chlorophenols and their impurities concluded that chlorophenols were estimated to be the major chemical sources of polychlorinated dibenzodioxins and dibenzofurans in the Canadian environment. Further, it has been reported that hexachlorophene was typically produced from 2,4,5-trichlorophenol (WHO, 1989). It has been reported that 2,3,7,8-TCDD may be produced during the industrial production of 2,4,5-trichlorophenol (WHO, 1989). It is also reported that HCX is a by-product of the synthesis of hexachlorophene and that hexachlorophene contains approximately 100 mg/kg of 1,2,4,6,8,9-hexachloroxanthene (WHO, 1989).

Also, the contamination of the environment with 2,3,7,8-TCDD has been associated with a facility in Verona, Missouri that manufactured hexachlorophene (IPCS, 1989). It is reported that contaminated oil was applied to soil to control dust. Although, in addition to the 2,3,7,8-TCDD,

other trichlorinated and tetrachlorinated dioxins were found in soils, the major component of the contamination was 1,2,4,6,8,9-hexachloroxanthene. The WHO (1989) indicates that the 1,2,4,6,8,9-hexachloroxanthene is a by-product of the production of hexachlorophene but has not been associated with production of 2,4,5-trichlorophenol.

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2.0 HAZARD IDENTIFICATION

The objective of this section is to present an orderly compilation of the available sampling data on the hazardous substances present at the site, to identify data sets suitable for use in a quantitative risk evaluation, and to identify contaminants of potential concern upon which the quantitative assessment of risk will be based. Summaries of the sampling data have been generated using Risk Assessment Guidance for Superfund (RAGS) Part D standard Table 2s, for each constituent detected in biota, sediment, surface water, and floodplain and bank soils. Table 2s include the minimum and maximum concentrations (including locations of the latter), minimum and maximum data qualifiers, units, frequency of detection, range of detection limits, concentration used for screening, screening toxicity value, potential regulatory criteria (i.e., FDA levels, state standards), whether a contaminant is chosen as a COPC, and the rationale for that choice. When choosing COPCs, USEPA guidance was followed.

Consistent with EPA's Guidance for *Comparing Background and Chemical Concentrations in Soil for CERCLA Sites*, September 2002, EPA Region I recommends a baseline risk assessment approach that retains all constituents that exceed risk-based screening concentrations as COPCs for further human health risk evaluation. Background chemical concentrations are not to be utilized in the selection of COPCs. Any site-specific background issues or background risks can be addressed in the risk characterization section or in the development of Preliminary Remediation Goals (PRGs) to distinguish site-related risks from background risks. Background risks are characterized in this BHHRA and the incremental risks above background are identified.

All pollutants detected during sampling efforts, not just site-related pollutants or those that bioaccumulate, have been considered in the selection of COPCs for the human health evaluation. This will result in a total estimate of risk (including risks associated with background conditions) to the receptors potentially exposed to biota, sediment, surface water, and bank soils. Per USEPA, Region I guidance, background data are not considered when choosing COPCs.

Chemicals that are infrequently detected may be artifacts in the data due to sampling, analytical, or other problems, and may not be related to site operations or disposal practices. Based on RAGS Part A, a chemical is considered for elimination from the quantitative risk assessment if: 1) it is detected infrequently in one or perhaps two environmental media, 2) it is not detected in any other sampled media or at high concentrations, and 3) there is no reason to believe that the chemical may be present. In addition, chemicals that are considered essential human nutrients (i.e., aluminum, copper, iron, magnesium, calcium, potassium, and sodium) will not be considered in the quantitative risk assessment.

2.1 SELECTION OF CHEMICALS OF POTENTIAL CONCERN

This section identifies the chemicals present at the Site and provides rationale for inclusion of analytes as COPCs.

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2.1.1 COPC Selection Methods

COPCs are chemicals for which data of sufficient quality are available, and which may pose more than a *de minimus* health risk. The procedure used to select COPCs for the HHBRA is summarized as follows, and is consistent with USEPA Region I (USEPA, 1999) methodology:

1) *Comparison to Available Criteria*

- Selected as a COPC in **bank soils** if the maximum detected concentration exceeds the USEPA Region IX PRGs for residential soils (USEPA, 2002a).

- Selected as a COPC in **sediments** if the maximum detected concentration exceeds the USEPA Region IX PRG for residential soils (USEPA, 2002a).

- Selected as a COPC in **surface water** if the maximum detected concentration exceeds the USEPA Region IX PRG for tap water (USEPA, USEPA, 2002a).

- Selected as a COPC in **fish tissue** if the maximum detected concentration exceeds the USEPA Region III Risk Based Concentration (RBC) for ingestion of fish (USEPA, 2002b).

The soil PRGs are protective for direct contact (ingestion and dermal contact) exposures, as well as for inhalation of particulate and volatile constituents that may be released to air. The RBCs for fish tissue are protective for ingestion of fish. The PRGs and RBCs are derived for a $1x10^{-6}$ cancer risk level or a non-cancer hazard quotient (HQ) of 1. Per USEPA Region I guidance (USEPA, 1995), the PRGs and RBCs based on noncarcinogenic effects have been adjusted to represent a HQ of 0.1 for the purposes of COPC selection. This adjustment of the RBCs and PRGs per the guidance is applied to account for the possible cumulative impacts of having several chemicals that might have similar mechanisms of toxic action.

The use of residential PRGs for selection of COPCs in bank soils ensures that analytes present at concentrations that could potentially pose more than a *de minimus* risk for residential land use exposures are identified. The use of these PRGs for selection of COPCs in bank soils and sediments represents a conservative approach, since potential exposures to these media will not occur at the frequency or intensity that would be associated with residential land use. The use of tap water PRGs to identify COPCs in surface water represents a very conservative approach, since potential exposures to surface water would involve only incidental ingestion of water (the Woonasquatucket River is not used as a source of potable water). Fish ingestion RBCs are protective for fishing and, therefore, are conservative for identifying COPCs for angling in the Woonasquatucket River.

2) Essential Nutrients:

- Eliminated as COPCs because they are considered essential human nutrients. The following inorganic analytes are considered essential human nutrients: aluminum, calcium, copper, magnesium, iron, potassium, and sodium.

The results of the COPC selection for each medium at each exposure area are summarized in Tables 2.1 through 2.14. Tables 2.1 through 2.7 present the COPC selection for sediment, surface water, American eel, white sucker, largemouth bass, bank soil, and surface soil for Allendale Pond, Lyman Mill Pond, Manton Pond, Dyerville Pond and the Fogarty Center. Table 2.8 through 2.14 document COPC selection for environmental media at Assapumpset Pond and Brook and Greystone Mill Pond (both areas combined). The following notes are used to denote the reasons for selection or exclusion of analytes as COPCs:

A. A: The concentration used for COPC screening (the maximum detected concentration) is greater than the risk-based concentration; the analyte is therefore selected as a COPC.

B. S: The concentration used for COPC screening (the maximum detected concentration) is less than the risk-based concentration; the analyte is therefore not selected as a COPC.

C. E: The analyte is an essential nutrient, and is therefore not selected as a COPC.

Chemicals for which risk-based concentrations were not available were retained as COPCs.

2.1.2 COPC Selection Results

COPCs have been selected for the background area and the reference area (both areas combined), for exposure areas EA1 through EA4 as a group, and also for the Fogarty Center. The COPC selection for each medium in each of the groupings is discussed below and documented in Tables 2.1 through 2.14. In general, dioxin-like compounds (dioxins/furans, HCX, and coplanar PCBs), pesticides, Aroclors, and PAH compounds were retained as COPCs in media in which they were detected.

Exposure Areas EA1 through EA4

Table 2.1 presents the selection of COPCs for American eel collected from the river exposure areas. COPCs include dioxin-like compounds, Aroclors 1254 and 1268, pesticides, PAHs, and five inorganics.

Table 2.2 presents the selection of COPCs for largemouth bass fillets collected from the river exposure areas. COPCs include dioxin-like compounds, Aroclors 1254 and 1268, pesticides, PAHs, and three inorganics.

Table 2.3 presents the selection of COPCs for white sucker collected from the river exposure areas. COPCs include dioxin-like compounds, Aroclors 1254 and 1268, pesticides, PAHs, and five inorganics.

Table 2.4 presents the selection of COPCs for surface water collected from the river exposure areas. COPCs include dioxin-like compounds, Aroclor 1254, one pesticide (aldrin), one phthalate (bis(2-ethylhexyl)phthalate), eleven inorganics (dissolved inorganics are identified as (d) in Table 2.5), and one volatile organic compound (tetrachloroethene).

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Table 2.5 presents the selection of COPCs for sediment collected from the river exposure areas. COPCs include dioxin-like compounds, Aroclors 1254, 1260, and 1268, nine pesticides, PAHs, phthalates, and ten inorganics.

Table 2.6 presents the selection of COPCs for bank soil collected from the western shore of Allendale Reach. The only analytical parameter, 2,3,7,8-TCDD, was retained as a COPC.

Table 2.7 presents the selection of COPCs for surface soil collected from the Fogarty Center property on the eastern shore of Allendale Reach. COPCs include dioxin-like compounds, Aroclor 1254, five pesticides, PAHs, and five inorganics.

Table 2.8 presents the selection of COPCs for American eel collected from the background area and reference area. COPCs include dioxin-like compounds, Aroclors 1254 and 1268, pesticides, PAHs, and three inorganics.

Table 2.9 presents the selection of COPCs for largemouth bass fillet collected from the background area and reference area. COPCs include dioxin-like compounds (HCX detected in only one of fourteen samples), Aroclors 1254 and 1268, pesticides, three PAHs, and two inorganics (mercury and methyl mercury).

Table 2.10 presents the selection of COPCs for white sucker collected from the background area and reference area. COPCs include dioxin-like compounds (HCX not detected), Aroclors 1254 and 1268, pesticides, PAHs, and three inorganics.

Table 2.11 presents the selection of COPCs for brown bullhead collected from the background area and reference area. COPCs include dioxin-like compounds, pesticides, PAHs, and three inorganics.

Table 2.12 presents the selection of COPCs for surface water collected from the background area and reference area. COPCs include dioxin-like compounds, no pesticides, no PAHs, and three inorganics.

Table 2.13 presents the selection of COPCs for sediment collected from the background area and reference area. COPCs include dioxin-like compounds (HCX detected in eight of fourteen samples), Aroclors 1254, 1260, and 1268, pesticides, PAHs, and twelve inorganics.

Table 2.14 presents the selection of COPCs for bank/floodplain soil collected from upstream of the source area and just north of Route 44 (Greystone Pond upstream area). COPCs include dioxin-like compounds (HCX detected in four of four samples), Aroclors 1254 and 1268, pesticides, PAHs, and eleven inorganics.

Overall, the number of COPCs for each medium are reasonably consistent between the combined background area and reference area and the other four river exposure areas. However, the frequency of detection of HCX is generally lower in the reference/background media than in the exposure areas.

3.0 EXPOSURE ASSESSMENT

As defined by the USEPA (USEPA, 1989a), exposure to a chemical is the contact of that chemical with the outer boundary of the body (i.e., skin and openings such as mouth, nostrils, or punctures and lesions). An exposure assessment is the quantitative or qualitative evaluation of that contact. It describes the intensity, frequency, and duration of contact, as well as the rates at which the chemical crosses the boundary (chemical intake or uptake rates), the route by which it crosses the boundary, and the resulting amount of chemical that actually crosses the boundary (a dose) and the amount absorbed (internal dose).

The objective of the exposure assessment is to estimate the type and magnitude of exposures to COPCs at or migrating from the site. The exposure assessment is conducted to: 1) characterize the populations of humans potentially exposed via consumption of biota from the Woonasquatucket River and direct contact with surface water, sediment and bank soil at and adjacent to the river; 2) identify the mechanisms by which receptors may be exposed; and 3) identify the intake, or dose, of COPCs that receptors may receive through the identified exposure pathways. The exposure assessment includes the following components:

- Characterization of the exposure setting (including current and future land use);
- Identification of exposure pathways (including receptor identification and exposure scenarios, and exposure points);
- Identification of EPCs;
- Quantification of exposures; and
- A summary of exposures by receptor and exposure point.

Present and future potential exposures to site contaminants include the ingestion of fish (and other biota) by potential human receptors and direct contact with surface water, sediment, and bank soil and include an evaluation of sensitive receptors. In addition, the biota populations consumed by the local population are identified. Narrative descriptions and summary tables of exposure scenarios are provided in this section. The exposure scenarios for current and future potential scenarios are summarized in RAGS Part D Table 4s.

3.1 CHARACTERIZATION OF EXPOSURE SETTING

The exposure setting has previously been described in Section 1.4. Photographs of relevant Site, background area, and reference area features are presented in Appendix B.

3.2 IDENTIFICATION OF EXPOSURE PATHWAYS, POTENTIAL RECEPTORS, AND EXPOSURE POINTS

This subsection describes the receptors, exposure pathways, exposure parameters and exposure points for the visiting angler and the neighborhood resident receptors.

This step involves the identification of all relevant exposure pathways through which specific populations may be exposed (current and future) to contaminants at the site. An exposure pathway consists of four necessary elements: 1) a source or mechanism of chemical release; 2) a

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transport or retention medium; 3) a point of human contact; and 4) a route of exposure at the point of contact (USEPA, 1989a). Table 1.2 is the exposure pathway summary table. This table identifies the receptor populations, exposure media and pathways, and exposure points for this BHHRA.

Exposures were evaluated based on two scenarios, the CT and RME scenarios. The CT exposure is the typical or average exposure that would be expected in a population. The RME is the highest exposure that is reasonably expected to occur at a site. The CT and reasonable maximum exposure RME scenarios are characterized by coupling the contaminant concentrations with conservative exposure parameters developed for each exposure scenario. The CT and RME scenarios are summarized in RAGS Part D Table 4s and are discussed in sections 3.2.1 and 3.2.2 below, and results are described in the text. Exposure parameters are obtained from USEPA guidance (USEPA, 1997a) and other USEPA-approved sources. In general, RME parameters represent 95th percentile values and CT parameters represent mean values.

3.2.1 Receptor Exposure Scenarios for Biota Consumption

Currently, there is a fish consumption advisory issued by the State of Rhode Island that is not believed to be a sufficient barrier to precluding all exposure to biota from the Woonasquatucket River in the short-term. Current and potential future exposure to fish and other biota (including high lipid content biota such as eels) may occur at the river. A current biota consumption scenario as well as a future scenario is evaluated in the BHHRA with the understanding that the fish consumption advisory is not always completely effective.

For the purposes of this BHHRA, it is assumed that currently, and in the future, neighborhood residents may be involved in recreational angling and associated consumption of caught fish at the background area, the reference area, and each of the four EAs associated with the portions of the Woonasquatucket River adjacent to and downstream of the source area. In addition, it is assumed that currently, and in the future, people who do not reside in the neighborhood may visit the background area, reference area, or each of the four EAs and be involved in either recreational angling, or subsistence angling (more likely to be a future exposure scenario). Because of the uncertainty associated with this latter scenario, risks associated with the subsistence angling scenarios are evaluated in Appendix F. In addition, high-end exposure scenarios for fish consumption by Visiting Recreational Anglers and Residents Living Along the River are evaluated in Appendix G.

Table 4.1.RME presents RME parameters for consumption of a combined fish diet (American eel, largemouth bass fillet, and white sucker) for the Resident Living Along the River and the Visiting Recreational Angler. Table 4.1.CT presents CT parameters for consumption of a combined fish diet (American el, white sucker, and largemouth bass) for the Resident Living Along the River and the Visiting Recreational Angler.

As shown in Table 4.1.RME, for the Resident Living Along the River and also for the Visiting Recreational Angler, the average daily consumption rate of fish caught at the Site is assumed to be 14 grams/day for the adult, 9.3 grams/day for the older child, and 4.7 gram/day for the young child. As shown in Table 4.1.CT, for the Resident Living Along the River and also for the Visiting Recreational Angler, the average daily consumption rate of fish caught at the Site is

assumed to be 8.9 grams/day for the adult, 5.9 grams/day for the older child, and 3.0 gram/day for the young child. These average daily consumption rates do not indicate the amount of fish consumed on days of fish consumption, but rather a daily average fish consumption rate over the course of the year (all seasons included). To provide some context for these rates, the RME adult fish consumption rate of 14 gram/day represents 98 grams/week (or approximately 6 ounces of fish per week) and the CT adult fish consumption rate of 8.9 gram/day represents 62.3 grams/week (or approximately 4 ounces of fish per week). These fish consumption rates include only fish caught from the water bodies at the Site, and do not include fish caught at other locations nor fresh or preserved fish purchased for consumption.

These fish consumption rates have been estimated after review of a considerable body of information from the literature. The consumption rates for adult are primarily based on information provided in the published paper, "Estimating Consumption of Freshwater Fish Among Maine Anglers" (Ebert et al., 1993). The information contained in the Ebert study was selected as the estimate for fish consumption rates for the Site because the study evaluated the consumption of freshwater fish by anglers, the study was conducted in New England (close to the Site), and because fish species identified at the Site (largemouth bass, white sucker, and brown bullhead) were included in the study. The RME fish consumption rate selected for adults represents, from the Ebert study, the 90th percentile fish consumption by household for fish taken from rivers and streams (flowing waters), assuming that the angler consumed all of the harvested fish and did not share that fish with any family members. The CT fish consumption rate selected for adults represents, from the Ebert study, the arithmetic mean (83rd percentile) fish consumption by household for fish taken from rivers and streams (flowing waters), assuming that the angler consumed all of the harvested fish and did not share that fish with any family members.

The Ebert study established estimated amounts of fish harvested per household, but did not determine fish consumption rates for specific family members. Some sharing of the catch likely occurs as a matter of course. However, there would be considerable uncertainty introduced into the fish consumption rates if assumptions were made concerning the number of household members who share the catch and if simplifying assumptions were made concerning the sharing (such as assuming that all household members, including children, share equally). In order to avoid introducing those additional uncertainties, it has been assumed the angler consumes all the fish brought into the household. Therefore, the selected fish consumption rates for adults are likely overestimates of fish consumption rates by recreational anglers.

The fish consumption rates for the older child and younger child are estimated as portions of the identified adult rates, factoring in differences in bodyweight and food consumption rates of the different age groups. The RME and CT older child fish consumption rates are 2/3 of the corresponding adult fish consumption rates. The RME and CT young child fish consumption rates are 1/3 of the corresponding adult consumption rates. These rates are reflective of age-group specific differences in bodyweight and they are generally consistent with the information presented in the Exposure Factors Handbook (USEPA, 1997b).

The documentation of the derivation of the fish consumption rates is presented in Appendix C. The anglers obviously have potential contact with surface water, sediment, and bank soil during angling activities. Those exposure scenarios are discussed below.

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3.2.2 Receptor Exposure Scenarios for Surface Water, Sediment, and Bank Soil

Using the information summarized in Table 1.2, receptor exposure scenarios were compiled. The following paragraphs discuss the receptor exposure scenarios.

There is evidence that residents who live along the Woonasquatucket River in the Providence and North Providence area have utilized the river and its banks for recreational activities. In summer 1998, the Urban River Use Survey-Woonasquatucket River was conducted by volunteers who recorded their observations concerning uses of the Woonasquatucket River. This information gathering activity was coordinated by Kristi N. Rea, Providence City Program Manager, Urban Environmental Initiative. The information sheets for this survey indicated that people were observed fishing and catching biota including sunfish, bass, pickerel, carp, eels, and turtles. People were also observed walking along the river. The information sheets report, for the people observed at the river, a diversity in ethnic backgrounds including Caucasian, Laotian, Latino, African-American, and Asian-Pacific. Adults, teens, and children (ages 2-11) were observed at some of the observation points.

In spring of 2001, the USEPA also distributed a questionnaire to collect information concerning fishing activities in the Woonasquatucket River in the area of the Site. Less than half of the respondents had ever caught fish from the river. Among those people who had caught fish in the river, most fished for enjoyment and participated on a "catch and release" basis. People reported catching bass, bluegill, sunfish, frogs, and turtles at some time in the past. Some respondents had indicated that they had consumed fish, turtles or frogs in the past. However, most respondents were aware of the current advisory against consumption of fish and other biota from the river. The majority of the respondents indicated they no longer consume fish from the river. Representatives of USEPA and its contractors have also observed both adults and teenagers fishing from the Woonasquatucket River in the time period of 2001 through 2003.

Exposure parameters for the RME were selected from USEPA guidance documents (USEPA, 1994; 1997; 2001) and were based on professional judgment considering the site-specific exposure conditions. This subsection describes the exposure scenarios and RME exposure parameters in detail. Exposure parameters for the CT were based on the RME values, with the following modifications:

- CT values for incidental ingestion of soil, sediment, and surface water were identified as one-half the RME values, based on USEPA Region I guidance (USEPA, 1994) which recommends using one-half the RME value as the CT value for incidental soil ingestion.
- CT values for soil and sediment dermal adherence were the recommended CT parameters from USEPA RAGS Part E guidance (USEPA, 2001b).
- The RME values assume that a receptor uses the Site for all of their outdoor activities (e.g., recreational play/exploration, recreational angling, or subsistence angling). The CT parameters accommodate the assumption that a more "typical" or "average" receptor would spend a portion of their outdoor time at the Site (i.e., would access other, non-Site related areas for recreational purposes).

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Area Resident Living Along the River

A resident who lives at the Centredale Manor Apartments, Brook Village Apartments, or a private residence at one of the residential lots along the eastern shore of the Woonasquatucket River may visit water bodies at the Site for recreational angling, recreational walking, exploring the banks of the river and ponds, and wading and swimming. It is assumed that area residents include young children (ages 1 through 6), older children (ages 7 through 18), and adults (assumed ages 19 through 30). Potential exposures to surface water and aquatic (submerged) sediment by incidental ingestion and dermal contact may occur during angling, wading, or swimming (no swimming in Assapumpset Brook) at Greystone Mill Pond area and Assapumpset Brook and Pond, Allendale Pond, Lyman Mill Pond, Manton Reach, and Dyerville Reach. In addition, possible exposures to bank surface soils at Greystone Mill Pond area, Allendale Pond and Lyman Mill Pond by incidental ingestion and dermal contact may occur when area residents access the water bodies for recreational angling, swimming and wading, or when walking or exploring the edges of the ponds.

Cancer and non-cancer risk estimates are calculated separately for each of the exposure media and exposure points identified in Table 1.2. The risks for each medium at each exposure point summed to derive a total risk for surface water, sediment, and bank soil (Allendale and Greystone Mill ponds only) at each exposure point. Those risks will then be added to the risks for consumption of fish to derive a cumulative risk estimate for each exposure point in the BHHRA.

The RME and CT exposure parameters for surface water are presented in Tables 4.2.RME and 4.2.CT, for sediment in Tables 4.3.RME and 4.3.CT, for bank soil in Tables 4.4.RME and 4.4.CT, and for surface soil at the Fogarty Center in Tables 4.5.RME and 4.5.CT.

Exposure Duration. For the RME scenario, it is assumed that an area resident is raised at and remains at the same residence over a 30-year period (USEPA, 1994). The 30-year duration is segregated into three age periods: young-child (ages 1 through 6) for 6 years; older child (ages 7 through 18) for 12 years; and adult (ages 19 through 30) for 12 years. The CT exposure duration values are based on the recommended CT parameters for exposure duration published in USEPA RAGS Part E of 9 years. The 9-year exposure duration value was segregated as follows: young child (2 years); older child (3 years); and adult (4 years).

Exposure Frequency. It is assumed that an area resident visits the water bodies or banks of the water bodies for walking/exploring or recreational angling May through October, and for wading or swimming June, July and August. It is further assumed that during summer months, wading (defined as standing or walking in water to a depth of the knees) occurs more frequently than swimming (defined as total submersion of the body in water). The exposure frequency associated with these various activities is broken down as follows:

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Activity	RME Frequency / Period	RME Total No. Days per Year	
		Young Child	Older Child / Adult
Walking/Exploring banks and recreational angling	2x/week: May, Sept, Oct 4x/week: June – Aug	78	78
Swimming	1x/week: Jun – Aug	0	13
Wading – Older Child/Adult	3x/week: Jun – Aug	--	39
Wading – Young Child	4x/week: Jun – Aug	52	--
Total Water Bodies Exposure	4x/week: Jun – Aug	52	52

Activity	CT Frequency / Period	CT Total No. Days per Year	
		Young Child	Older Child / Adult
Walking/Exploring banks and recreational angling	1x/week: May, Sept, Oct 2x/week: June – Aug	39	39
Swimming	1x/week: Jun – Aug	0	13
Wading – Older Child/Adult	1x/week: Jun – Aug	--	13
Wading – Young Child	2x/week: Jun – Aug	26	--
Total Water Bodies Exposure	2x/week: Jun – Aug	26	26

The exposure frequency during the summer months assumes that walking/exploring banks, recreational angling, and wading/swimming each occur on the same day, such that a total of four visits to the water bodies occur each week.

For the young child receptor, it is assumed that all time spent in the water during the summer months would be associated with wading rather than swimming (as young children are usually not able to swim), such that the RME exposure frequency for wading is 52 days per year, and the exposure frequency for swimming is zero days per year.

It is assumed that potential exposures to bank soil (Greystone Mill Pond area, Allendale and Lyman Mill Ponds only) occur each day that access to the Site occurs (78 days per year). Contact with submerged sediment is not likely to be substantial under any of the scenarios, as explained under "Ingestion Rate" below. However, if contact with sediment were to occur, it would be during wading activities when a person is standing in the water (i.e., standing in the sediment), and not when a person is swimming (i.e., when body parts do not contact the sediment for more than a minute or two). Therefore, the exposure frequency for sediment is based on the exposure frequency for wading (39 days per year for adults/older child, and 52 days per year for young children). The exposure frequency for surface water is based on the total frequency for wading and swimming (52 days per year for all age groups).

Exposure Time and Event Frequency. Exposures to surface water during swimming or wading activities are assumed to occur 1 hour per event, 1 event per day (i.e., 1 hour per day), based on the recommended exposure time for recreational swimming (USEPA, 1997).

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Body Weight. Body weight values for young children and adults are based on values recommended in USEPA guidance (USEPA, 1994). Body weight values for older children are based on the average of 50th percentile body weights for males ages 7 through 18 (USEPA, 1997).

Incidental Ingestion Rate and Fraction Ingested. The incidental ingestion rate for surface water is based on the recommended value for incidental ingestion of water during swimming of 50 ml per hour (USEPA, 1988). The fraction ingested parameter for surface water is 100%, indicating that 100% of surface water intake on the day-exposed is assumed to occur at the Site.

The incidental ingestion rates for bank surface soil are the default ingestion rate values for soil recommended in USEPA (1994) guidance; the ingestion rate for adults is applied to older children who are less likely than young children to place soil-covered hands in the mouth. The fraction ingested parameter for bank surface soil is 100%.

Human exposure parameter values specifically applicable to sediment are not provided in USEPA Region I or USEPA national guidance. Since incidental ingestion exposure to soil (or sediment) primarily occurs through hand-to-mouth transfer of material that has adsorbed to the skin, it is unlikely that ingestion exposure to COPCs in submerged sediment would occur through hand-mouth contact because sediment would wash off of the hand while the hand was being removed from the water. Nonetheless, it is possible that some sediment would adhere to the skin when leaving a water body (i.e., some sediment may not wash off), and it is possible that sediment entrained in the surface water could be ingested if surface water is incidentally ingested.

Incidental ingestion values for soil may be used for sediment. However, values for soil are generally considered conservative for sediment because: 1) The mechanism of exposure to sediment is different from soil, resulting in less particle adherence to the skin and lower dermal and ingestion exposures, as outline above; and 2) soil incidental ingestion rate values are based on daily intakes from all sources of soil and sediment; it is not generally appropriate to assume that a receptor's total daily intake of soil and sediment is derived from sediment on the days of sediment exposure. Nonetheless, it is possible that some residents could spend most of their outdoor recreational time at the river's edge rather than at their own yard. Given these considerations, the soil ingestion rates published by USEPA Region I (USEPA, 1994) are used as sediment ingestion rates, and the fraction ingested parameter for sediment of 100% is applied for the resident receptor only.

Dermal Surface Area and Adherence Factor. The dermal surface area for surface water is different between wading and swimming exposures. Wading exposures for the resident living Along the River would involve contact with only the lower legs, feet, and hands, whereas swimming exposures would involve contact with the entire body. For adults and older children, the surface water exposures are quantified using the frequency-weighted average dermal surface area associated with wading and swimming. These weighted average surface area values reflect the relative frequencies of wading and swimming activities and they are presented in Tables 4.2.RME and 4.2.CT. For the young child, the dermal surface area is based on the areas of the legs, feet, and hands (average of 50th percentile values for males ages 1 through 6), as these are the body parts that would contact surface water during wading.

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Exposures to aquatic sediments that are submerged beneath the water are unlikely to be substantial. In order for dermal absorption of COPCs from sediment to occur, the material must adhere to the skin (USEPA, 2001). Sediment that is submerged would not adhere to skin, as the surrounding surface water would prevent binding of the sediment to the skin. In addition, when a body part that contacts sediment is removed from the surface water body, the majority of sediment would wash off, thereby preventing adherence of the material. Hence, although dermal exposures to COPCs in sediment are likely to be negligible, they are quantified to account for the possibility that some sediment may remain adhered to the skin following contact with surface water.

Exposure parameter values for soil may be used for sediment. However, exposure parameter values for soil are generally too conservative for sediment as outlined above. Therefore, dermal surface area and adherence values are based on the following:

- Child recreational wader: USEPA RAGS Part E child resident default values for body surface area and soil adherence. These variables account for the possibility that a child may play in shallow water and contact sediment with body parts other than just feet and lower legs.

- Older child and adult wader: Body surface area values for feet, lower legs, and hands (average of 50th percentile values for males), and RAGS Part E resident default values for soil adherence for sediment exposure. These values account for the fact that upper legs, arms, and face would not contact sediment.

The dermal surface area and adherence factor values for bank surface soil are based on the RAGS Part E (USEPA, 2001c) default values for residential exposures to soil.

Visiting Recreational Angler

A person who lives in the vicinity of the Site may access the water bodies for recreational fishing. It is assumed that recreational anglers include older children and adults; young children would not participate in fishing, but are assumed to eat the fish that may be caught by older siblings and parents. Potential exposures to surface water and aquatic (submerged) sediment by incidental ingestion and dermal contact are assumed to occur during angling (e.g., if wading occurs as part of the angling) at Greystone Mill Pond area, Assapumpset Brook and Pond, Allendale Pond, Lyman Mill Pond, Manton Reach, and Dyerville Reach. This scenario assumes that a recreational angler does not use waders while standing in the water. In addition, possible exposures to bank surface soils at Greystone Mill Pond area, Allendale Pond and Lyman Mill Pond by incidental ingestion and dermal contact may occur when recreational anglers access the water bodies for angling.

Cancer and non-cancer risk estimates were calculated separately for each of the exposure media and exposure points identified in Table 1.2. The risks for each medium at each exposure point were summed to derive a total risk for surface water, sediment, and bank soil (Allendale and Lyman Mill ponds only) at each exposure point. Those risks were then be added to the risks for consumption of fish to derive a cumulative risk estimate for each exposure point in the BHHRA.

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The RME and CT exposure parameters for surface water are presented in Tables 4.2.RME and 4.2.CT, for sediment in Tables 4.3.RME and 4.3.CT, and for bank soil in Tables 4.4.RME and 4.4.CT and are discussed below.

Incidental Ingestion Rate and Fraction Ingested. The incidental ingestion rate for surface water is based on the recommended value for incidental ingestion of water during swimming of 50 ml per hour (USEPA, 1988). The fraction ingested parameter for surface water is 100%, indicating that 100% of surface water intake on the day-exposed is assumed to occur at the Site.

The incidental ingestion rates for bank surface soil are the default ingestion rate values for soil recommended in USEPA (1994) guidance; the ingestion rate for adults is applied to older children who are less likely than young children to place soil-covered hands in the mouth.

Human exposure parameter values specifically applicable to sediment are not provided in USEPA Region I or USEPA national guidance. Since incidental ingestion exposure to soil (or sediment) primarily occurs through hand-to-mouth transfer of material that has adsorbed to the skin, it is unlikely that ingestion exposure to COPCs in submerged sediment would occur through hand-mouth contact because sediment would wash off of the hand while the hand was being removed from the water. Nonetheless, it is possible that some sediment would adhere to the skin when leaving a water body (i.e., some sediment may not wash off), and it is possible that sediment entrained in the surface water could be ingested if surface water is incidentally ingested.

Incidental ingestion values for soil may be used for sediment. However, values for soil are generally considered conservative for sediment because: 1) The mechanism of exposure to sediment is different from soil, resulting in less particle adherence to the skin and lower dermal and ingestion exposures, as outline above; and 2) soil incidental ingestion rate values are based on daily intakes from all sources of soil and sediment; it is not generally appropriate to assume that a receptor's total daily intake of soil and sediment is derived from sediment on the days of sediment exposure.

Fraction Ingested. The fraction ingested parameter for sediment and bank soil is established for each medium at 100% for area resident and 50% for non-resident (i.e., visiting recreational angler, visiting subsistence angler, and commercial/industrial worker). The fraction ingested value of 50% accommodates the likelihood that visiting receptors who do not live immediately near the Site would incur only a portion (50%) of their daily exposure to soil or sediment at the Site, and the remainder at off-site locations such as residential yards.

Exposure Duration. It is assumed that the visiting recreational angler is an area resident that accesses the water bodies at the Site while they live in the area. Consistent with USEPA Region I policy, the RME exposure duration for fish consumption is 30 years and the CT exposure duration for fish consumption is 9 years. The RME exposure duration values, therefore, are based on a 30-year duration segregated into three age periods: young-child (ages 1 through 6) for 6 years; older child/older child (ages 7 through 18) for 12 years; and adult (ages 19 through 30) for 12 years. These exposure parameters are consistent with those used to evaluate potential risks associated with biota consumption. The CT exposure duration values are based on the assumption that a recreational angler lives in the vicinity of the Site for 9-years (consistent with USEPA Region I assumption for the RME exposure duration of a resident and for fish

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consumption). The 9-year duration is segregated into three age periods: young-child (2 years); older child/older child (3 years); and adult (4 years).

Since the visitor angling activity and visitation to the Site is assumed to involve only the older child/older child and adult populations, only those age groups are quantitatively evaluated for potential exposure to surface water sediment, and bank soil. A young child is not considered to be a component of the Visiting Recreational Angler with respect to potential exposure to surface water, sediment, and bank soil. The Visiting Recreational Angler is assumed to be involved in wading but not swimming during site visits.

Exposure Frequency. It is assumed that a recreational angler visits the Site once per week from May through October. However, contact with surface water and sediment is assumed to occur only during the warmer months (i.e., June through August). Therefore, the exposure frequency for surface water and sediment is 13 days per year.

Exposure Time and Event Frequency. Exposures to surface water are assumed to occur 1 hour per event, 1 event per day (i.e., 1 hour per day). It is unlikely that a person who is angling for recreation would stand in the water for more than one hour without the use of waders.

Body Weight, Incidental Ingestion Rate, Dermal Surface Area and Adherence Factor. The values for these parameters are the same as those used for the area resident (above), with the following exceptions:

- *Surface water ingestion rate*: It is assumed that a recreational angler would have little, if any, incidental ingestion of water. Therefore, the surface water ingestion rate used for the recreational angler was estimated at 5 ml/hour, representing one-tenth the USEPA-published value for incidental ingestion of surface water during swimming activities.

- *Dermal surface area for surface water*: It is assumed that the feet, entire legs, and hands of a recreational angler standing in surface water would be exposed to surface water (values based on the average of 50th percentile values for males) during wading activities. The surface area values for this receptor and exposure pathway are presented in Tables 4.2.RME and 4.2.CT.

Cancer and non-cancer risk estimates are calculated separately for each of the exposure media and exposure points identified in Table 1.2. The risks for each medium at each exposure point will have been summed to derive a total risk for surface water, sediment, and bank soil (Greystone Mill Pond, Allendale and Lyman Mill Ponds only for soil) and fish consumption at each exposure point to derive a cumulative risk estimate for each exposure point in the BHHRA.

The RME and CT exposure parameters for surface water are presented in Tables 4.2.RME and 4.2.CT, for sediment in Tables 4.3.RME and 4.3.CT, and for bank soil in Tables 4.4.RME and 4.4.CT and are discussed below.

Commercial/Industrial Worker

Exposure parameters are presented in Tables 4.8.RME and 4.8.CT for a commercial/industrial worker at the Fogarty Center. A commercial/industrial worker employed at the Fogarty Center could contact surface soils during work-day breaks, lunch hour, etc. It is assumed that the worker

would contact the soil via incidental ingestion and dermal contact each work-day when the weather is not inclement or the ground frozen or snow covered (assumed to be approximately 150 days per year per USEPA Region I guidance [USEPA, 1994]). This receptor's exposure duration is assumed to be 25 years, consistent with a commercial/industrial scenario. Since potential contact with the soil would only be through incidental exposure during short periods (e.g., when on work breaks), it is not reasonable to assume that an employee's entire daily incidental intake of soil would occur at the Fogarty Center. Therefore, the fraction ingested value is established at 0.5. Values for other exposure parameters are USEPA default values for commercial/industrial land use (USEPA, 1994; 2001).

3.2.3 Exposure Points

Five Site exposure points have been identified, mostly defined by impoundments. The five separate site exposure points include Allendale Reach, Lyman Mill Reach, Manton Reach, Dyerville Reach, and the property at the Fogarty Center. This facility is located on the eastern shore of Lyman Mill Pond. In addition, an upstream background exposure point (Greystone mill Pond) and a reference area exposure point (Assapumpset Pond and Brook) have also been identified as exposure points for the BHHRA.

Exposure Point Concentrations

A single concentration is selected as representative of the actual concentration for each COPC in a given medium for a given exposure point. This value, called the EPC, is used in the estimates of health risks at the site. An EPC is selected for every COPC identified in the screening process described earlier.

For both RME and CT, the 95% Upper Confidence Limit (UCL) on the mean is typically used as the EPC. There are two exceptions to this rule. In the case where the 95% UCL is greater than the maximum detected concentration; and/or if there are fewer than 10 samples in a data set (the UCL is not calculated). For these two situations, the maximum detected concentration should be used as the RME EPC and the arithmetic average concentration should be used for the CT EPC.

Risk Assessment Guidance for Superfund (RAGS): Volume I Human Health Evaluation Manual (Part D: Standardized Planning, Reporting and Review of Superfund Risk Assessments) Final December 2001 has superseded the interim version published in 1998 and is used for the ongoing risk assessment work at Centredale Manor.

RAGS Part D requires distribution testing to determine if the data distribution is normal, lognormal, or non-parametric prior to calculating the 95% UCL. This information is used to identify the method to be used to calculate the 95% UCL.

RAGS Part D, Instructions for Table 3 (p. B3-8): These instructions indicate choices of statistics for EPCs. Three options for 95% UCLs are offered, based on the distribution type of the data. These options include:

- 95% UCL - N (95% UCL of Normal Data)
- 95% UCL - T (95% UCL of Log-transformed Data)

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- 95% UCL - NP (Mean of Nonparametric Data)

Distribution testing of data sets was conducted using the Shapiro Wilks W Test. Using that test, data are determined to be either normally distributed, lognormally distributed (the log-transformed data are normally distributed), or non-parametrically distributed.

RAGS Part D Guidance suggests that the 95% UCL would be calculated per the 1992 USEPA Guidance for Calculating the Concentration Term for normal and lognormal data. RAGS Part D indicates that the arithmetic mean concentration would be used to approximate the 95% UCL for nonparametric data and would be used as the exposure point concentration.

The equations for calculating the 95% UCL for normally and lognormally distributed data presented in *Calculating the Concentration Term, Supplemental Guidance to RAGS* (USEPA, 1992b) have been used in the calculation of 95% UCLs for normally and lognormally distributed data. The arithmetic average concentration has been used for non-parametrically distributed data.

In calculating the 95 percent UCLs, non-detects have been assigned a concentration equal to one-half the sample quantification limit (SQL). If an SQL was not available one of the following values will be substituted: the method detection level or the contract required quantitation limit (CRQL) for organics, or contract required detection limit (CRDL) for inorganics. For chemicals detected at least once in a particular medium, one-half the SQL will be used to represent non-detects when calculating arithmetic averages. For duplicate pairs that have one detect and one non-detect reported, the detected value is used to represent that location.

Tables 3.1.RME through 3.8.RME and Tables 3.1.CT through 3.8.CT document the calculation and identification of both RME and CT EPCs for all media evaluated. Each table contains all of the EPCs for each of the exposure points for that medium.

The RME EPCs for each fish species is the 95% UCL on the mean wet weight concentration or the maximum concentration if the data set contains fewer than 10 samples or if the 95% UCL is greater than the maximum concentration. The CT EPCs for each species is the arithmetic mean concentration for each analyte. This is consistent with EPA Region I's policy in calculating EPCs.

The EPCs for the combined fish diet (presented in Tables 3.4.RME and 3.4.CT) have been calculated as the arithmetic mean of the EPCs for the fish species sampled at each exposure point. This approach assumes, absent information to the contrary, that each of the species present at a given exposure point would be consumed at the same rate. In other words, the receptor consumes equal amounts of the fish species that are identified at each exposure point. The combined fish diet EPC at a given exposure point was calculated as follows if three fish species were sampled and analyzed (Assapumpset Pond and Brook, Greystone Mill Pond, and Lyman Mill Pond):

$$EPC_{\text{combined fish diet}} = [(EPC_{\text{species 1}}) + (EPC_{\text{species 2}}) + (EPC_{\text{species 3}})] / 3$$

When two species have been sampled and analyzed at a given exposure point (Allendale Pond), the combined fish diet EPC at a given exposure point was calculated as follows:

$$EPC_{combined\ fish\ diet} = [(EPC_{species\ 1}) + (EPC_{species\ 2})] / 2$$

For an exposure point with only one species sampled and analyzed (Manton Pond and Dyerville Pond), the EPC for that species is considered to be the EPC for the combined fish diet at that exposure point.

The surface water EPCs for the Lyman Mill reach have been used as estimators of surface water quality for the Manton and Dyerville reaches for the purposes of this risk assessment. Surface water samples were not available for those two reaches. The surface water data for Lyman Mill Pond are considered reasonable surrogates for water quality in Manton and Dyerville Ponds because the surface water in Lyman Mill Pond is located immediately upstream of Manton Pond and the water in Lyman Mill Pond would actually flow through Manton Pond and Dyerville Pond in time.

3.3 QUANTIFICATION OF EXPOSURES

The next step is to calculate COPC intakes, via consumption of fish and other biota and direct contact with surface water, sediment, and bank soil for each of the potentially exposed populations. Population-related variables have been selected that describe the characteristics associated with individual receptors in that population.

3.3.1.1 Estimation of Chemical-Specific Intakes

The chemical-specific intake, or the average daily dose (ADD), is the amount of COPC absorbed into the body. When appropriate, it is the product of the average daily exposure and an absorption factor (ABS). Chemical-specific intakes were calculated in a manner consistent with USEPA guidance for risk assessment (USEPA, 1989a; 2001a).

A Lifetime Average Daily Dose (LADD) is calculated in order to estimate carcinogenic risk. The Averaging Time (AT) over which the total intake of COPC is averaged is 70 years for carcinogenic effects (USEPA, 1989a).

For noncarcinogenic effects, depending on the duration of the exposure period, an Average Daily Dose, Chronic (ADD_c) for long-term exposure (seven years or longer) or Average Daily Dose, Subchronic (ADD_s) for exposure periods from a month up to seven years may be calculated.

Soil and Sediment Direct Contact Exposures

The ADD received by a receptor via direct contact with soil (ADD_{soil}) is the sum of the ADDs for exposure via the routes of dermal contact with the contaminated soil and ingestion of the contaminated soil. The same approach is used for evaluation of sediment contact. Possible inhalation exposures are evaluated as described below. Thus,

$$ADD_{soil} = ADD_{dermal} + ADD_{ingestion}$$

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Dermal Contact. The ADD due to dermal contact with COPC-contaminated soil ($ADD_{dermal\ absorption}$) may be calculated:

$$ADD_{dermal\ absorption} = \frac{DAevent * SA * EF * ED}{BW * AT}$$

and:

$$DAevent = [COPC]soil * AF * ABS * C$$

Where:

$ADD_{dermal\ absorption}$	=	Average daily dose of COPC received through dermal contact with soil during the period of exposure (dimensions: mass/mass×time, typical units: mg/kg×day)
DAevent	=	Dose of COPC absorbed during each exposure event (dimensions: mg/cm^2)
$[COPC]_{soil}$	=	EPC of COPC in the soil at the exposure point during the period of exposure (dimensions: mg/kg)
SA	=	Skin surface area in contact with the soil on days exposed (dimensions: cm^2/day)
AF	=	Mass of soil adhered to the unit surface area of skin exposed (dimensions: mg/cm^2)
ABS	=	Absorption Factor; represents the fraction of COPC that may be absorbed through the skin from soil (unitless)
EF	=	Exposure Frequency: the number of exposure events during the exposure period divided by the number of days in the exposure period (dimensions: days/year)
ED	=	Exposure Duration: the period of time over which exposure may occur (dimension: years)
BW	=	Body Weight of the receptor of concern during the exposure duration dimension: kg)
AT	=	Averaging Time (dimension: days)
C	=	Appropriate units conversion factor(s)

Ingestion. The ADD due to the incidental ingestion of COPC contaminated soil ($ADD_{ingestion}$) may be calculated:

$$ADD_{ingestion} = \frac{[COPC]_{soil} * IR * EF * ED * C}{BW * AT}$$

Where:

ADDingestion	=	Average daily dose of COPC received through the ingestion of soil during the period of exposure (dimensions: mass/mass×time, typical units: mg/kg×day)
[COPC]soil	=	EPC of the COPC in soil (dimensions: mass/mass, typical units: mg/kg)

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IR	=	Daily soil ingestion rate on days exposed during the exposure period (dimensions: mass/time, typical units: mg/day)
EF	=	Number of exposure events during the exposure period divided by the number of days in the exposure period (dimensions: events/time, typical units: days/year)
ED	=	Duration of the exposure period (dimensions: time, typical units: years)
C	=	Appropriate units conversion factor(s)
BW	=	Body weight of the receptor of concern during the averaging period (dimensions: mass, typical units: kg)
AT	=	Averaging Time (dimension: time, typical units: days)

Inhalation Exposures

Receptors at the Site, under certain conditions, have the potential to be exposed via inhalation to COPC adsorbed to wind-eroded particles or dust, or to vapors that may migrate from soil to air.

The methodology for evaluating inhalation exposures differs from that used for other exposure pathways in that the toxicity values used are reference concentrations (RfCs) and unit risks (URs) instead of reference doses (RfDs) and slope factors (SFs). Because concentration and not dose is the basis for these toxicity values, body weight (BW), and respiration rate (IR) are not used in calculating potential risk estimates for carcinogenic and noncarcinogenic chemicals. Therefore, an average air concentration, rather than an ADD, is calculated.

The general equation for estimating the average air concentration due to inhalation is as follows:

$$COPC_{air} = \frac{[CA]_{air} * ET * EF * ED}{C * AT}$$

Where:

$[COPC]_{air}$	=	Representative concentration of OHM in the air at the Exposure Point during the period of exposure (dimensions: mass/volume)
[CA]air	=	Measured or modeled EPC (dimensions: mass/volume, typical units: ug/m^3)
EF	=	Number of exposure events during the exposure period divided by the number of days in the exposure period (dimensions: events/time, typical units: days/year)
ET	=	Number of hours per day the exposure occurs (dimensions: time, typical units: hours/day)
ED	=	Duration of the exposure period (dimensions: time, typical units: years).
C	=	Appropriate units conversion factor(s) (24 hours per day)
AT	=	Averaging Time (dimension: time, typical units: days).

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Water Direct Contact Exposures

Potable use of groundwater, as well as non-potable use of groundwater in a wading pool, includes ingestion and dermal contact exposures. Exposures are quantified in accordance with USEPA guidance (USEPA, 1989a; 1995; 2001a).

Ingestion. The ADD due to the incidental ingestion of COPC contaminated surface water ($ADD_{ingestion}$) may be calculated:

$$ADD_{ingestion} = \frac{[COC]_{water} * IR * EF * ED * C}{BW * AT}$$

Where:

$ADD_{ingestion}$	=	Average daily dose of COPC received through the ingestion of surface water during the period of exposure (dimensions: mass/mass×time, typical units: mg/kg×day)
$[COPC]_{water}$	=	EPC of the COPC in surface water (dimensions: mass/mass, typical units: mg/L)
IR	=	Daily water ingestion rate on days exposed during the exposure period (dimensions: mass/time, typical units: L/day)
EF	=	Number of exposure events during the exposure period divided by the number of days in the exposure period (dimensions: events/time, typical units: days/year)
ED	=	Duration of the exposure period (dimensions: time, typical units: years)
C	=	Appropriate units conversion factor(s)
BW	=	Body weight of the receptor of concern during the averaging period (dimensions: mass, typical units: kg)
AT	=	Averaging Time (dimension: time, typical units: days)

Dermal. The ADD due to dermal contact with COPC-contaminated water ($ADD_{dermal\ absorption}$) may be calculated:

$$ADD_{dermalabsorption} = \frac{DAevent * EV * ED * EF * SA}{BW * AT}$$

and:

For Inorganics: $DA_{event} = K_p * [COPC]_{water} * t_{event}$

For Organics when $t_{event} \leq t^*$: $DA_{event} = 2 * FA * K_p * [COPC]_{water} * [6 * T_{event} * t_{event} / 3.14]^{0.5}$

For Organics when $t_{event} > t^*$: $DA_{event} = FA * K_p * [COPC]_{water} * [(t_{event} / 1 + B) + 2 * T_{event} (1 + 3 * B + 3 * B^2 (1 + B)^{-2})]$

Where:

$ADD_{dermal\ absorption}$	=	Average daily dose of COPC received through dermal contact with water during the period of exposure (dimensions: mass/mass×time, typical units: mg/kg×day).

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$[COPC]_{water}$	=	EPC of COPC in the water at the exposure point during the period of exposure (dimensions: mg/L)
DA_{event}	=	Dose absorbed per event (mg/cm^2-event)
SA	=	Skin surface area in contact with the water on days exposed (dimensions: cm^2/day)
K_p	=	Permeability Constant; represents the amount of COPC that may be absorbed through the skin from water (units: cm/hr)
t_{event}	=	Number of hours per day the exposure occurs (dimensions: time, typical units: hours/event)
T_{event}	=	Lag time per event (hr/event)
FA	=	Fraction absorbed (unitless)
t*	=	Time to reach steady state (hr)
B	=	Ratio of the permeability coefficient of a compound through the stratum corneum relative to its permeability coefficient across the epidermis (dimensionless)
EF	=	Exposure Frequency: the number of exposure events during the exposure period divided by the number of days in the exposure period (dimensions: days/year)
EV	=	Event Frequency: the number of exposure events per day (dimensions: events/day)
ED	=	Exposure Duration: the period of time over which exposure may occur (dimension: years)
BW	=	Body Weight of the receptor of concern during the exposure duration dimension: kg)
AT	=	Averaging Time (dimension: days)

Values for FA, K_p, T_{event}, t_{event}, t*, and B were obtained from RAGS Part E (USEPA, 2001a). Due to uncertainties associated with dermal exposures for dioxins and furans in surface water, the evaluation of exposures and risks for dioxins and furans in surface water are presented in Appendix I.

Fish Ingestion

The ADD due to ingestion of COPC contaminated fish tissue ($ADD_{ingestion}$) may be calculated:

$$ADD_{ingestion} = \frac{[COPC]_{fish} * IR * EF * ED * C}{BW * AT}$$

Where:

ADDingestion	=	Average daily dose of COPC received through the ingestion of fish during the period of exposure (dimensions: mass/mass×time, typical units: mg/kg×day).
[COPC]fish	=	EPC of the COPC in fish (dimensions: mass/mass, typical units: mg/kg).
IR	=	Daily fish ingestion rate on days exposed during the exposure period (dimensions: mass/time, typical units: mg/day)
EF	=	Number of exposure events during the exposure period divided by the number of days in the exposure period (dimensions: events/time, typical units: days/year).

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ED	=	Duration of the exposure period (dimensions: time, typical units: years).
C	=	Appropriate units conversion factor(s)
BW	=	Body weight of the receptor of concern during the averaging period (dimensions: mass, typical units: kg).
AT	=	Averaging Time (dimension: time, typical units: days).

The daily chemical intakes have been calculated separately for non-cancer and cancer endpoints using the EPCs presented in Tables 3.1.RME through 3.8.RME and Tables 3.1.CT through 3.8.CT, the exposure parameters and intake equations shown in Tables 4.1.RME through 4.5.RME and Tables 4.1.CT through 4.5.CT. The daily intakes for visiting recreational anglers, residents living along the river, and employees at the Fogarty Center, all age-groups, and all media are calculated in Tables 7.1.RME through 7.37.RME and Tables 7.1.CT through 7.37.CT for RME and CT exposures respectively. Each of those tables shows all daily intake calculations for all exposure media for a receptor group/age-group/exposure point combination.

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SECTION 4.0

4.0 TOXICITY ASSESSMENT

4.1 TOXICITY ASSESSMENT

The purpose of the toxicity assessment is to characterize the relationship between the dose of COPC administered or received and the incidence of adverse health effects in the exposed population. From this quantitative dose-response relationship, toxicity values (e.g., slope factors, reference dose values, or reference concentrations) are derived that can be used to estimate the likelihood of adverse effects as a function of human exposure to an agent. These toxicity values are used in the risk characterization process to estimate the potential for adverse effects occurring in humans at different exposure levels.

The dose-response relationship(s) for each chemical that has been selected as a COPC is presented in this section. The dose-response information may be divided into two major categories:

- Toxicity information associated with threshold (non-carcinogenic) health effects.
- Toxicity information concerning carcinogenicity, either from human epidemiologic data or from laboratory studies.

All the chemicals selected as COPCs are evaluated for potential *non-carcinogenic* health effects. In addition, any substance considered to be a *known*, *probable*, or *possible* human carcinogen is also evaluated for its potential carcinogenic effects. The classification of a chemical as a carcinogen does not preclude an evaluation of that same chemical for potential non-carcinogenic health risks, as all potentially carcinogenic chemicals may also exert non-carcinogenic health effects.

4.1.1 Dose-Response Assessment for Carcinogenic Effects

It has generally been assumed that carcinogenic effects are non-threshold effects (IRIS, 2003). This means that any dose, no matter how small, is assumed to pose a finite probability of generating a response. Thus, no dose of a carcinogen is thought to be risk-free. For carcinogenic effects, USEPA uses a two-part evaluation in which the substance is first assigned a weight-of-evidence classification, and then a slope factor (SF) or unit risk (UR) is calculated to reflect the carcinogenic potency.

The weight-of-evidence evaluation involves determining the likelihood that the agent is a human carcinogen. USEPA has developed a system for characterizing the overall weight of evidence for a chemical's carcinogenicity based on the availability of animal, human, and other supportive data (USEPA, 1989a). The weight-of-evidence classification rates the likelihood that an agent is a human carcinogen. It qualitatively affects the interpretation of potential health risks. Three major factors are considered in characterizing the overall weight-of-evidence for carcinogenicity: (1) the quality of evidence from human studies, (2) the quality of evidence from animal studies, and (3) other supportive information, such as mutagenicity data and structure-activity data.

USEPA's final classification of the overall weight-of-evidence has the following five categories:

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Group A - Human Carcinogen. This category indicates there is sufficient evidence from epidemiological studies to support a causal association between an agent and human cancer.

Group B - Probable Human Carcinogen. This category generally indicates there is at least limited evidence from epidemiologic studies of carcinogenicity to humans (Group B1) or that, in the absence of data on humans, there is sufficient evidence of carcinogenicity in animals (Group B2).

Group C - Possible Human Carcinogen. This category indicates that there is limited evidence of carcinogenicity in animals in the absence of data on humans.

Group D - Not Classified. This category indicates that the evidence for carcinogenicity in animals is inadequate.

Group E - No Evidence of Carcinogenicity to Humans. This category indicates that there is evidence of noncarcinogenicity in at least two adequate animal tests in different species or in both epidemiologic and animal studies.

The ability of a chemical to increase the incidence of cancer in a target population is described by one of two values: the carcinogenic SF or the UR. Cancer SFs or URs are typically calculated for chemicals in Groups A, B1, and B2. Cancer dose-response values for chemicals in Group C are calculated on a case-by-case basis.

The cancer SF for a chemical is derived by the USEPA's Cancer Risk Assessment Verification Endeavor (CRAVE). Using data derived from animal studies, the SF is an estimate of the upper 95% Confidence Limit of the slope of the dose-response curve extrapolated to low doses.

For some chemicals, human epidemiologic data is the basis of an estimate of the carcinogenic potency, although the most common basis of these values is an animal study. The SF is given in units of $(mg/kg/day)^{-1}$ and is based upon the concept of a lifetime average daily dose. Oral SFs are used to estimate the risks associated with exposure to carcinogens via ingestion. No SFs are available for the dermal route of exposure, but are instead calculated from oral SFs using the methodology described in Section 4.1.3.

The dose-response data used in this HHBRA for carcinogenic effects, including SF and UR values, are presented in Table 6.1.

4.1.2 Dose-Response Assessment for Noncarcinogenic Effects

In contrast to carcinogens, noncarcinogens are believed to have threshold exposure levels below which adverse effects are not expected. USEPA has derived standards and guidelines based on acceptable levels of exposure for such compounds. Noncarcinogenic effects of concern on which many of the standards and guidelines are based include liver toxicity, reproductive effects, neurotoxicity, teratogenicity, and other chronic toxicities. Various criteria have been developed from experiments that can be used to estimate the dose-response relationship of noncarcinogens. Some of the same uncertainties involved in deriving cancer risk estimates (namely, selection of an appropriate data set and extrapolation of high-dose animal data to low-dose human exposure)

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are also involved in deriving noncarcinogenic dose-response criteria. Dose-response values used most often to evaluate noncarcinogenic effects are reference doses (RfDs).

The RfD, expressed in units of mg/kg/day, is defined as an estimate (with uncertainty spanning perhaps an order of magnitude or greater) of a daily exposure level for the human population, including sensitive subpopulations, that is likely to be without an appreciable risk of deleterious effects during a lifetime (USEPA, 1989). When available, the RfD is the dose-response criterion most appropriate for quantitatively estimating noncarcinogenic effects. The RfD is derived from the following equation:

$$RfD\ (mg/kg/day) = \frac{NOAEL\ or\ LOAEL}{UF\ and/or\ MF}$$

The No Observable Adverse Effect Level (NOAEL) represents the dose of a chemical at which there are no statistically or biologically significant differences in the frequency of an adverse effect between the exposed population and its appropriate control. The Lowest Observable Adverse Effect Level (LOAEL) represents the lowest dose at which a statistically significant difference in the frequency of an effect is noted. Both the NOAEL and the LOAEL are reported in terms of mg/kg/day. An uncertainty factor (UF) of ten per type of uncertainty (e.g., extrapolation from animal sensitivity to human sensitivity, relationship between lowest adverse effect level and no adverse effect level) is used to account for interspecies and interspecies differences, severity of the adverse effect, whether the dose was an NOAEL or an LOAEL, and the adequacy of the data. The magnitude of the UF will therefore vary from chemical to chemical, ranging from 10 to 10,000. A modifying factor (MF), ranging from less than 1 to 10 may also be added to reflect qualitative uncertainties not explicitly addressed in the UFs. The toxicity endpoint upon which the RfD is derived and the UF and/or MF used in the calculation are presented in the dose-response tables. No RfDs are available for the dermal route of exposure but are instead calculated from oral RfDs using the methodology described below (USEPA, 2001b).

The use of chronic RfDs to evaluate the potential for adverse health effects resulting from substantially less-than-lifetime exposures may be overly protective. Subchronic Reference Doses (RfD_ss) have been developed for many chemicals to evaluate the potential noncarcinogenic effects of limited duration exposures. RfD_ss are similar to chronic RfDs; the distinction is the length of exposure duration. When available, RfD_ss/RfC_ss are used in this risk assessment to evaluate noncarcinogenic effects to a construction worker. When RfD_ss are unavailable, chronic RfDs are used to evaluate noncarcinogenic effects for these receptors.

The dose-response data for noncarcinogenic effects (RfDs) and their critical toxic effects are presented in Table 5.1, for both chronic and subchronic effects.

4.1.3 Dermal Dose-Response Values

Cancer SFs and non-cancer RfDs were developed to evaluate risk associated with the dermal contact exposure route. In accordance with USEPA guidance (USEPA, 2001a), dermal dose-response values are calculated from oral dose-response values using an oral absorption factor. The oral absorption factor represents the amount of substance that is absorbed from the gastrointestinal tract following oral administration of a substance. The absorbed dose represents the amount of substance that is potentially available for biological interaction; it is this dose-

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response relationship that the toxicity of a dermally absorbed substance must be evaluated by. Thus, for potentially carcinogenic substances, the dermal dose-response value is calculated as follows:

$$SF_d = SF_{oral} / Oral\ ABS$$

The dermal dose-response value for evaluating non-carcinogenic effects is calculated as follows:

$$RfD_d = RfD_{oral} x\ Oral\ ABS$$

The Oral ABS is the fraction of contaminant absorbed in the gastrointestinal tract (dimensionless) in the critical toxicity study. Chemical-specific Oral ABS values are published by USEPA (USEPA, 2001b). In accordance with USEPA guidance (USEPA, 2001b), oral dose-response values are only adjusted using an Oral ABS value if the COPC has an oral ABS value less than 50%. Otherwise, the oral dose-response value is used as the dermal dose-response value. Dermal SFs and RfDs are presented in Tables 6.1 and 5.1 respectively.

4.1.4 Sources of Dose-Response Values

The following hierarchy of sources for dose-response values has been utilized in identifying dose-response values for this BHHRA.

Tier 1- IRIS (http://www.epa.gov/iris/). In accordance with USEPA guidance, the main source of dose-response values is the USEPA Integrated Risk Information System (IRIS), which is a database established by USEPA containing all validated data on many toxic substances found at hazardous waste Sites. This database was used to identify the SFs and RfDs applied in this risk assessment (USEPA, 2003a).

Tier 2- NCEA's peer reviewed toxicity values (PRTVs). NCEA's PRTVs are developed by the Superfund Technical Support Center (STSC) for the EPA Superfund program. STSC's reassessment of HEAST toxicity values, as well as development of PRTVs in response to Regional or Headquarters Superfund program requests, are consistent with Agency practices on toxicity value development, use the most recent scientific literature, and are supported by both internal and external peer review, providing a high level of confidence in the use of these values in the Superfund Program. USEPA Region I has provided PRTVs and associated documentation prepared by the STSC for aluminum, copper, 2-methylnapthalene, and 4-nitrophenol (USEPA, 2003b).

Tier 3 - Other toxicity values

- Cal EPA's toxicity values. Cal EPA develops toxicity values for both cancer and non-cancer effects. Cal EPA toxicity values are obtained on the Cal EPA website at http://www.oehha.ca.gov/risk/chemicalDB//index.asp.

- ATSDR's MRLs address non-cancer effects only, and are available on the ATSDR website at http://www.atsdr.cdc.gov/mrls.html.

- Toxicity values remaining in current versions of HEAST (1997a).

In this HHBRA, the majority of dose-response values used are published in IRIS. For some Site-related COPCs required dose-response data are only available as NCEA provisional values or from CAL-EPA. These dose-response values were used in this HHBRA in order to provide a more complete evaluation of potential risks.

For tetrachloroethylene, the toxicity values from Cal EPA and not the NCEA provisional values have been utilized.

Uncertainties related to the absence of dose-response data, particularly for COPCs for which the exposure pathway, which represents the only pathway or most significant exposure pathway, has no toxicity criterion, will be discussed in the risk assessment uncertainty analysis.

Evaluation of Dioxin-Like Compounds in Site Media

Due to the limited toxicological data available for many individual dioxin, furan, coplanar PCB congeners, and HCX, and to simplify the risk assessment process, a methodology has been developed that quantifies the toxicities of various dioxin, furan, and coplanar PCB congeners relative to the toxicity of 2,3,7,8-TCDD (Van den Berg et al., 1998). TCDD is widely accepted to be the most toxicological significant chemical among these groups of chemicals, all of whose toxicological properties are assumed to be regulated by their individual abilities to bind to the cytosolic Ah receptor (AhR).

Based on the potency of 2,3,7,8-TCDD, and the greater amount of research that has been devoted to 2,3,7,8-TCDD, a CSF has only been developed for this congener. Other dioxin, furan, and co-planar PCB compounds exert toxicity through the same mechanism of action as 2,3,7,8-TCDD, but the threshold effects levels for the other compounds are directly related to their affinity to interact with the AhR. Therefore, dioxin, furan, and co-planar PCB compounds (dioxin-like compounds) are evaluated using the dose-response data for 2,3,7,8-TCDD, but the concentrations are weighted according to their potency relative to the 2,3,7,8-TCDD using Toxicity Equivalent Factors (TEFs).

The procedure for weighting the concentrations of dioxin-like compounds is documented in Appendix D. In summary, since 2,3,7,8-TCDD has the greatest affinity for the AhR, it is arbitrarily assigned a TEF of 1. Other congeners are assigned a TCDD TEF relative to 2,3,7,8-TCDD based on experimental evidence concerning their relative binding potential to the AhR. The potency of the congener is then estimated by multiplying the measured media concentration by the TEF for the particular congener to yield a toxic equivalent quotient (TEQ). Finally, a TEQ for the entire sample can be determined by summing the calculated TEQs for each AhR binding congener; the resulting concentration is a measure of the potency of the entire mixture represented in terms of 2,3,7,8-TCDD, and is expressed as a TCDD-equivalent concentration. This methodology assumes that the combined effects of the different congeners are dose or concentration additive, and this has been generally supported by results of many studies. This approach fails to consider the toxicological significance of effects that are not mediated by the Ah receptor (e.g., neurotoxicity and various hormonal effects). However, current consensus is that the TEF approach is the best methodology for assessing the impacts associated with exposure to dioxin-like compounds (Van den Berg et al., 1998).

The CSF for 2,3,7,8-TCDD of 1.5E+05 per mg/kg/day (USEPA, 1997) is used to evaluate the potential risks associated with dioxin-like compounds. Specifically, this CSF is applied to the

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calculated intakes for the dioxin-TEQ (sum of TEQs for individual dioxin and furan compounds) and the co-planar PCB TEQ (sum of TEQs for individual co-planar PCB congeners). The TEFs for dioxin-like PCBs, polychlorinated dibenzodioxins (PCDDs) and polychlorinated dibenzofurans (PCDFs) identified for mammals (Van den Berg et. al. 1998) are applied to the fish tissue, surface water, and sediment EPCs for each of the congeners to identify a toxic equivalence concentration (TEQ). The TEQ are used in conjunction with the oral cancer SF for 2,3,7,8-TCDD (available in HEAST, USEPA 1997) to estimate cancer risk for those compounds. The dioxin/furan TEQ has been kept distinct from the co-planar PCB TEQ for purposes of risk calculations.

The dioxin-like PCB congeners evaluated in this manner include congeners 105, 114, 118, 123, 156, 157, 167, 189, 81, 77, 126, and 169. The remainder of the reported PCB congeners are not evaluated in a quantitative manner. Rather, they are evaluated, by inclusion, with the Aroclor-1254 or Aroclor-1260 using the High Risk and Persistence, Upper Bound cancer SF for PCBs. Aroclor-1254 and Aroclor-1260 exposure point concentrations have not been adjusted (reduced) to account for the twelve co-planar PCB congeners that are evaluated using the dioxin TEF approach. Therefore, there is an overestimation of cancer risk associated with these co-planar PCB congeners. The Reference Dose (Aroclor 1254) obtained from the IRIS database is used to evaluate non-cancer risks associated with Aroclors.

The compound HCX has been identified as a dioxin-like compound. However, there is no formally published USEPA CSF, RfD, or dioxin TEF for HCX. The HCX toxicity information in the literature is limited. USEPA has decided to evaluate the risks associated with HCX in an appendix because: none of the available studies documents long-term toxicity of HCX; there is limited information concerning metabolites of HCX; there are many uncertainties associated with the available study reports; HCX concentrations in fish tissue are relatively low; and the Hahn study report, which suggests a dioxin TEF, is a preliminary, unpublished report. Appendix H contains an exposure assessment, toxicity assessment, and risk characterization for HCX. A TEF of 0.0002 has been applied for HCX in Appendix H. In addition, there is discussion in Appendix H of the potential impact of the uncertainty in this toxicity value on the results of the risk assessment.

DIOXIN REASSESSMENT

EPA recently reviewed available toxicity studies on 2,3,7,8-TCDD and other dioxin-like compounds. A preliminary draft document, *Exposure and Human Health Reassessment of 2,3,7,8-Tetrachlorodibenzo-p-Dioxin (TCDD) and Related Compounds*, (EPA, 2000) presents EPA's scientific reassessment of the health risks resulting from exposure to these compounds. This draft document has been reviewed by the public and the USEPA's Science Advisory Board (SAB) in its publication, *Dioxin Reassessment – An SAB Review of the Office of Research and Development's Reassessment of Dioxin*, (EPA-SAB, 2001). At this time, the dioxin reassessment document and its contents remain in draft status. The draft reassessment document draws some important conclusions and makes recommendations concerning health risk assessment for dioxins and furans.

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Cancer Effects

In its review of available toxicity studies on 2,3,7,8-TCDD and other dioxin-like compounds (EPA 2000), USEPA recommended a revised CSF of 1E+06 $(mg/kg\text{-}d)^{-1}$ to estimate upper-bound cancer risk for background intakes and incremental intakes above background. This estimate compares well with the published estimates of cancer slope and risk from epidemiological studies by Becher et al., 1998 (99-1299) and Steenland et al., 2001 (99-1302) on the Hamburg and NOISH cohorts. Use of the recommended CSF (EPA 2000) would result in an approximately 7-fold increase in the cancer risk estimates based on the current upper-bound slope factor (1.56 E+05) associated with 2,3,7,8-TCDD and other dioxin-like compounds.

The epidemiological literature suggests an association with increases in all cancers combined, in respiratory tumors, and possibly in soft tissue sarcoma. EPA found that a weight-of-evidence evaluation suggests that mixtures of 2,3,7,8-TCDD and other dioxin-like PCDDs, PCDFs, and PCBs are strong cancer promoters and weak direct or indirect tumor initiators.

Based on the most sensitive cancer responses in animal and human studies, EPA estimated CSFs ranging from approximately 1E+06 to 9E+06 $(mg/kg\text{-}d)^{-1}$. EPA estimated an upper-bound CSF of 1E+06 $(mg/kg\text{-}d)^{-1}$ based on human data from a meta-analysis of three occupational cohorts, and an upper bound CSF of 1.4E+06 $(mg/kg\text{-}d)^{-1}$ based on animal data. Other analyses of these data have recently been published (Starr, 2001, 99-1301 and Crump, 2003, 99-1300). The shape of the low-dose exposure response relationship could not be determined from available data. Therefore, EPA used a linear dose extrapolation model to derive upper-bound CSF estimates.

Non-cancer Effects

EPA (2000) evaluated the "margin-of-exposure" (MOE), for several toxicology studies on non-cancer effects (DeVito et al., 1995; Gray et al., 1997; Mabley et al., 1992a,b,c; Murray et al., 1979; Narashimhan et al., 1994; Rier et al., 1993; Schantz et al., 1992; Schrenk et al., 1994; Sewall and Lucier, 1995; Smialowicz et al., 1994; Van Birgelen et al., 1995; Vecchi et al., 1983 Vogel et al., 1997; Walker et al., 1999). MOE is defined here as the ratio of the effect level in the comparison species to the current background human body burden. The effect level in the comparison species pertains to the body burden in laboratory species that results in some low level effect, such as a LOAEL, or the ED01 (the effective dose at which 1% of the tested population shows the effect in question). For the most sensitive endpoints identified, MOEs were found to range from, for example, less than one for enzyme induction in mice and rats, less than four for developmental effects and to four for endometriosis in nonhuman primates. In other words, the body burden in the laboratory species that showed a particular effect was only four times (and less) higher than the current body burden in humans. In evaluating MOEs, consideration should be given to uncertainties in distinguishing between adaptive biochemical changes and adverse effects, both on an individual level and as these changes impact whole populations. Children's non-cancer risks from dioxin and related compounds may be greater than for adults, but more data are needed to fully address this issue.

A reference dose, RfD, for dioxin-like compounds has not been developed. Further, EPA (2000) concluded that a reference dose for dioxin calculated in the manner typical of the way EPA determines RfDs would result in a dose that is significantly lower than current average background doses. RfDs are used primarily to evaluate increments of exposure from specific

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sources when background exposures are low and insignificant, and background exposures are not insignificant as indicated by the MOE discussion above.

This assessment quantifies non-cancer effects using RfDs to calculate hazard quotients and hazard indices. Because an RfD has not been developed for PCDDs and PCDFs, the potential for non-cancer effects from exposure to dioxin-like compounds is not quantitatively evaluated in this assessment.

In Appendix E, alternative cancer risk estimates for the biota consumption pathway are provided. The alternative cancer risk estimates are calculated in the same manner as discussed above, but the 2,3,7,8-TCDD CSF used in these calculations is the draft value ($1x10^{6}$/(mg/kg/day) presented in the 2000 Dioxin Reassessment Document (USEPA, 2000c). This information is intended to provide additional perspective for the cancer risks presented in the BHHRA.

Evaluation of Chromium in Site Media

The most common forms of chromium in environmental media are chromium III (trivalent chromium) and chromium VI (hexavalent chromium). Although chromium was detected in Site media, no speciation analyses were performed. To provide a conservative assessment of toxicity and health risks associated with potential exposures to chromium, chromium data was evaluated as hexavalent chromium in this risk assessment.

Exposures to hexavalent chromium have been associated with chronic non-cancer health effects via ingestion, dermal contact, and inhalation, and allergic contact dermatitis via direct dermal contact with hexavalent-chromium containing materials. The chronic oral RfD of 0.003 mg/kg/day (IRIS, 2001d) and the subchronic oral RfD of 0.02 mg/kg/day (HEAST, 1997c) have been applied to the estimated daily doses in order to characterize potential non-cancer risks for ingestion and dermal contact with soil. Hexavalent chromium RfCs are published for chromium particulates and chromic acid mists.

Evaluation of Lead in Site Media

No dose-response values are published for potential exposures to lead. In the absence of dose-response data, USEPA recommends use of lead biokinetic uptake models to evaluate potential lead exposures, and comparison of the lead intake estimated using the models to threshold blood lead levels for children and adults. Lead is screened as a potential COPC by comparing the maximum detected concentration to USEPA's Interim Soil Lead Screening Value of 400 mg/kg (USEPA, 1994), which is considered by USEPA to be protective for residential exposures to lead in soil. The OSWER screening values are used to evaluate potential risks associated with lead exposure at these areas. Lead EPCs in soil or sediment were greater than 400 mg/kg only in sediment in Allendale Pond. The evaluation of those sediments via the biokinetic uptake model is presented in Appendix E. Lead is screened as a potential COPC in surface water by comparing the maximum detected concentration to the USEPA's action level of 0.015 mg/l (USEPA, 2002d).

Lead exposure associated with fish consumption was not quantified and lead risks associated with fish consumption were not quantified in the BHHRA. The health effects associated with lead exposure are generally considered to be correlated to levels of lead in the blood. Because of this, the typical hazard index calculated as the ratio of a daily intake and an oral Reference Dose is not applicable to lead exposure assessment and risk assessment. Currently, models are used to

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evaluate lead exposures via various media and exposure routes. These models estimate, from the identified intakes, equilibrium blood lead levels and the probability of exceedance of identified blood lead level thresholds.

The USEPA's Integrated Exposure Uptake Biokinetic (IEUBK) Model and the Adult Lead Model are commonly utilized models for lead exposure assessment. The blood lead levels generated by these models are used to estimate the probability that specific thresholds for children (who are more sensitive to lead exposure) and for adults would be exceeded. While these models do not contain a component specifically designed to consider fish consumption, it would be possible to incorporate fish consumption into the models by assuming that some portion of the receptors meat consumption consists of fish. USEPA Region 10 released, in 2002, the report *Columbia River Basin Fish Contaminant Survey 1996-1998*, EPA 910-R-02-006 (USEPA, 2002c). In that report, risks associated with lead in fish tissue was evaluated by substituting fish consumption for a portion of the meat diet in the USEPA's IEUBK Model and the Adult Lead Model. That report concluded that lead levels in fish analyzed from the Columbia River are unlikely to cause a blood lead level greater than 10 ug/deciliter. The highest lead concentration reported in fish tissue samples in this study was 1,200 ug/kg. There is less than 5% chance that the threshold would be exceeded in children at lead concentrations in fish less than 500 ug/kg. Also, there is less than 5% chance that fetal blood lead levels would exceed 10 ug/deciliter at lead concentrations in fish less than 700 ug/kg. The IEUBK Model used a child fish consumption rate of 16.2 gram/day.

The child fish consumption rate utilized in the BHHRA is lower (4.7 gram/day for RME and 3.0 gram/day for CT) than that used for the Columbia River Study. The mean lead concentrations in fish tissue (largemouth bass, American eel, and white sucker) for each of the Site exposure points are below 500 ug/kg. The mean lead concentrations in largemouth bass fillet samples are 6.6 ug/kg and 11.3 ug/kg for Lyman Mill Pond and Manton Pond respectively. Mean concentrations of lead in American eel samples are 428 ug/kg, 270 ug/kg, and 341 ug/kg respectively for Allendale Pond, Lyman Mill Pond, and for Dyerville Pond. For white sucker, the mean lead concentrations are 160 ug/kg and 428 ug/kg for Allendale Pond and Lyman Mill Pond respectively. That would suggest that if the models were applied to this Site, the lead in fish tissue would be unlikely to result in blood lead levels in children or fetuses that are above thresholds of concern.

Evaluation of Mercury in Site Media

Mercury may exist as elemental mercury, inorganic mercury salts, and organic mercury. Typically, mercury is present in environmental media as inorganic mercury salts or organic mercury (methyl mercury) that may be produced by bacterial methylation of inorganic mercury. Methyl mercury is known to bioaccumulate. In this HHBRA, mercury detected in fish is evaluated using oral dose-response values for organic mercury (methyl mercury), whereas mercury detected in other media (e.g., soil) is evaluated using oral dose-response values for inorganic mercury.

Descriptive Toxicity Summaries for Chemicals of Concern

Brief toxicity profiles for selected COPCs are presented in Appendix D to this report.

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5.0 RISK CHARACTERIZATION

5.1 RISK CHARACTERIZATION

The final step of the risk assessment is the risk characterization. This step involves the integration of the exposure and toxicity assessment into quantitative expressions of potential human health risks associated with COPC exposure. Quantitative estimates of both carcinogenic and noncarcinogenic risks are made for each COPC and each exposure point. Risks associated with RME exposure scenarios and CT exposure scenarios are calculated separately.

Cancer Risks

Carcinogenic risks associated with exposure to individual chemicals are estimated by multiplying the chemical intake for each carcinogen by its CSF. This value represents an upper bound of the probability of an individual developing cancer over a lifetime as the result of exposure to a chemical. For each receptor and each exposure pathway (exposure to a specific medium) such as consumption of a specific fish species, the chemical-specific risks for all carcinogenic compounds will be summed to determine the lifetime cancer risk for that receptor for that medium. The following equations are used to estimate the chemical- and pathway-specific cancer risks.

Chemical-Specific Excess Lifetime Cancer Risk:

$$Risk_i = CDI_i \times CSF_i$$

where:

$Risk_i$	=	unitless probability of an individual developing cancer as the result of exposure to a chemical i
CDI_i	=	chronic daily intake of chemical i averaged over 70 years (mg/kg-day)
CSF_i	=	USEPA cancer slope factor for chemical i $(mg/kg\text{-}day)^{-1}$

According to RAGS Part A, p. 8-6 and 8-11, if the estimated risk is equal to or greater than 0.01, an alternative approach (one-hit equation for high carcinogenic risk levels) for calculating cancer risk should be used:

$$Risk_i = 1 - e^{(-CDI_i \times CSF_i)}$$

where:

$Risk_i$	=	unitless probability of an individual developing cancer as the result of exposure to a chemical i
CDI_i	=	chronic daily intake of chemical i averaged over 70 years (mg/kg-day)
CSF_i	=	USEPA cancer slope factor for chemical i $(mg/kg\text{-}day)^{-1}$
e	=	exponent (value of 2.1817)

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Pathway-Specific Incremental Lifetime Cancer Risk:

$$Risk_T = \Sigma\ Risk_i$$

where:

$Risk_T$	=	unitless probability of an individual developing cancer as the result of multiple chemical exposures
$Risk_i$	=	unitless cancer risk estimate for a single chemical associated with biota consumption

The results from the carcinogenic risk assessment are compared to acceptable risk ranges established by the USEPA. The USEPA's guidelines, established in the National Hazardous Substances and Pollution Contingency Plan (NCP) identify acceptable exposure levels as those concentration levels "that represent an excess upper bound lifetime cancer risk to an individual of between 10^{-4} and 10^{-6} using information on the relationship between dose and response" (USEPA 1990).

Non-cancer Risks

Non-cancer risk estimates will be calculated by dividing specific chemical intake by the appropriate RfD. The result is called the hazard quotient (HQ). The HQs for individual compounds within an exposure pathway are summed to obtain the hazard index (HI) for that particular pathway.

Following are the equations used to determine the HQs and HIs.

The following equation is used to determine the hazard quotient:

$$HQ_i = \frac{I_i}{RfD_i}$$

where:

HQi	=	hazard quotient of chemical i
Ii	=	intake of chemical i averaged over the exposure period (mg/kg-day)
RfDi	=	reference dose for chemical i corresponding to the same exposure duration as the intake (mg/kg-day)

The following equation is used to determine the hazard index:

$$HI = \Sigma\ HQ_i$$

where:

HI	=	potential for noncarcinogenic effects from multiple chemical exposures

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HQi = hazard quotient for each chemical associated with biota consumption

An HI of less than 1 indicates that noncarcinogenic toxic effects are unlikely. An HI greater than 1 indicates a greater possibility of a noncarcinogenic toxic effect occurring, but the circumstances must be evaluated on a case-by-case basis. Generally, as the HI increases, so does the likelihood that adverse effects might be associated with exposure. However, the relationship between increased risk and larger HI values may not be linear.

Calculation of Non-cancer and Cancer Risks

RAGS Part D Table 7s are used to present the risk calculations. In simplistic terms, for a given receptor/age-group, cancer risks are calculated for each chemical in each medium (e.g., sediment) and exposure route (e.g., ingestion, dermal contact). Risks across exposure routes are summed to yield the risk for that medium. Cancer risks associated with all exposure media for that receptor/age-group are summed to yield the cumulative receptor cancer risk for that receptor/age-group. For a given receptor (by age-group), the non-cancer Hazard Index is calculated for each chemical, and exposure route for a given medium. Hazard Index values associated with all exposure media for each receptor/age-group are summed to yield the screening cumulative Hazard Index for that receptor/age-group. This summing of Hazard Index values across chemicals and exposure media is a conservative screening approach; because chemicals can have different target organs, non-cancer risks are not necessarily additive.

The calculated RME and CT cancer risks and Hazard Index values for the reference area exposure point (Assapumpset Pond and Brook), the upstream background area (Greystone Mill Pond), the four Site exposure points for the visiting recreational angler and the resident living along the river, and workers at the Fogarty Center are presented in Tables 7.1.RME through 7.37.RME and Tables 7.1.CT through 7.37.CT. The RAGS Part D Table 7s document the risk calculations by identifying the COPCs, exposure point concentrations, daily chemical intakes by chemical for both cancer and non-cancer endpoints, the CSFs and RfDs, and the calculated cancer risk and Hazard Quotient for each chemical in each exposure medium. Further, the Table 7s present summed risks for each medium/exposure route combination, each medium, and for all exposure media combined.

Table 7s are presented in the order discussed below. The CT risks are presented in Tables 7.1.CT through 7.37.CT. The tables are presented by exposure point (Assapumpset Brook and Pond, Greystone Mill Pond, Allendale Pond, Lyman Mill Pond, Manton Pond, Dyerville Pond, and the Fogarty Center). Each of the scenarios evaluated has been identified as both current and future exposure scenarios. For each receptor group, each of the age groups is presented in a separate table.

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The RME risk calculation spreadsheets are presented in the following order.

ASSAPUMPSET BROOK AND POND

- Visiting recreational angler at Assapumpset Brook and Pond – adult, older child, child (consumption of fish only for child) – surface water, sediment, consumption of combined fish diet. Tables 7.1.RME through 7.3.RME.
- Resident living along the Woonasquatucket River at Assapumpset Brook and Pond - adult, older child, child – surface water, sediment, consumption of combined fish diet. Tables 7.4.RME through 7.6.RME.

GREYSTONE MILL POND

- Visiting recreational angler at Greystone Mill Pond – adult, older child, child (consumption of fish only for child) – bank soil, surface water, sediment, consumption of combined fish diet. Tables 7.7.RME through 7.9.RME.
- Resident living along the Woonasquatucket River at Greystone Mill Pond – adult, older child, child – bank soil, surface water, sediment, consumption of combined fish diet. Tables 7.10.RME through 7.12RME.

ALLENDALE POND

- Visiting recreational angler at Allendale Pond – adult, older child, child (consumption of fish only for child) – bank soil, surface water, sediment, consumption of combined fish diet. Tables 7.13.RME through 7.15.RME.
- Resident living along the Woonasquatucket River at Allendale Pond – adult, older child, child – bank soil, surface water, sediment, consumption of combined fish diet. Tables 7.16.RME through 7.18.RME.

LYMAN MILL POND

- Visiting recreational angler at Lyman Mill Pond – adult, older child, child (consumption of fish only for child) – surface water, sediment, consumption of combined fish diet. Tables 7.19.RME through 7.21.RME.
- Resident living along the Woonasquatucket River at Lyman Mill Pond – adult, older child, child – surface water, sediment, consumption of combined fish diet. Tables 7.22.RME through 7.24.RME.

MANTON REACH

- Visiting recreational angler at Manton Reach – adult, older child, child (consumption of fish only for child) – surface water, sediment, consumption of combined fish diet. Tables 7.25.RME through 7.27.RME.
- Resident living along the Woonasquatucket River at Manton Reach – adult, older child, child – surface water, sediment, consumption of combined fish diet. Tables 7.28.RME through 7.30.RME.

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DYERVILLE REACH

- Visiting recreational angler at Dyerville Reach – adult, older child, child (consumption of fish only for child) – surface water, sediment, consumption of combined fish diet. Tables 7.31.RME through 7.33.RME.
- Resident living along the Woonasquatucket River at Dyerville Reach – adult, older child, child – surface water, sediment, consumption of combined fish diet. Tables 7.34.RME through 7.36.RME.

FOGARTY CENTER

- Adult worker at Fogarty Center – adult - incidental ingestion and dermal contact with surface soil. Table 7.37.RME.

The CT risk calculation spreadsheets are presented in the same order with the same numbering scheme as the RME spreadsheets, but the table numbers contain "CT" rather than "RME (e.g., Table 7.1.CT).

There are no RAGS Part D Table 8s for this BHHRA. Table 8s are specifically for the calculation of radiological risks. No radiological COPCs were identified in this assessment. Therefore, no Table 8s are required.

Summary of Non-cancer and Cancer Risks

RAGS Part D Table 9s summarize the information that is documented in the Table 7s. In addition, consistent with USEPA guidance (USEPA, 1989, 2001c), the Table 9s segregate the Hazard Index calculations by target organ system, and calculate a Hazard Index for each target organ system. This presentation of the Hazard Index calculations is an enhancement of the screening Hazard Index calculations presented previously in the Table 7s. RAGS Part D Tables 9.1.RME through 9.37.RME and Tables 9.1.CT through 9.37.CT summarize the risk calculations that are documented in the Table 7s that were discussed previously.

Alternate Cancer Risk Calculation Method

Consistent with USEPA guidance (USEPA, 1989), if the cancer risk calculated by the standard linear dose-response model is greater than 0.01, cancer risks should be recalculated using the one-hit model presented in Section 5.1. None of the calculated cancer risks are greater than 0.01 for any exposure pathways either individually or combined. Therefore, it is not necessary to recalculate cancer risks using the one-hit model.

Identification of Chemicals of Concern

RAGS Part D Table 10s identify, for each of the three age groups for the Visiting Recreational Angler and the Resident Living Along the Woonasquatucket River at Allendale Pond, Lyman Mill Pond, Manton Reach, and Dyerville Reach and the Industrial/Commercial Worker at the Fogarty Center, those chemicals that are considered chemicals of concern (COCs). COCs are those substances that have associated cancer risk greater than one in one-million (1×10^{-6}) and/or

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a Hazard Index greater than one for a given medium/receptor/age-group combination. COCs are those substances that would typically be considered in the Feasibility Study process. Tables 10.13.RME through 10.37.RME and Tables 10.13.CT through 10.37.CT identify the COCs and the risks associated with each of the COCs in each medium for the Visiting Recreational Angler and Resident Living Along the River at each of the four river-related exposure points. These tables are numbered as such to indicate that they correspond with the risk calculations that are summarized in Tables 7.13.RME through 7.36.RME and Tables 7.13.CT through 7.36.CT respectively. There are no table 10s for Assapumpset Pond and Brook nor for Greystone Mill Pond, which are the reference area and background area respectively. Those areas are addressed by Tables 7.1.RME through 7.12.RME and 7.1.CT through 7.12.CT respectively. COCs were identified for the Fogarty Center. However, the total receptor cancer risk for that location is within the Superfund cancer risk range and the Hazard Index is below one.

The COCs for RME scenarios are identified and summarized by exposure point, medium, and receptor in the text below.

COCs at Allendale Pond (Tables 10.13.RME through 10.18.RME)

Visiting Recreational Angler

- Sediment COCs include dioxins and furans (primarily 2,3,7,8-TCDD).
- Combined Fish Diet COCs include Aroclor-1254, Aroclor-1268, dieldrin, technical chlordane, dioxin-like PCBs, and dioxins and furans (primarily 2,3,7,8-TCDD).

Resident Living Along the Woonasquatucket River

- Bank soil COCs include 2,3,7,8-TCDD.
- Sediment COCs include benzo(a)pyrene, dibenz(a,h)anthracene, arsenic and dioxins and furans (primarily 2,3,7,8-TCDD).
- Combined fish diet COCs include Aroclor-1254, Aroclor-1268, dieldrin, technical chlordane, dioxin-like PCBs, and dioxins and furans (primarily 2,3,7,8-TCDD).

COCs at Lyman Mill Pond (Tables 10.19.RME through 10.24.RME)

Visiting Recreational Angler

- Sediment COCs include dioxins and furans (primarily 2,3,7,8-TCDD).
- Combined Fish Diet COCs include benzo(a)pyrene, Aroclor-1254, Aroclor-1268, dieldrin, technical chlordane, arsenic, dioxin-like PCBs, and dioxins and furans (primarily 2,3,7,8-TCDD).

Resident Living Along the Woonasquatucket River

- Sediment COCs include benzo(a)pyrene, dibenz(a,h)anthracene, N-nitroso-di-n-propylamine, arsenic, dioxins and furans (primarily 2,3,7,8-TCDD), and dioxin-like PCBs.
- Combined fish diet COCs include benzo(a)pyrene, Aroclor-1254, Aroclor-1268, dieldrin, technical chlordane, dioxin-like PCBs, and dioxins and furans (primarily 2,3,7,8-TCDD).

COCs at Manton Reach (Table 10.25.RME through 10.30.RME)

Visiting Recreational Angler

- Sediment COCs include dioxins and furans (primarily 2,3,7,8-TCDD).

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- Combined Fish Diet COCs include 4,4'-DDE, Aroclor-1254, Aroclor-1268, dieldrin, technical chlordane, methyl mercury, and dioxins and furans (primarily 2,3,7,8-TCDD).

Resident Living Along the Woonasquatucket River

- Sediment COCs include dioxins and furans (primarily 2,3,7,8-TCDD).
- Combined fish diet COCs include Aroclor-1254, Aroclor-1268, technical chlordane, methyl mercury, and dioxins and furans (primarily 2,3,7,8-TCDD).

COCs at Dyerville Reach (Table 10.31.RME through 10.37.RME)

Visiting Recreational Angler

- No sediment COCs were identified.
- Combined Fish Diet COCs include Aroclor-1254, dieldrin, technical chlordane, and dioxins and furans (primarily 2,3,7,8-TCDD).

Resident Living Along the Woonasquatucket River

- Sediment COCs include dioxins and furans and benzo(a)pyrene.
- Combined fish diet COCs include Aroclor-1254, dieldrin, technical chlordane, and dioxins and furans (primarily 2,3,7,8-TCDD).

COCs at the Fogarty Center

Commercial/Industrial Worker

- Surface soil COCs include benzo(a)pyrene and arsenic.

All of the COCs identified in sediment, soil and biota at Allendale Pond and Lyman Mill Pond were identified as COCs based on cancer risk and one COC for biota, Aroclor-1254, was also identified as a COC based on non-cancer risk. In Manton Reach, dioxins and furans, 4,4'-DDE, dieldrin, and technical chlordane were identified as COCs based on cancer risk. In Manton Reach, Aroclor-1254 and Aroclor-1268 were identified as COCs in biota based on both cancer risk and non-cancer risk. In Manton Reach, methyl mercury was identified as a COC in biota based on non-cancer risk only. In Dyerville Reach, dioxins and furans (sediment, surface water, and biota), benzo(a)pyrene (sediment only), dieldrin (biota only), and technical chlordane (biota only) were identified as COCs based on cancer risk only. Aroclor-1254 was identified as a COC in biota in Dyerville Reach based on both cancer risk and non-cancer risk. At the Fogarty Center, benzo(a)pyrene and arsenic were identified as COCs based on cancer risk only.

At the Fogarty Center, the total receptor cancer risk associated with potential surface soil exposure is less than one in ten thousand and the non-cancer hazard index is less than one for both RME and CT scenarios. No COCs are identified for surface soil at the Fogarty Center for the CT scenario. For the RME scenario at the Fogarty Center, arsenic and benzo(a)pyrene were identified as COCs for surface soil based on cancer risk only. The maximum reported dioxin/furan TEQ for surface soil at the Fogarty Center is 0.0066 ug/kg (0.0066 ppb). This maximum concentration is below the 1 ppb action level for residential soils as identified in the Action Memorandum for the Non-Time Critical Removal Action (USEPA, 2001a).

COCs were also identified for CT scenarios as shown in Tables 10.13.CT through 10.37.CT.

Summary of Calculation of Receptor Risks

Tables 11.1.RME and 11.1.CT are risk summary tables. These tables present the medium-specific and route-specific risks for each receptor/age-group combination. In addition, Tables 12

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and 13 present total receptor non-cancer and cancer risks (all age groups combined for cancer risk) for each medium for each receptor at each exposure point and for all exposure media combined. Tables 12 and 13 also show the incremental risks (the difference between the total calculated risk and the risk at the background location, Greystone Mill Pond).

In addition to the RAGS Part D Table 7s and 9s, Figures 51 through 58 graphically summarize the RME and CT risk calculations. For each receptor group, there are two RME figures and two CT figures. One RME figure summarizes for one receptor group the non-cancer risk by exposure medium for each of the four river exposure points, Greystone Mill Pond, and Assapumpset Pond and Brook, while the other RME figure summarizes cancer risk by exposure medium for each of the four exposure points, Greystone Mill Pond, and Assapumpset Pond and Brook. The two CT figures also present first, non-cancer, and then cancer risks.

It is apparent from viewing these figures that among the exposure media scenarios, the fish consumption scenario is associated with the highest non-cancer risks for all receptors at all exposure points (except the industrial/commercial worker at the Fogarty Center, for which biota consumption was not evaluated).

As shown in Table 12, the Hazard Index for the RME and CT scenarios for the Commercial/Industrial Employee of the Fogarty Center are well below one. These Hazard Index values are based upon surface soil exposure.

As shown in Table 12, the total (all media combined) receptor Hazard Index is equal to or greater than one for the Resident Living Along the River and the Visiting Recreational Angler (at least one age group) at all four exposure areas (EA1, EA2, EA3, and EA4) and at Greystone Mill Pond and Assapumpset Pond and Brook for the RME and CT (excluding Assapumpset Pond and Brook) scenarios. The total (all media combined) Incremental Hazard Index, as shown in Table 12, has been calculated as the difference between the total receptor Hazard Index calculated at a given exposure point and the corresponding total receptor Hazard Index calculated for the same receptor at the background location, Greystone Mill Pond. The Incremental Hazard Index (in other words, the site-related Hazard Index) is equal to or above one for the Resident Living Along the River and the Visiting Recreational Angler (at least one age group) at three of the four river exposure areas (Manton Reach is the exception) for the RME and CT scenarios. The consumption of fish is by far the largest contributor to the total Incremental Hazard Index for both receptors at all four exposure points. For the RME and CT scenarios and the four river exposure points, no other medium has a Hazard Index greater than one.

As shown in Table 12, for the RME scenarios, the Incremental Hazard Index for both the Resident Living Along the River and the Visiting Recreational Angler show a similar trend among exposure points. Based on Incremental Hazard Index, the exposure points ordered from highest to lowest are Lyman Mill Pond, Allendale Pond, Dyerville Reach, and Manton Reach. For the Resident Living Along the River (child), the RME Incremental Hazard Index values are 27, 24, 1, and zero (the risk is equal to the risk at Greystone Mill Pond) for Lyman Mill Pond, Allendale Pond, Dyerville Reach, and Manton Reach, and respectively. For the Visiting Recreational Angler, the RME Incremental Hazard Index (child) values are 28, 25, 2, and zero (the risk is equal to the risk at Greystone Mill Pond) for Lyman Mill Pond, Allendale Pond, Dyerville Reach, and Manton Reach respectively.

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As shown in Table 12, for the CT scenarios, the Incremental Hazard Index for both the Resident Living Along the River and the Visiting Recreational Angler show a similar trend among exposure points. Based on Incremental Hazard Index, the exposure points ordered from highest to lowest are Lyman Mill Pond, Allendale Pond, Dyerville Reach, and Manton Reach. For the Resident Living Along the River (child), the RME Incremental Hazard Index values are 18, 16, 0.5, and zero (the risk is equal to the risk at Greystone Mill Pond) for Lyman Mill Pond, Allendale Pond, Dyerville Reach, and Manton Reach, and respectively. For the Visiting Recreational Angler, the RME Incremental Hazard Index (child) values are 18, 16, 0.7, and zero (the risk is equal to the risk at Greystone Mill Pond) for Lyman Mill Pond, Allendale Pond, Dyerville Reach, and Manton Reach respectively.

For both receptors, for all four river exposure points, and for both RME and CT scenarios, the largest contributor to incremental Hazard Index is Aroclor 1254 in fish tissue and the associated fish consumption exposure scenario.

As shown in Table 13, the Cancer Risk for the RME and CT scenarios for the Commercial/Industrial Employee of the Fogarty Center are well below one in ten thousand. The cancer risks are based upon surface soil exposure.

As shown in Table 13, the RME total (all media combined) receptor Cancer Risk is equal to or greater than one in ten thousand (the upper end of the Superfund risk range) for the Resident Living Along the River at all four river exposure areas (EA1, EA2, EA3, and EA4) and at Greystone Mill Pond and for the Visiting Recreational Angler at all four river exposure areas (but not at Greystone Mill Pond nor at Assapumpset Pond and Brook). The CT total (all media combined) receptor Cancer Risk is equal to or greater than one in ten thousand (the upper end of the Superfund risk range) for the Resident Living Along the River and the Visiting Recreational Angler at all four river exposure areas (EA1, EA2, EA3, and EA4). The CT total (all media combined) receptor Cancer Risk is less than one in ten thousand (the upper end of the Superfund risk range) for the Resident Living Along the River and the Visiting Recreational Angler at Greystone Mill Pond and Assapumpset Pond and Brook.

The total (all media combined) Incremental Cancer Risk, as shown in Table 13, has been calculated as the difference between the total receptor cancer risk calculated at a given exposure point and the corresponding total receptor cancer risk calculated for the same receptor at the background location, Greystone Mill Pond. The Incremental Cancer Risk (in other words, the site-related cancer risk) is equal to or above one in ten thousand for the Resident Living Along the River and the Visiting Recreational Angler at all four river exposure areas for the RME and CT scenarios.

As shown in Table 13, the RME Incremental Cancer Risks for the Resident Living Along the River for Allendale Pond, Lyman Mill Pond, Manton Reach, and Dyerville Reach are similar at 5×10^{-3}, 6×10^{-3}, 4×10^{-4}, and 1×10^{-3} respectively. The RME Incremental Cancer Risks for the Visiting Recreational Angler are 5×10^{-3}, 6×10^{-3}, 4×10^{-4}, and 1×10^{-3} for Allendale Pond, Lyman Mill Pond, Manton Reach, and Dyerville Reach, respectively. This suggests slightly higher Incremental Cancer Risk in Allendale Pond and Lyman Mill Pond compared to Manton Reach and Dyerville Reach.

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The CT Incremental Cancer Risks for the Resident Living Along the River for Allendale Pond, Lyman Mill Pond, Manton Reach, and Dyerville Reach are 1×10^{-3}, 1×10^{-3}, 2×10^{-4}, and 1×10^{-4}, respectively. The CT Incremental Cancer Risks for the Visiting Recreational Angler are 9×10^{-4}, 1×10^{-3}, 2×10^{-4}, and 1×10^{-4} for Allendale Pond, Lyman Mill Pond, Manton Reach, and Dyerville Reach, respectively. This suggests slightly higher Incremental Cancer Risk in Allendale Pond and Lyman Mill Pond compared to Manton Reach and Dyerville Reach.

For the Resident Living Along the River and the Visiting Recreational Angler, the largest contributors to Incremental Cancer Risk are fish consumption and contact with sediment. The cancer risk associated with fish consumption is the largest contributor to Incremental Cancer Risk for RME and CT Resident Living Along the River and the Visiting Recreational Angler for Allendale Pond, Lyman Mill Pond, Manton Pond, and Dyerville Pond.

The consumption of fish (2,3,7,8-TCDD) is by far the largest contributors to the total Incremental Cancer Risk for both receptors at all four river exposure points. Among the RME and CT scenarios and the four river exposure points, only the RME sediment exposure scenario at Allendale Pond and Lyman Mill Pond for the Resident Living Along the River has a cancer risk greater than one in ten thousand. The remaining sediment and bank soil exposure scenarios have cancer risk equal to or below one in ten thousand.

As summarized in Table 13, the calculated cancer risk for surface water is less than 1×10^{-4} at all four exposure points and at the reference/background areas for the Resident Living Along the River and the Visiting Recreational Angler for the RME and CT exposure scenarios. It should be noted that for Manton Pond and Dyerville Pond, the calculated risks are based on water quality data collected in Lyman Mill Pond. In the absence of analytical data for those ponds, the Lyman Mill surface water quality data were conservatively considered to be representative for those ponds. The available data are from unfiltered surface water samples and may contain sediment particulate matter. The calculated cancer risks associated with dermal exposures to surface water do not include dioxins and furans. Due to uncertainties with those exposures and risks, they have been evaluated in Appendix I.

As summarized in Table 13, cancer risks associated with sediment exposure are greater than 1×10^{-4} for the Resident Living Along the River at Allendale Pond and Lyman Mill Pond for the RME scenarios. Cancer risks associated with sediment exposure are less than or equal to 1×10^{-4} for the Resident Living Along the River and the Visiting Recreational Angler at all exposure points and reference/background areas for the CT scenarios and for the Resident Living Along the River at Manton Pond and Dyerville Pond and the reference/background areas for the RME scenarios, and the Visiting Recreational Angler at all four exposure points and reference/background areas for the RME scenario. The cancer risks for RME and CT scenarios for bank soil (Greystone Mill Pond and Allendale Pond) as summarized in Figures 51 through 54 and surface soil at the Fogarty Center (adult worker only) as shown in Tables 7.37.RME and 7.37.CT are below one in ten thousand for both receptors and all age groups.

Table 12 and Table 13 present the summary of RME and CT non-cancer and cancer risks respectively. These tables present the risks calculated for each receptor at each of the exposure points, the upstream background area, and the reference area, and also the Incremental risks

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associated with each receptor at Allendale Pond, Lyman Mill Pond, Manton Pond, and Dyerville Pond. As shown in Table 12, for both RME and CT scenarios for both the Resident Living Along the River and the Visiting Recreational Angler, the fish consumption exposure pathway is by far the predominant contributor to non-cancer risk among the exposure pathways at EA1 through EA4. As shown in Table 13, fish consumption is the predominant contributor, among the exposure pathways, to the RME and CT cancer risk for the Resident Living Along the River at EA1 through EA4. For the Visiting Recreational Angler, fish consumption is the predominant contributor to the RME and CT cancer risk at EA1 through EA4. Overall, Aroclor-1254 contributes the greatest portion of the non-cancer risk for the Resident Living Along the River and the Visiting recreational Angler at the exposure points adjacent to and downstream of the Site source area. Dioxin equivalents represent the largest chemical contributor to RME and CT cancer risk for the Resident Living Along the River and the Visiting Recreational Angler at the exposure points adjacent to and downstream of the Site source area.

Spatial Distribution of Fish Consumption Risks

Fish consumption risks have been assessed for the upstream background area (Greystone Mill Pond), the reference area (Assapumpset Pond and Brook), and the four river exposure points adjacent to and downstream of the source area (Allendale Pond, Lyman Mill Pond, Manton Pond, and Dyerville Pond). Therefore, it is possible to review the risk assessment results for fish consumption to obtain a sense of how fish consumption risks compare among the background area, reference area, and the four river exposure points. Table 11.RME summarizes, and Figures 53 and 54 present graphically the RME Hazard Index values and cancer risks for consumption of the combined fish diet by the Visiting Recreational Angler at all of the exposure points.

As shown in Figure 53, the RME Hazard Index values for the adult Visiting Recreational Angler for consumption of fish from Assapumpset Pond and Greystone Mill Pond are 1 and 3 respectively. The RME Hazard Index values (driven almost exclusively (greater than 99%) by Aroclor-1254) for the adult Visiting Recreational Angler for fish consumption in Allendale Pond, Lyman Mill Pond, Manton Pond and Dyerville Pond are 19, 21, 3, and 5, respectively. The Hazard Index values are highest at Allendale Pond and Lyman Mill Pond adjacent to and immediately downstream of the source area. The Hazard Indices are four to seven times lower further downstream in Manton Pond and Dyerville Pond. The Hazard Index values for fish consumption in Allendale and Lyman Mill Pond are approximately six to seven times higher than the corresponding values in the background area, Greystone Mill Pond.

As shown in Figure 54, the RME cancer risk (predominantly driven by 2,3,7,8-TCDD) associated with fish consumption for the adult Visiting Recreational Angler for Assapumpset Pond and Greystone Mill Pond are 2×10^{-5} and 3×10^{-5}. As shown in Figure 54, the RME cancer risk (predominantly driven by 2,3,7,8-TCDD) associated with consumption of the combined fish diet for Allendale Pond and Lyman Mill Pond for the adult Visiting Recreational Angler data are 2×10^{-3} and 2×10^{-3} respectively. The corresponding cancer risk for Manton Pond is 2×10^{-4}. The corresponding cancer risk for Dyerville Pond is 4×10^{-4}. The cancer risks for fish consumption are highest in Allendale Pond and Lyman Mill Pond (adjacent to and immediately downstream of the source area) and are approximately 10-fold lower further downstream at Manton Pond and Dyerville Pond. The fish consumption cancer risks are approximately 100 times higher in

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Allendale Pond and Lyman Mill Pond than the calculated risks for the background area, Greystone Mill Pond.

In conclusion, for the Resident Living Along the River and the Visiting Recreational Angler, the incremental cancer and non-cancer risks (the difference between the risks at a given exposure point and the risk at Greystone Mill Pond) for the four river exposure points are presented in Figure 59 and Figure 60 respectively. The incremental cancer and non-cancer risks are greatest in Allendale Pond and Lyman Mill Pond for both RME and CT scenarios. All four river exposure points have incremental RME cancer risks that exceed one in ten thousand (the upper end of the Superfund cancer risk range) and incremental non-cancer Hazard Index values equal to or greater than one.

Because 2,3,7,8-TCDD is the primary contributor to cancer risk via fish consumption and because Aroclor-1254 is the primary contributor to non-cancer Hazard Index, a review of the spatial distribution of concentrations of those substances in largemouth bass tissue (whole body and fillet) at the Greystone Mill Pond reference area, Lyman Mill Pond, and at Manton Pond are indicators of the spatial distribution of cancer and non-cancer risks associated with fish consumption. As can be seen in the two figures below, the concentrations of the primary risk contributors are lowest at the reference area, are highest at Lyman Mill Pond, and are lower at Manton Pond than in Lyman Mill Pond but are still higher than those from the reference area. The figures also show clearly that concentrations of the primary risk contributors are higher in whole body samples than in fillet samples of largemouth bass. The distribution of 2,3,7,8-TCDD and Aroclor-1254 in fish tissue among the river exposure areas and the background area Greystone Mill Pond is generally consistent with the distribution of those analytical parameters in sediment among the river exposure areas and the background area Greystone Mill Pond. This indicates that the incremental cancer and non-cancer risks are associated with analytical parameters that appear to be site related.

Figure 61 summarizes the contribution of 2,3,7,8-TCDD congener to the total dioxin TEQ for American eel and largemouth bass at the reference area, background area, and at Allendale Pond, Lyman Mill Pond, Manton Pond, and Dyerville Pond. As shown in Figure 61, it is clear that the 2,3,7,8-TCDD congener is generally not detected in American eel and largemouth bass tissue from Assapumpset Pond and Brook (reference area) and from Greystone Mill Pond (background area). In contrast, the 2,3,7,8-TCDD congener is detected in all American eel and largemouth bass tissue samples collected from Allendale Pond, Lyman Mill Pond, Manton Pond, and Dyerville Pond. With a few exceptions, the 2,3,7,8-TCDD congener represents less than 20% of the total dioxin TEQ for American eel and largemouth bass at the reference area and the background area. Again, in contrast, the 2,3,7,8-TCDD congener represents greater than 95% of the total dioxin TEQ for American eel and largemouth bass at the exposure areas adjacent to and downstream of the source area (Allendale Pond, Lyman Mill Pond, Manton Pond, and Dyerville Pond). This is a clear indication that the calculated Incremental (above background) cancer risk associated with fish consumption is associated with higher frequency of detection, higher concentrations, and higher contribution to the total dioxin TEQ by the 2,3,7,8-TCDD at the exposure areas adjacent to and downstream of the source area relative to the reference area and background area.

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Figure 62 presents a graphical summary of the concentrations of the 2,3,7,8-TCDD congener in sediment and the contribution of the 2,3,7,8-TCDD congener to the total dioxin TEQ in sediment for the reference area, background area, and the four exposure areas adjacent to and downstream of the source area. Consistent with the observations concerning fish tissue, the concentrations of the 2,3,7,8-TCDD concentrations in sediment are substantially higher in the four exposure areas adjacent to and downstream of the source area than at the reference area and background area. The mean 2,3,7,8-TCDD congener sediment concentrations at Allendale Pond (4126 ppt) and Lyman Mill Pond (1542 ppt) are approximately 105 times and 40 times higher than the mean concentration in sediment (39 ppt) at Greystone Mill Pond. The mean 2,3,7,8-TCDD sediment concentrations at Manton Pond (384 ppt) and Dyerville Pond (110 ppt) are also higher than the mean at Greystone Mill Pond. The concentrations of 2,3,7,8-TCDD in sediment show a clear trend from higher to lower concentrations starting at Allendale Pond (this exposure area contains the source area) and moving downstream to Dyerville Pond.

Also, as shown in Figure 62, the contribution of the 2,3,7,8-TCDD congener to the total dioxin TEQ of the sediments is substantially higher at Allendale Pond, Lyman Mill Pond, and Manton Pond (ranging between approximately 89% and 96%) than at either the reference area Assapumpset Pond and Brook (approximately 41%) or the background area Greystone Mill Pond (approximately 27%). The literature reports that dioxin concentrations in fish are directly related to corresponding sediment concentrations. Given that sediment 2,3,7,8-TCDD concentrations at the areas adjacent to and downstream from the source area are substantially higher than at the reference and background areas, and that the 2,3,7,8-TCDD congener's contribution to the total dioxin TEQ in sediments is also higher at the areas adjacent to and downstream from the source area than at the reference area and background area concentrations of 2,3,7,8-TCDD, it is clear that the 2,3,7,8-TCDD congener concentrations in sediments adjacent to and downstream of the source area are contributing to the Incremental (above background) cancer risk associated with fish consumption and also sediment contact. In other words, the available information indicates that the 2,3,7,8-TCDD in fish tissue and sediment is Site related and almost all of the Incremental cancer risk is associated with the 2,3,7,8-TCDD congener in sediment and fish tissue.

A similar analysis and graphical presentation has been prepared with respect to the major contributor (Aroclor-1254) to Incremental (above background) non-cancer Hazard Index values. Aroclor-1254 is associated with greater than 90% of the non-cancer Hazard Index for the Resident Living Along the River and the Visiting Recreational Angler. As shown in Figure 63, Aroclor-1254 was not detected in any largemouth bass and only two American eels from the reference area Assapumpset Pond and Brook, and Aroclor-1254 was detected in almost all of the American eel and largemouth bass samples collected from the background area Greystone Mill Pond and from Allendale Pond, Lyman Mill Pond, Manton Pond, and Dyerville Pond. For American eel, the portion of Total Aroclors contributed by Aroclor-1254 at the upstream background area Greystone Mill Pond is approximately 67% on average. For the exposure areas adjacent to and downstream of the source area, the Aroclor-1254 concentrations in American eel contribute virtually all (greater than 98%) of the Total Aroclor concentration. For largemouth bass, the Aroclor-1254 contribution to Total Aroclors is approximately 56% for the background area Greystone Mill Pond, approximately 95% for Lyman Mill Pond, and approximately 79% for Manton Pond. The data indicate that Aroclor-1254 contributes a greater portion of the Total Aroclors at the exposure areas adjacent to and downstream of the source area than at the

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background area. The data also suggest there are one or more sources of Aroclor-1254 that has resulted in higher fish tissue concentrations at Greystone Mill Pond than at the reference area Assapumpset Pond and Brook. In other words, the upstream background location is affected by some input of Aroclor-1254 to the Woonasquatucket River.

As shown in Figure 64, the concentrations of Total Aroclor and of Aroclor-1254 in sediments from Allendale Pond, Lyman Mill Pond, and Dyerville Pond are higher than the corresponding sediments from the reference area Assapumpset Pond and Brook and from the upstream background area Greystone Mill Pond. The mean Aroclor-1254 sediment concentration in Allendale Pond (1.51 mg/kg) and Lyman Mill Pond (0.27 mg/kg) are approximately 10 and 1.8 times higher than the corresponding mean sediment concentration in Greystone Mill Pond (0.15 mg/kg).

Aroclor-1254 is the primary contributor to Incremental (above background) non-cancer risk, but there also seems to be a source of Aroclor-1254 to Greystone Mill Pond that is responsible for the levels of Aroclor-1254 that are present at that upstream background location. The specific source of Aroclor-1254 to Greystone Mill Pond is unknown.

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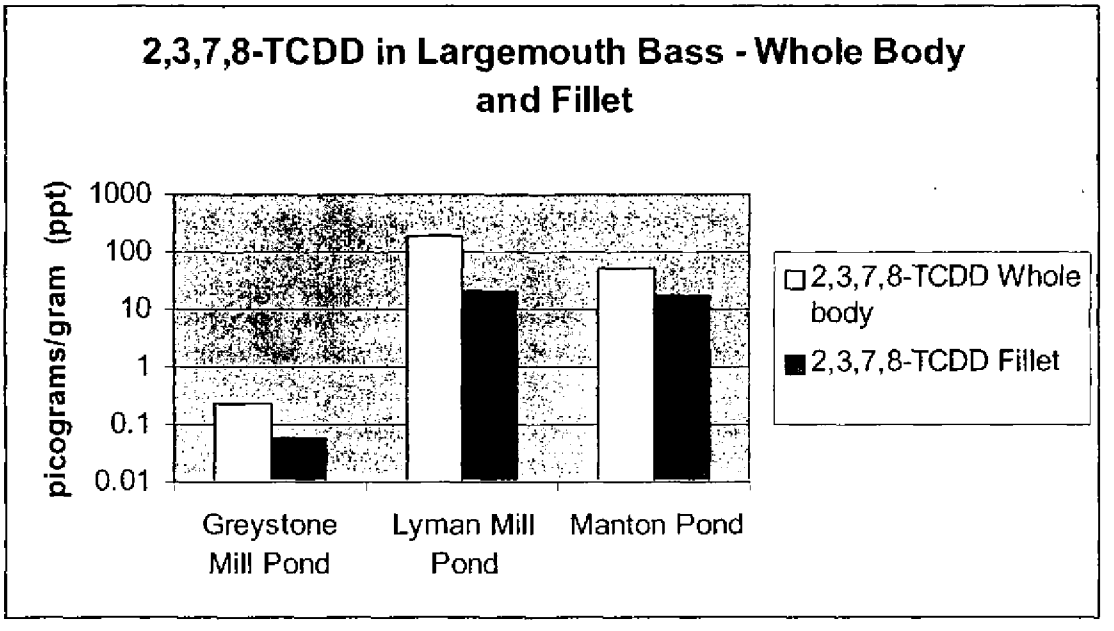

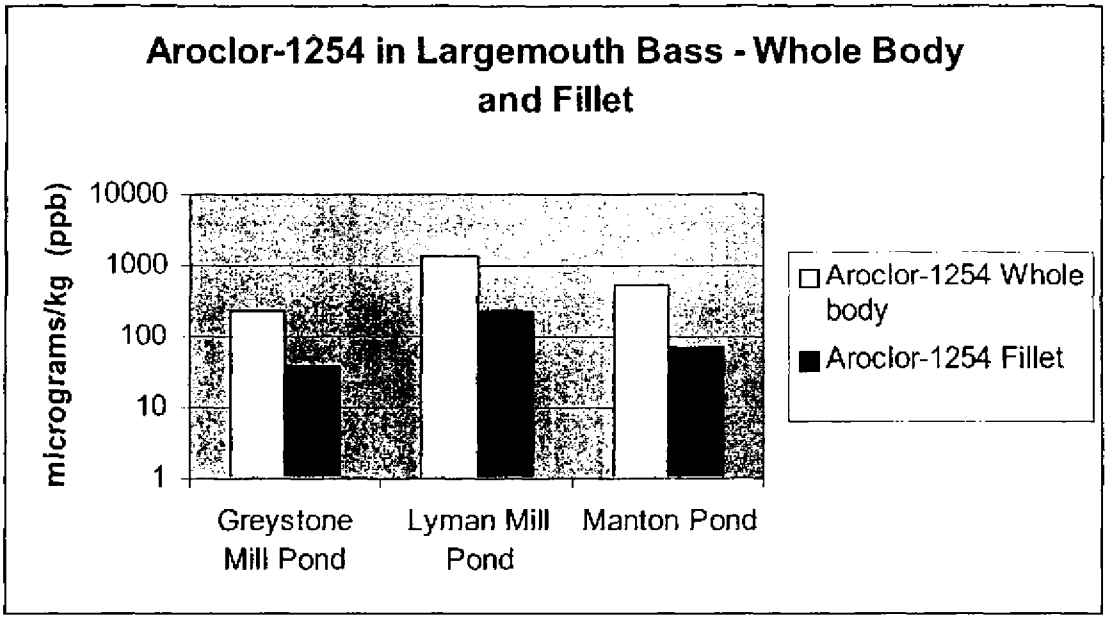

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6.0 UNCERTAINTY ANALYSIS

This section includes a discussion of major limitations of the analyses, any sources of uncertainties, and, if possible, any indication as to whether these uncertainties and limitations may have resulted in and over- or under-estimation of risk. The uncertainty section may also include unusual site conditions or extenuating circumstances that may be pertinent to risk management decisions. Other factors such as the inadequacy of toxicity factors to describe all possible COPC-receptor interactions and individual differences within the human population are included in this section. Uncertainties in the quantification of risk associated with the site are identified and their impacts on risk estimates are discussed below.

Hazard Identification

The extensive environmental investigations have provided a substantial body of information that has been utilized in the BHHRA. The identification of COPCs has been conducted consistent with USEPA guidance and has been done in a health protective manner. Based on currently available information, it is unlikely that any detected substances that have not been selected as COPCs would have a substantial impact on the BHHRA results and conclusions if they had been retained in the BHHRA.

Background conditions have not been specifically considered in the selection or elimination of substances as COPCs. Several of the persistent organic COPCs (such as dioxins, furans, and PCBs), while they are not naturally-occurring substances, are detectable at some concentration almost ubiquitously in environmental samples such as biota and sediments. Therefore, exposure concentrations of those COPCs represent "total" exposure potential from both site-related and non-site-related sources.

As shown in Table 1 of the Data Evaluation Report (MACTEC, 2003), collection of 10 fillet and 10 offal samples of largemouth bass in each of the following areas had been planned: Allendale Pond; Lyman Mill Pond; Greystone Mill Pond, and Assapumpset Brook/Pond. Three fillet and 3 offal samples of largemouth bass had been proposed for Manton Pond and for Dyerville Pond. However, because the Allendale Dam had again been breached, and the water levels in Allendale had receded, no largemouth bass samples could be collected from Allendale Pond in summer 2001. At Assapumpset Brook/Pond, 4 rather than 10 fillet and offal samples of largemouth bass were collected due to low numbers of largemouth bass obtained. No largemouth bass samples were obtained from Dyerville Pond. Three American eel whole body samples were collected from Dyerville Pond as a contingency.

There are some fish sample results that have not been incorporated into the biota consumption risk assessment. These samples were collected in April 2001 prior to the summer 2001 biota data collection program. Although the white sucker samples are limited in number and no other fallfish were collected as a point of comparison, these samples might provide a snapshot of conditions at a time just prior to the 2001 complete breaching of the Allendale Dam. The April 2001 fish samples included white sucker and fallfish collected from Allendale Pond and Lyman Mill Pond. The following information concerning these April 2001 fish samples is taken from

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Task 22F ELS Fish/Egg Tissue Chemistry Data Report, Post-Third Party Validation (Battelle, Feb. 7, 2002).

- Although five fallfish were collected from Allendale Pond, per USEPA direction, samples AP-SC-01 and AP-SC-02 were composited, and that composite sample was composited with AP-SC-03 to form the composite sample YU40COMP + W2030. Samples AP-SC-04 and AP-SC-05 were composited per USEPA direction to form composite sample YU41COMP. One whole body white sucker sample was collected from Allendale Pond (AP-CC-06).

- One female white sucker was collected from Lyman Mill Pond. The eggs were removed for analysis and the remaining whole body (minus the eggs) was submitted for analysis (LP-CC-07, LP-CC-07 EGG). One additional whole body male white sucker sample was collected from Lyman Mill (LP-CC-08).

The April 2001 fish samples were analyzed for a limited analytical suite including dioxins and furans and PCB congeners only. Aroclor 1254 (the non-cancer risk driver for fish consumption in this assessment) was not included in the analytical suite.

The single April 2001 white sucker whole body sample from Allendale Pond is much smaller (only 90 grams or roughly 3 ounces) and most likely younger than the other ten white sucker samples collected in summer 2001 from Allendale Pond (weights ranged from 900 grams to 1450 grams or roughly 2 pounds to 3.2 pounds). Thus, the April 2001 Allendale white sucker is smaller (only one tenth the size of all other samples) and younger than any of the existing samples from Allendale. A fish that small is not likely to be brought home and consumed by an angler. Therefore, that sample was not included in the Allendale Pond biota consumption risk assessment.

One white sucker sample from Lyman Mill Pond from April 2001 (LP-CC-08) has a substantially higher concentration of PCB-congener dioxin equivalents (0.976 ug/kg) than any of the other white sucker samples collected from Lyman Mill Pond (mean of 3 post-breach samples is 0.0401 ug/kg and the maximum is 0.0503 ug/kg). The other April 2001 Lyman Mill Pond white sucker sample (LP-CC-07) has 0.137 ug/kg of the PCB-congener dioxin equivalents. Neither of the April 2001 sample concentrations are more than two orders of magnitude higher than the post-breach concentrations, but both are higher than the maximum concentration. However, if these two samples had been added to the risk assessment data set for Lyman Mill Pond, the exposure point concentration for the RME PCB TEQ would have changed from .0503 ug/kg to 0.976 ug/kg. If this change had been made, the exposure point concentration for PCB TEQ would be higher than the exposure point concentration for dioxins and furans TEQ and the estimated RME cancer and non-cancer risks from co-planar PCBs would increase by a factor of approximately 19. The dioxins/furans TEQ for these two April 2001 white sucker samples (0.423 ug/kg and 0.457 ug/kg are reasonably consistent (perhaps somewhat lower) with the post-2001 breach samples from Lyman Mill Pond (mean of 0.580 ug/kg and maximum of 1.37 ug/kg). The April 2001 Lyman Mill white sucker samples appear to have higher PCB TEQs (number of samples is very small) and lower dioxins/furans TEQs than do the post-2001-breach white sucker samples.

The April 2001 samples were not analyzed for Aroclor-1254, the non-cancer risk driver. Incorporation of the April 2001 Lyman Mill white sucker samples would have substantially increased the calculated cancer risks for coplanar PCBs (PCB TEQ) but would not change the non-cancer hazard index values because the risk driver for non-cancer risk (Aroclor-1254) was not analyzed for in those samples. If it had been included in the BHHRA, a single sample (LP-CC-08) that may be an outlier, it would have been responsible for the substantial increase in the cancer risk estimates for consumption of white sucker in Lyman Mill Pond.

The representative nature of the fish tissue data for the four exposure points may not be equivalent. For Allendale Pond, data for twenty whole body fish tissue samples were used in the BHHRA. For Lyman Mill Pond, data for twenty whole body and ten fillet fish tissue samples were utilized. However, for Manton Pond, data for only three fillet fish tissue samples, and in Dyerville, data for only three whole body fish tissue samples were evaluated in the BHHRA. Therefore, based on the number of samples available and the number of species represented, it appears that the results of the BHHRA for fish consumption in Allendale Pond and Lyman Mill Pond are based on more representative data (more certain) than are the results for the fish consumption evaluation for Manton and Dyerville Pond. There are considerably fewer sediment samples in Manton Pond and Dyerville Pond than in Allendale Pond and Lyman Mill Pond. The sediment data for Allendale Pond and Lyman Mill Pond appear to be more representative of direct and indirect (bioaccumulation) exposure potential than the sediment data for Manton Pond and Dyerville Pond.

Exposure Assessment

The selection of receptors is conservative and health-protective for the conditions identified at the Site. The identification of the visiting recreational angler and the resident living along the river who is also involved in recreational angling and consumption as receptors to be evaluated is conservatively realistic for the current and likely future conditions at the river. There is considerable uncertainty concerning the presence of subsistence anglers/consumers at this portion of the river at this time. Although advisories have been issued in 1999 (Rhode Island Department of Health fish consumption advisories concerning dioxins and mercury) and 2003 (Rhode Island Department of Health and USEPA consumption advisory for fish, other animals, plants, and river water and an advisory against wading and swimming in the river water), those advisories may not be completely effective in preventing human exposure at the river. The fish consumption advisories and posted signs likely reduce the likelihood that subsistence anglers/consumers would choose this portion of this river as a source of food. The subsistence angler/consumer scenario may represent a conservative exposure scenario for future conditions at such time that the river is restored to a fishable, swimmable condition. The subsistence angler risk characterization has been presented as supplemental information in Appendix F. As expected, the risks for that exposure scenario are higher than for the Visiting Recreational Angler/consumer and for the Resident Living Along the River that is involved in recreational angling and fish consumption.

Due to the uncertainty associated with current or future subsistence angling in the lower Woonasquatucket River, the evaluation of human health risk associated with potential subsistence angler exposure scenario has been evaluated in Appendix F. While the USEPA objective is the return of the river to a fishable condition, there is no clear evidence that subsistence angling in the lower Woonasquatucket River has taken place or that it is likely to take place in the future.

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Subsistence angling would likely be more productive and efficient in the marine environment in the Providence area where saltwater species such as flounder and pollack are abundant. To provide additional context to the BHHRA, the subsistence angler scenario, including people who would rely on the Woonasquatucket River fish as a major component of the diet, have been evaluated, using fish consumption rates that have been obtained primarily from studies of subsistence angling in Connecticut. The studies are based on harvesting and consumption of saltwater species in Connecticut waters. No studies of fish consumption rates for freshwater subsistence angling in the New England area were available as a basis for the subsistence angler scenario.

The subsistence angler fish consumption rates have been selected from the 1999 report *Quantification of Fish and Seafood Consumption Rates for Connecticut* (Balcolm et al., 1999). The RME adult fish consumption rate of 148 gram/day is the calculated 95th percentile value for total fish consumption for individuals of Southeast Asian Families based on information provided in Table 12 of the Balcolm study. The CT adult fish consumption rate of 59.2 gram/day is the calculated mean value for total fish consumption for individuals of Southeast Asian Families based on information provided in Table 12 of the Balcolm study. Corresponding older child and child subsistence fish consumption rates were estimated by applying the ratios of child:adult (0.36) and older child:adult (0.66) that were identified in the derivation of high-end recreational angler fish consumption rates as discussed below. The RME subsistence angling fish consumption rates for the older child and child are 98.6 gram/day and 53.5 gram/day respectively. The CT subsistence angling fish consumption rates for the older child and child are 29.7 gram/day and 21.1 gram/day respectively. These relative subsistence angler fish consumption rates among the age groups are consistent with the estimated fish consumption rates for child and older child recreational anglers using 1/3 and 2/3 of the adult rates.

The RME and CT adult subsistence fish consumption rates (148 gram/day and 59.2 gram/day) are reasonably consistent with the 95th percentile and mean daily fish consumption rates for Native American subsistence populations of 170 gram/day and 70 gram/day) that were identified on page 10-26 of the Exposure Factors Handbook (USEPA, 1997b).

The assumed combined fish diet consisting of equal portions of the fish species collected at each exposure point and the fish consumption rates that have been assumed for the combined fish diet for the Resident living Along the River and the Visiting Recreational Angler are associated with some uncertainties. Given that there is an advisory in place that recommends that people not consume fish caught from the Woonasquatucket River, it is not currently possible to conduct a survey that would determine what species would comprise a typical diet for the identified receptors or to determine fish consumption rates for the identified receptors. Many potential sources of information were reviewed in an attempt to determine likely composition of a fish diet and to estimate fish consumption rates for the identified receptor groups at the Woonasquatucket River in Rhode Island. Regulatory agencies, public interest groups, the open literature, and USEPA publications were reviewed in order to estimate the composition of the fish diet and to estimate reasonable fish consumption rates.

No freshwater Rhode Island fish consumption studies were identified during the information search. One study, however, was identified as being particularly relevant to the fish consumption

scenario for the Woonasquatucket River. The study was reported in *Estimating Consumption of Freshwater Fish among Maine Anglers* (Ebert et al., 1993) published in the North American Journal of Fisheries Management. The Ebert et al. study was conducted to provide information concerning fish consumption rates for freshwater anglers and ultimately to provide some context for regulatory rule-making concerning 2,3,7,8-TCDD in Maine's rivers which is in the same New England region. The Ebert et al. study was conducted for Maine waters, which is in the same region of the country as the Woonasquatucket River. The study also focused on recreational angling as opposed to commercial fishing. This is important, because there is no commercial fishing in the Woonasquatucket River. All but one (American eel) of the fish species collected and evaluated for the Woonasquatucket River were also included in the list of fish species harvested by freshwater anglers in Maine. Largemouth bass, brown bullhead, and white sucker were included in the list of harvested species in the Ebert et al. paper. The survey information from Ebert et al. was also coded and separated by surface water type, clearly distinguishing fish consumption from flowing waters from ponds and lakes. The estimated fish consumption rates for "rivers and streams" have been selected for use in this BHHRA.

The Ebert et al. study compiled data from licensed freshwater anglers concerning the numbers and average length of each species group, caught by a household member, that were consumed within a household. From the numbers of fish and average length, the weight of fish consumed from each species was estimated for each household. Given the average household fish consumption rates (all species combined), it is possible to estimate per-person fish consumption rates based upon certain assumptions. For example, it can be assumed that one adult consumes all of the freshwater fish consumed within the household, or that two adults share equally all of the freshwater fish consumed within the household, or that an entire family of four (two adults and two children) share equally all of the freshwater fish consumed within the household. As a conservative measure, the fish consumption estimates for one adult consuming all of the freshwater fish consumed within the household have been selected for use for adult consumers of freshwater fish. In this BHHRA, consumption rates for the older child and child age groups for the Resident Living Along the River and the Visiting Recreational Angler have been estimated as 2/3 and 1/3 of the adult fish consumption rates. This approach is consistent with the assumption that fish consumption rates are directly related to the bodyweight of the age groups and the resultant fish consumption estimates are consistent with the information presented in the Exposure Factors Handbook (USEPA, 1997b).

The fish consumption scenarios for the Resident Living Along the River and the Visiting Recreational Angler evaluated in the BHHRA are considered to be reasonable and conservative evaluations of fish consumption. A more conservative, "high-end" estimate of fish consumption for these receptors has also been conducted for these receptors, as presented in Appendix G. In this evaluation, rather than evaluating a combined fish diet, each species has been evaluated as a separate exposure medium. Using this approach with the assumption that a person may eat the same species for a lifetime, the risk estimates for the individual species provide some information concerning the range of high-end risks associated with variability in the fish consumption diet. It is not known what the typical composition of the recreational angler diet might be. By evaluating each of the species of fish, the risks for the combined diet are bracketed (equal to or lower than the risks for the species with the highest risk and equal to or higher than the species with the lowest risk).

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The high-end fish consumption rates for the Resident Living Along the River and the Visiting Recreational Angler were selected based on information provided in USEPA's Exposure Factors Handbook, Volume II (1997). For the RME adult scenario, a fish consumption rate of 38.7 gram/day, which is the 95th percentile for recreational fish consumption as shown in Table 10-63 of the Exposure Factors Handbook, was selected. For the CT adult scenario, the mean recreational fish intake (15.8 gram/day) for age groups 21 through 80+ years in Table 10-61of the Exposure Factors Handbook was selected. The Exposure Factors Handbook did not provide analogous fish consumption rates for children and older children. Therefore, fish consumption rates for these age groups were estimated from published total (from all sources) fish consumption rates for these age groups. It was assumed that the ratio of fish consumed from recreational angling to total fish consumed was constant among adults, older children, and adults. The adult ratio (0.89) was applied to total fish consumption rates for the older child and child (from the Exposure Factors Handbook) to obtain the RME fish consumption rates of 25.8 gram/day and 14.0 gram/day for the older child and child respectively. For the CT scenario, the fish consumption rates of 7.94 gram/day and 5.63 gram/day were identified for the older child and child respectively. The RME and CT high-end adult recreational fish consumption rates (38.7 gram/day and 15.8 gram/day respectively) are reasonably consistent with, but higher than, the USEPA default exposure parameters for freshwater anglers of 25 gram/day and 8 gram/day respectively as identified on page 10-26 of the Exposure Factors Handbook.

The risks associated with the high-end exposure scenario are based on literature-based information rather than local or regional information concerning fish consumption rates associated with recreational angling. The risk associated with the high-end exposure scenario likely overestimate exposures and risks for the typical Resident Living Along the River and the Visiting Recreational Angler.

The values for receptor-specific exposure parameters such as soil and water contact rates, soil and sediment ingestion rates, and fish consumption rates have been identified in a conservative manner. Values have been identified based on available guidance and professional judgment. In risk assessment, when values are assigned in lieu of actual measurements, there is some uncertainty in the values, and that uncertainty may have an impact on the results of the risk assessment. In that context, the exposure estimates and associated risk estimates in this assessment would likely be overestimated rather than underestimated. Some factors that were not specifically addressed in the calculations could result in lower risk estimates.

It has been suggested that food preparation and cooking may result in reductions of dioxin/furan concentrations from the raw fish to the ready-to-eat-fish, particularly if the preparation removes the skin and other portions of the fish that contain greater amounts of fat that tend to be reservoirs for dioxins and furans. However, it has also been reported that, in general, cooking (as opposed to food preparation) does not reduce dioxin concentrations in food (USEPA, 2000). Therefore, no reduction in dioxin concentrations associated with cooking of biota has been incorporated into the BHHRA. The potential impact of fat removal as a fish preparation technique has not been specifically included in the BHHRA. This may be a factor in reducing dioxin exposures and risk if fat-trimmed fillets are consumed, but if whole body fish are pan cooked or prepared in stews before consumption, the reduction in dioxin exposure would be minimal. The BHHRA did evaluate consumption of fish based on whole body concentrations for all species evaluated as well as on fillets (skin on) concentrations (largemouth bass). It is clear that fillet dioxin/furan

concentrations (and associated risks) are lower in the Largemouth Bass fillets that contain less lipid than in the largemouth bass whole body samples. It is likely that risks associated with consumption of skinned fillets would be lower than the risks identified for the skin-on fillets that were evaluated in the BHHRA.

The exposure points have been identified geographically. In other words, each of the reaches of the river has been identified as an exposure point. It has been assumed that a Resident Living Along the River or a Visiting recreational Angler would have a favorite fishing spot and that spot is where a particular receptor would fish most, if not all, of the time. This is a simplifying assumption that will enable the evaluation of risks for each of the reaches separately. It is possible, however, that anglers could fish at multiple locations in a random sequence, and that a given receptor could be exposed to sediment, surface water, and by consumption of fish at multiple exposure points within the Woonasquatucket River. If it had been assumed that a receptor could be exposed at multiple exposure points, then the calculated human health risks would have reflected some type of allocation of the receptor's overall exposure. Such an allocation might have been done assuming equal exposure at all potential exposure points or by weighting exposure frequency at each exposure point based on some other site-specific information. In any case, if each receptor had been assumed to be exposed at all of the exposure points, the calculated mean or weighted exposures and risks would likely have been lower than the risks at the exposure point with the highest risk (Lyman Mill Pond for the Visiting recreational Angler and both Allendale Pond and Lyman Mill Pond for the Resident Living Along the River) and higher than the exposure point with the lowest risk (Dyerville Pond).

In this BHHRA, all submerged sediments within the reaches evaluated have been included in the exposure assessment and risk characterization for potential human sediment contact. Those sediments closest to the river bank and under shallow water would typically represent those sediments that have a greater potential for human contact during wading and other recreational activities. Sediments that are located beneath several feet of water would typically have less potential for human contact. If only sediments that are located beneath shallow water had been selected for the exposure assessment, the exposures to dioxins/furans and associated risks would have been lower than reported in this BHHRA.

The sediment exposure point concentrations for dioxins/furans were the maximum reported concentration for each of the following exposure areas: Assapumpset Brook and Pond; Lyman Mill Pond; Manton Pond; and Dyerville Pond. Therefore, selection of sediment samples located beneath only shallow water from those exposure areas would result in exposure estimates and risk estimates for dioxins/furans that are equal to or lower than the values presented in the BHHRA. This would not change the conclusions of the BHHRA with respect to direct contact with sediments, since the receptor cancer risk estimates for direct contact with sediments is below 1×10^{-4} for all exposure areas evaluated for both RME and CT scenarios.

Dioxins/furans were the largest contributor to cancer risk for sediments at Allendale, Lyman Mill, Manton, and Dyerville Ponds. No sediment COPCs were associated with direct contact sediment Hazard Index values greater than one. At Allendale Pond and at Greystone Mill Pond, the 95% Upper Confidence Limit on the mean was utilized as the direct contact exposure point concentration. At Greystone Mill Pond, the 95% Upper Confidence Limit on the mean dioxin/furan TEQ concentration (0.00012 mg/kg) is approximately 50% lower than the maximum

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detected concentration in that exposure area (0.00018 mg/kg). At Allendale Pond, the 95% Upper Confidence Limit on the mean dioxin/furan TEQ concentration (0.0057 mg/kg) was utilized as the direct contact sediment exposure point concentration. The highest dioxin/furan TEQ sediment concentration in Allendale Pond was reported to be 0.073 mg/kg. As shown in Figure 41, the highest concentrations of dioxin/furans in sediment in Allendale Pond were from samples collected in the central, deeper water areas and near Allendale Dam (also deeper water). Based on that information, it appears that selection of only sediment samples from locations with shallow water (less than one or two feet) at Allendale Pond would have resulted in lower estimates of exposure and risks for direct contact with sediments for dioxins/furans. This would not have changed the conclusion that the receptor cancer risk estimates for direct contact with sediments is below one in ten thousand for all exposure areas evaluated for both RME and CT scenarios.

The estimation of dermal absorption rates of dioxins and furans in surface water is highly uncertain. There are published estimated permeability constants (K_p) for dioxins and human skin. However, the values presented in the USEPA Dermal Exposure Assessment Guidance are identified as being outside the Effective Model Domain (EMD) for the predictive model. Therefore, the use of those values would introduce considerable uncertainty into the exposure assessment. It appears the dioxin/furans that have been measured in surface water may be associated with suspended particulate matter, perhaps derived from sediments. This is likely because 2,3,7,8-TCDD has such a low solubility in water and it has a high affinity for sediments that contain organic carbon as well as for high lipid materials such as fish. This dioxin congener has a water solubility of 1.92×10^{-5} mg/liter. The dioxin/furan/HCX surface water data represent grab surface water samples collected in 1999 (TetraTech NUS, Inc., 2000b). The average suspended solids in the Allendale Pond and Lyman Mill Pond surface water samples were 5.4 mg/liter and 3.6 mg/liter. However, an evaluation of available suspended solids and turbidity data did not show a good correlation between dioxin surface water concentrations and either suspended solids or turbidity measurements.

More than 99% of the dioxin/furan/HCX TEQ in the surface water samples is associated with 2,3,7,8-TCDD. This congener was detected in seven of sixteen surface water samples from Allendale Pond and in six of ten samples collected from Lyman Mill Pond. The maximum detected concentrations of 2,3,7,8-TCDD were 1.27×10^{-5} mg/liter and 8.53×10^{-7} mg/liter in Allendale Pond and Lyman Mill Pond respectively. Average concentrations were 1.1×10^{-6} mg/liter and 1.25×10^{-7} mg/liter for Allendale Pond and Lyman Mill Pond respectively.

If the 2,3,7,8-TCDD in the surface water is associated with the suspended solids in the samples, then the compound would be expected to stay bound to the suspended solids, rather than to enter the dissolved phase in the surface water. The calculation of the dermal exposures employed the K_p with an underlying assumption that the chemical is in the dissolved phase in water. If the chemical is bound to suspended solids, then the K_p would substantially overestimate the diffusion of the chemical from the water into the skin. In addition, Exhibit B-2 of the USEPA guidance, *Risk Assessment Guidance for Superfund, Volume I: Human Health Evaluation Manual (Part E Supplemental Guidance for Dermal Risk Assessment) Interim* (USEPA, 2001) indicates that although a permeability constant K_p has been calculated for 2,3,7,8-TCDD, the chemical's characteristics place the chemical outside of the Effective Predictive Domain (EPD) of the model that is used to estimate the K_p. Therefore, the K_p value produced by the model and used in this

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BHHRA is highly uncertain. In fact, this model of diffusion-based absorption from water to the skin would likely estimate absorption rates when, in fact, no absorption might be taking place. In Appendix I, the range of cancer risks that might be associated with potential dermal exposures to dioxin in surface water were evaluated.

Factors suggesting that the exposures are calculated in a health-protective manner include:

- With the source areas controlled, it would be expected that over time, with influx of sediment from upstream during storm events and also typical conditions, that concentrations of COPCs in sediment and in fish would decrease. The BHHRA assumes that concentrations in sediment and fish tissue will remain constant for the entire duration of the exposure.
- There is variability within human populations with respect to behavior. The RME scenarios are intended to be representative of those people whose behavior and other characteristics result in higher than typical levels of exposure.
- The visiting subsistence angler scenario assumes that fish are consumed almost daily for the person's lifetime.

The surface water EPCs for Lyman Mill reach were used as surrogate to model surface water concentrations for Manton and Dyerville reaches. Because Lyman Mill Pond is upgradient of these two reaches, data collected from this pond are considered conservative estimators for water quality standards at these two reaches. This assumption may actually result in overestimation of dioxin concentrations in surface water in the Manton and Dyerville reaches, since the likely source of the dioxin in surface water would be suspended sediments and the dioxin concentrations in sediments of these two reaches are lower than the corresponding sediments in Lyman Mill Pond.

Toxicity Assessment

The toxicity assessment has been conducted consistently with available USEPA guidance. Dose-response information has been obtained from the IRIS database, NCEA, CAL-EPA, and USEPA's Health Effects Assessment Summary Tables. These sources of dose-response values are commonly used for regulatory risk assessment activities and are generally considered to be conservative in nature. The use of surrogate toxicity values for chemicals lacking US EPA recommended values is conservative since it is likely that the chemical specific toxicity would be lower than those exhibited by their surrogate.

HCX

There are no dose-response values listed in IRIS or by NCEA for the compound HCX (also known as 1,2,4,5,7,8-hexachloroxanthene). There is also a paucity of laboratory toxicity testing data for HCX, particularly with respect to chronic exposure in animals. A structurally similar compound, 2,3,6,7-terachoroxanthene, has been reported to be as effective in causing the blue sac syndrome in fish fry and sac fry mortality as is 2,3,7,8-TCDD (Zabel and Peterson, 1996). These authors report an acute toxicity toxic equivalency factor of 1.09 to 2.29 for the compound 2,3,6,7-tetrachloroxanthene.

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An in-vitro bioassay evaluating mouse epithelial cell proliferation (Viswanathan et al., 1987) suggests HCX is one million-fold less toxic than 2,3,7,8-TCDD. In addition, a multi-species study by DeCaprio et al. (1987), concluded that HCX was less toxic than 2,3,7,8-TCDD by factors ranging from 5-000 fold less toxic to one million-fold less toxic. In a draft unpublished report, Mark Hahn of Woods Hole reports that HCX is able to displace TCDD from trout and human AhRs. Although the study was not able to determine relative affinities (only two concentrations tested), the author suggests a very rough estimate of the relative affinity of HCX of approximately 5000-fold less than TCDD. In the absence of other toxicity data, a TEF of 0.0002 has been applied to HCX concentrations in order to assess cancer potential of that compound. Due to the uncertainty associated with the toxicity of this compound, the risk assessment for HCX is presented in Appendix H. In order to provide additional perspective concerning the uncertainty associated with the toxicity of HCX, Appendix H also discusses the potential impact on the risk assessment of the use of a TEF of 1 for HCX.

Lead

Lead exposures and associated risks associated with fish consumption have been evaluated qualitatively by comparison of the fish consumption scenarios to a quantitative evaluation that was conducted for the Columbia River Basin. It appears the qualitative assessment does not bias the risk assessment results either in the high or low direction.

Dioxin and Furan Congeners

Dioxin and furan congeners have been evaluated using the 2,3,7,8-TCDD oral CSF of 1.5×10^5 per mg/kg/day (taken from USEPA's HEAST, 1997) and the mammalian TEFs from Van den Berg et al. 1998. This approach represents the most recent risk assessment approach for evaluating dioxins and furans. This approach has been employed because there is not adequate toxicity testing for each of the hundreds of dioxin and furan congeners. Although the TEFs do have scientific basis, the use of the TEFs to estimate the cancer potency of each of the congeners does have some uncertainty associated with it. However, the predominant congener in environmental media at the four exposure points is 2,3,7,8-TCDD. Since 2,3,7,8-TCDD is considered to be the most toxic of the dioxin and furan congeners, the use of the TEFs has less impact on the risk assessment than would be the case where 2,3,7,8-TCDD was not the predominant congener and other congeners without published CSFs were the focus of the assessment.

The oral CSF for dioxin that was utilized in this assessment is taken from HEAST, 1997. The USEPA's *Exposure and Human Health Reassessment of 2,3,7,8-Tetrachlorodibenzo-p-Dioxin (TCDD) and Related Compounds, Draft* from September 2000 identifies another potential CSF of 1×10^6 per mg/kg/day. Using the alternative CSF, the cancer risk for fish ingestion would increase by a factor of approximately 7. As an example, and in order to provide additional perspective on the cancer risks calculated in this BHHRA, this alternative oral CSF has been employed to recalculate cancer risks for consumption of the combined fish diet from Lyman Mill Pond for the Adult Resident Living Along the River. In Appendix E, Tables E.1 and E.2 present and summarize those calculations for the RME scenario, and Tables E.3 and E.4 present and summarize the CT calculations. For the RME scenario, the calculated cancer risk for fish consumption using the alternative CSF is 1×10^{-2} versus 2×10^{-3} using the HEAST value. For the

CT scenario, the calculated cancer risk for fish consumption using the alternative CSF is 3×10^{-3} versus 4×10^{-4} using the HEAST value.

Since fish ingestion is the predominant exposure pathway for most exposure scenarios and because dioxins/furans are the predominant COPC, the overall cancer risk estimates for the Resident Living Along the River and the Visiting Recreational Angler would increase by a factor of approximately 7 if the alternative cancer slope factor were used. This would not change the overall conclusions of the BHHRA with respect to cancer risk. The risk associated with consumption of the combined fish diet and the total receptor RME and the CT Incremental (above background) cancer risk for the Resident Living Along the River and the Visiting Recreational Angler would remain above the upper end of the Superfund risk range. However, the RME bank soil cancer risk for the Resident Living Along the River (Allendale Pond exposure point) would increase from within the Superfund cancer risk range (2×10^{-5}) to the upper end of the range (revised risk would be 1×10^{-4}). In addition, the RME sediment (direct contact) cancer risk for the Resident Living Along the River at the Manton Pond and Dyerville Pond exposure points would increase from within the Superfund cancer risk range to above (approximately 2×10^{-4}) and equal to (approximately 1×10^{-4}) the upper end of the range for Manton Pond and Dyerville Pond respectively. The Fogarty Center RME and CT cancer risks would not change substantially (they would remain within the Superfund cancer risk range). The cancer risk for the adult employee at that location is driven by carcinogenic PAHs and arsenic in surface soil. Dioxin TEQs are not a predominant contributor to cancer risk for that receptor and this would not change with the use of the alternative CSF.

Non-cancer risk was not quantitatively evaluated for potential exposures to dioxins, furans, and HCX. There is not currently a published USEPA oral RfD available for 2,3,7,8-TCDD, any other dioxin or furan congener, or dioxin-like compounds such as HCX. USEPA has concluded that the current average dioxin exposure to the human population is greater than the RfDs that would be calculated based on available data. USEPA, therefore, concluded that RfD values would not be informative for safety assessment (USEPA, 2000). Non-cancer effects such as effects on reproduction and development, suppression of the immune system, and chloracne (USEPA, 2000) have been associated with these compounds in animal studies and it is likely that similar effects might occur with human exposure. Therefore, the non-cancer risk associated with potential exposure to dioxins, furans, and HCX are understated in this BHHRA.

Risk Characterization

Appendix F presents the human health risk estimates for a potential future Subsistence Angler scenario. If a Subsistence Angler scenario did occur, the expected fish consumption rates would be higher than those identified for the recreational angling exposure scenarios that have been evaluated in the main body of the BHHRA. For the Subsistence Angler scenario, fish consumption remains the predominant risk contributor, with 2,3,7,8-TCDD and Aroclor-1254 being the largest contributors to cancer risk and non-cancer risk respectively. Because the fish consumption rates are higher, the calculated risk are also higher for this scenario than are the risks calculated for the recreational angling scenarios.

Appendix G presents a high-end fish consumption exposure scenario for recreational angling. This appendix utilizes fish consumption rates drawn from literature sources which are not

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necessarily well suited to the assessment of recreational angling and associated fish consumption at the Woonasquatucket River in Rhode Island. The fish consumption rates utilized in this appendix are higher than the regional freshwater recreational fish consumption rates that were used to evaluate the recreational angling scenario in the main body of the BHHRA. For this high-end fish consumption scenario, 2,3,7,8-TCDD and Aroclor-1254 remain the largest contributors to cancer risk and non-cancer risk respectively. The calculated cancer and non-cancer risks for this scenario are higher than for the recreational angling and fish consumption scenario that is evaluated in the main body of the BHHRA.

Due to the uncertainty associated with the toxicity of HCX, the exposure assessment and risk characterization for that compound has been evaluated in Appendix H. In order to evaluate the potential contribution of HCX to human health risk, the risks associated with HCX for the Resident living Along the River at Lyman Mill Pond exposure point have been evaluated and compared to the cumulative receptor risks for all other COPCs for the Resident Living Along the River at the Lyman Mill Pond exposure point. This receptor and exposure point was associated with the highest cumulative cancer risk estimate (all COPCs except HCX) among the receptors and exposure points evaluated in the main body of the BHHRA. By comparing HCX risks to risks for all other COPCs, the impact of HCX toxicity uncertainty on the BHHRA results and conclusions can be evaluated.

The HCX exposure assessment in Appendix H utilized exposure parameters identical to those used in the main body of the BHHRA for all of the other COPCs. Since no non-cancer Reference Dose is available for HCX, no non-cancer Hazard Index has been calculated for HCX. Therefore, the non-cancer risk for all receptors may be somewhat underestimated for all receptors. HCX has been identified as a dioxin-like compound and it has been evaluated with respect to cancer risk. As discussed in Appendix H, a preliminary dioxin TEF of 0.0002 has been identified for HCX based on the available information. As shown in Table 11.RME for the Resident Living Along the River at the Lyman Mill Pond exposure point, the RME cancer risk for all COPCs other than HCX was $1x10^{-2}$ and the associated non-cancer Hazard Index ranged from 14 to 22 for the three age groups. The RME cancer risk for HCX, using the TEF of 0.0002, for the Resident Living Along the River at the Lyman Mill Pond exposure point, was $4x10^{-7}$. In other words, the calculated HCX RME cancer risk is 25,000 times lower than the calculated cancer risks for the other COPCs. In Appendix H, the HCX CT cancer risk ($4x10^{-8}$) for the same receptor and exposure point have been compared to the calculated cancer risk for all other COPCs ($1x10^{-3}$). Once again, the calculated HCX CT cancer risk is 25,000 times lower than the calculated CT cancer risk for all other COPCs.

To add additional context, a more conservative (health-protective) approach has also been applied in Appendix H. If it were assumed that HCX has the same cancer potency as the most toxic of the dioxin (furan congeners 2,3,7,8-TCDD), a dioxin TEF of 1 could be assigned to HCX. The resultant HCX cancer risks would be 5,000 times higher (1/0.0002) than the HCX cancer risks discussed above. For the RME scenario, the HCX cancer risk would be $2x10^{-3}$ (cancer risk for all other COPCs = $1x10^{-2}$) and for the CT scenario, the HCX cancer risk would be $2x10^{-4}$ (cancer risk for all other COPCs = $1x10^{-3}$). The HCX cancer risk would be approximately five times lower than the risk for all other COPCs. Therefore, even if HCX had been assumed to be as toxic as 2,3,7,8-TCDD, it would not be the predominant contributor to cancer risk. This indicates,

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given the available toxicity information for HCX, that the lack of a published CSF or dioxin TEF is unlikely to have a significant impact on the results and conclusions of the cancer risk compound of BHHRA.

The Incremental (above background) risks have been compared to Superfund risk management criteria and benchmarks in order to draw conclusions concerning the Site-related risks. An evaluation has been conducted to confirm that the largest chemical contributors to the Incremental risk are Site-related. As has been discussed previously, more than 99% of the cancer risk for the Resident Living Along the River and Visiting Recreational Angler at Allendale Pond, Lyman Mill Pond, Manton Pond, and Dyerville Pond is associated with the dioxin TEQ. The concentrations of dioxin TEQ in fish tissue, and sediment have been reviewed to further evaluate the chemical contributors to the incremental risks.

Although the dioxin TEQ is clearly the predominant contributor to cancer risk at the Site, it is also clear that the congener 2,3,7,8-TCDD is by far the major risk contributor to the dioxin TEQ. It is also clear that Aroclor-1254 is the specific aroclor that is the largest contributor to the non-cancer risks at the Site.

Table 14 provides a brief overview of the uncertainties associated with the BHHRA and the potential impacts of those uncertainties on the results and conclusions.

Overall, the risk characterization provides conservative estimates of non-cancer and cancer risks consistent with USEPA risk assessment guidance (USEPA, 1989).

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7.0 CONCLUSIONS

The potential risks associated with current and future exposure for residents living along the river, visiting recreational anglers, and visiting subsistence anglers have been characterized. The risk characterization included the evaluation of fish consumption, exposures to surface water and sediment at four exposure points (Allendale Pond, Lyman Mill Pond, Manton Pond, and Dyerville Pond) within the Woonasquatucket River, and exposure to bank soil within Allendale Pond and Greystone Mill Pond background area. The risks associated with potential surface soil exposures at the Fogarty Center were also evaluated. Risks have been calculated using both RME and CT exposure scenarios. The calculated risks have been compared to the Superfund cancer risk range of 10^{-6} to 10^{-4} and to a Hazard Index value of 1.

Human health risks have also been characterized for an upstream riverine background area (Greystone Mill Pond) and a reference area (Assapumpset Brook and Pond). The risks associated with potential exposures at Allendale Pond, Lyman Mill Pond, Manton Pond, and Dyerville Pond have been compared to the calculated risks at the background area and the reference area. In addition, the incremental risks above those identified for the background area have been identified for Allendale Pond, Lyman Mill Pond, Manton Pond, and Dyerville Pond.

For the RME scenarios, the following conclusions have been drawn for the four river exposure areas (Allendale Pond, Lyman Mill Pond, Manton Pond, and Dyerville Pond) and the Fogarty Center.

- The calculated cancer risk (all exposure media) for each age group of each receptor group at each of the four exposure points is greater than the Superfund risk range of 10^{-6} to 10^{-4} and the Hazard Index of one for each age group of each receptor group at each of the four exposure points is greater than one.
- Only the fish consumption pathway is associated with Hazard Index values greater than one. Hazard Index values for exposure to surface water, sediments, and bank soil are each less than one.
- Consumption of fish and ingestion and dermal contact with sediment are the pathways that are the largest contributors to cancer risk.
- Dioxins and furans (particularly 2,3,7,8-TCDD), Aroclor-1254, coplanar PCBs, and technical chlordane are the largest contributors to cancer risk for the fish consumption pathway.
- Aroclor-1254, technical chlordane, and Aroclor-1268, are the largest contributors to non-cancer risk for the fish ingestion pathway.
- Cancer and non-cancer risks are higher in Allendale and Lyman Mill Ponds than in Manton and Dyerville Ponds for all receptors. This is, however, affected by the types and numbers of fish collected at the exposure points.
- Cancer and non-cancer risks are higher in Allendale Pond, Lyman Mill Pond, Manton Pond (cancer risk only), and Dyerville Pond than in the reference area Assapumpset Brook and Pond and in the background Greystone Mill Pond area.
- For residents living along the river and visiting recreational anglers, at Allendale Pond, Lyman Mill Pond, Manton Pond, and Dyerville Pond, the incremental risks above those

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risks calculated for the background area are greater than the USEPA risk criteria (cancer risk range of 10^{-6} to 10^{-4}). For Allendale Pond, Lyman Mill Pond, and Dyerville Pond, the incremental non-cancer Hazard index is equal to or greater than one for residents living along the river and the visiting recreational angler. The incremental Hazard Index for these receptors is less than one for Manton Pond.

- For Allendale Pond, Lyman Mill Pond, Manton Pond, and Dyerville Pond, the incremental risks represent a large difference based on percent of the risks calculated for the reference area and background area. At Allendale Pond, Lyman Mill Pond, Manton Pond, and Dyerville Pond the incremental cancer risk is 50 times, 60 times, 4 times, and 10 times the calculated cancer risk at the background Greystone Mill Pond area. The incremental risk is greater in Lyman Mill Pond and Allendale Pond than in Manton and Dyerville Ponds. The incremental non-cancer risk expressed as the Hazard Index (driven by Aroclor-1254) at Allendale Pond and Lyman Mill Pond are greater than the calculated non-cancer risk at the Greystone Mill Pond area. Incremental non-cancer Hazard Index at Manton Pond is zero. The incremental non-cancer risk at Dyerville Pond is approximately equal to the background non-cancer risk. The incremental non-cancer risk is greater in Lyman Mill Pond and Allendale Pond than in Manton and Dyerville Ponds.
- The cancer and non-cancer risks for the RME scenario are within the Superfund cancer risk range and below a Hazard Index value of one for adult worker surface soil exposures at the Fogarty Center.

For CT scenarios, the following conclusions have been drawn for the four river exposure areas (Allendale Pond, Lyman Mill Pond, Manton Pond, and Dyerville Pond) and the Fogarty Center.

- The calculated cancer risk (summed for all exposure media) for each age group of each receptor group at Allendale Pond and Lyman Mill Pond is equal to or greater than the Superfund risk range and the Hazard Index of one for each age group of each receptor group. At Manton Pond and Dyerville Pond, the cumulative cancer risk for each of the three age groups for the visiting recreational angler and the resident living along the river is within the Superfund risk range (10^{-6} to 10^{-4}). However, the calculated cancer risk for all three age groups combined are greater than the high end of the Superfund cancer risk range.
- Consumption of fish is the pathway that is the largest contributor to cancer risk.
- Dioxins and furans (particularly 2,3,7,8-TCDD), Aroclor-1254, coplanar PCBs, and technical chlordane are the largest contributors to cancer risk for the fish consumption pathway.
- Aroclor-1254, technical chlordane, and Aroclor-1268, are the largest contributors to non-cancer risk for the fish ingestion pathway.
- Cancer and non-cancer risks are higher in Allendale Pond, Lyman Mill Pond, Manton Pond, and Dyerville Pond than in the reference area Assapumpset Brook and Pond and in the background Greystone Mill Pond area.
- For Allendale Pond, Lyman Mill Pond, Manton Pond, and Dyerville Pond, the incremental risks above those risks calculated for the background area are generally greater than the typically applied Superfund risk management criteria (cancer risk range of one in one million to one in ten thousand and a hazard index of greater than one). The CT incremental cancer risk for the resident living along the river and the visiting

recreational angler at Dyerville Pond are equal to the upper end of the Superfund cancer risk range. Another exception is that the CT incremental non-cancer risk (Hazard Index) for each age group for the resident living along the river and the visiting recreational angler at Manton Pond and Dyerville Pond are less than one.

- For Allendale Pond, Lyman Mill Pond, Manton Pond, and Dyerville Pond, the incremental risks above those risks calculated for the background area represent a large difference based on percent of the risks calculated for the background area. Incremental cancer risks are generally higher in Lyman Mill Pond and Allendale Pond than in Manton and Dyerville Ponds.
- The cancer and non-cancer risks for the CT scenario are within the Superfund cancer risk range and below a Hazard index value of one for the adult worker surface soil exposures at the Fogarty Center.

The concentrations in fish tissue of the predominant risk contributors are directly related to corresponding sediment concentrations. Although direct contact exposures to sediments are not associated with the largest risks at the Site, the sediments appear to be the "source" of the largest risks at the Site.

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8.0 CALCULATION OF PRELIMINARY REMEDIATION GOALS

The baseline risk assessment has identified the chemicals that most significantly contribute to human health risks, particularly for the fish consumption pathway, as shown in Tables 10.13.RME through 10.37.RME and Tables 10.13.CT through 10.37.CT. Those chemicals that are associated with an excess lifetime cancer risk greater than one-in-one-million and/or a non-cancer hazard quotient greater than 1 in any medium have been identified as COCs. Preliminary Remediation Goals (PRGs) will be established by a two-step process. In the first step, risk-based biota PRGs for the most significant COCs will be identified for various fish consumption risk levels (cancer risk of 10^{-6}, 10^{-5}, 10^{-4}, and hazard quotients of 0.1, 1, and 10). PRGs for one or more media will be developed for dioxin TEQ, Aroclor-1254, coplanar PCBs, technical chlordane, dieldrin, benzo(a)pyrene, and arsenic. In the second step, using the concept of steady-state conditions between lipophilic substances in sediment and fish tissue, sediment concentrations corresponding to the risk-based tissue concentrations that were identified. These sediment concentrations will be risk-based sediment concentrations for consideration in the remedial decision-making process.

Site-specific, chemical-specific, Biota-Sediment Accumulation Factors (BSAFs) will be utilized in the development of sediment PRGs. The BSAF "is the lipid-normalized concentration of the chemical in the organism with respect to the organic carbon-normalized concentration in the sediments" (Cook et al., 1996). The sediment PRGs for human health will be derived by dividing the risk-based biota PRGs by BSAFs. The sediment PRGs will be presented in a separate document.

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ACRONYMS

AhR	Ah Receptor
ARAR	Applicable or Relevant and Appropriate Requirements
BCRA	Biota Consumption Risk Assessment
BERA	Baseline Ecological Risk Assessment
BSAF	Biota-Sediment Accumulation Factor
COPCs	chemicals of potential concern
CRAVE	Cancer Risk Assessment Verification Endeavor
CSM	conceptual site model
CT	Central Tendency
DER	Data Evaluation Report
DQOs	data quality objectives
EAs	exposure areas
ELS	early life stage
EPC	exposure point concentrations
FSP	Field Sampling Plan
HCX	hexachloroxanthene
LOAEL	Lowest Observable Adverse Effect Level
mf	modifying factor
NOAEL	No Observable Adverse Effect Level
OHM	oil and/or hazardous materials
PAHs	polycyclic aromatic hydrocarbons
PCBs	polychlorinated biphenyls
ppt	parts per trillion
PRGs	preliminary remediation goals
QAPP	Quality Assurance Project Plan
RBP	Rapid Bioassessment Protocol
RBC	Risk Based Concentration
RfC	Reference Concentration
RfDs	Reference Doses
RfD_ss	Subchronic Reference Doses
RIDEM	Rhode Island Department of Environmental Management
RME	Reasonable Maximum Exposure

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SF	Slope Factor
SOW	Statement of Work
SQL	sample quantification limit
SVOCs	semivolatile organic compounds
TCDD	tetrachlorodibenzo-p-dioxin
TEF	Toxicity Equivalent Factor
TEQ	toxic equivalent quotient
UCL	upper concentration limit
UF	Uncertainty Factor
UR	Unit Risk
USACE	U.S. Army Corps of Engineers, New England District
USEPA	U.S. Environmental Protection Agency
VOCs	volatile organic compounds

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FIGURES

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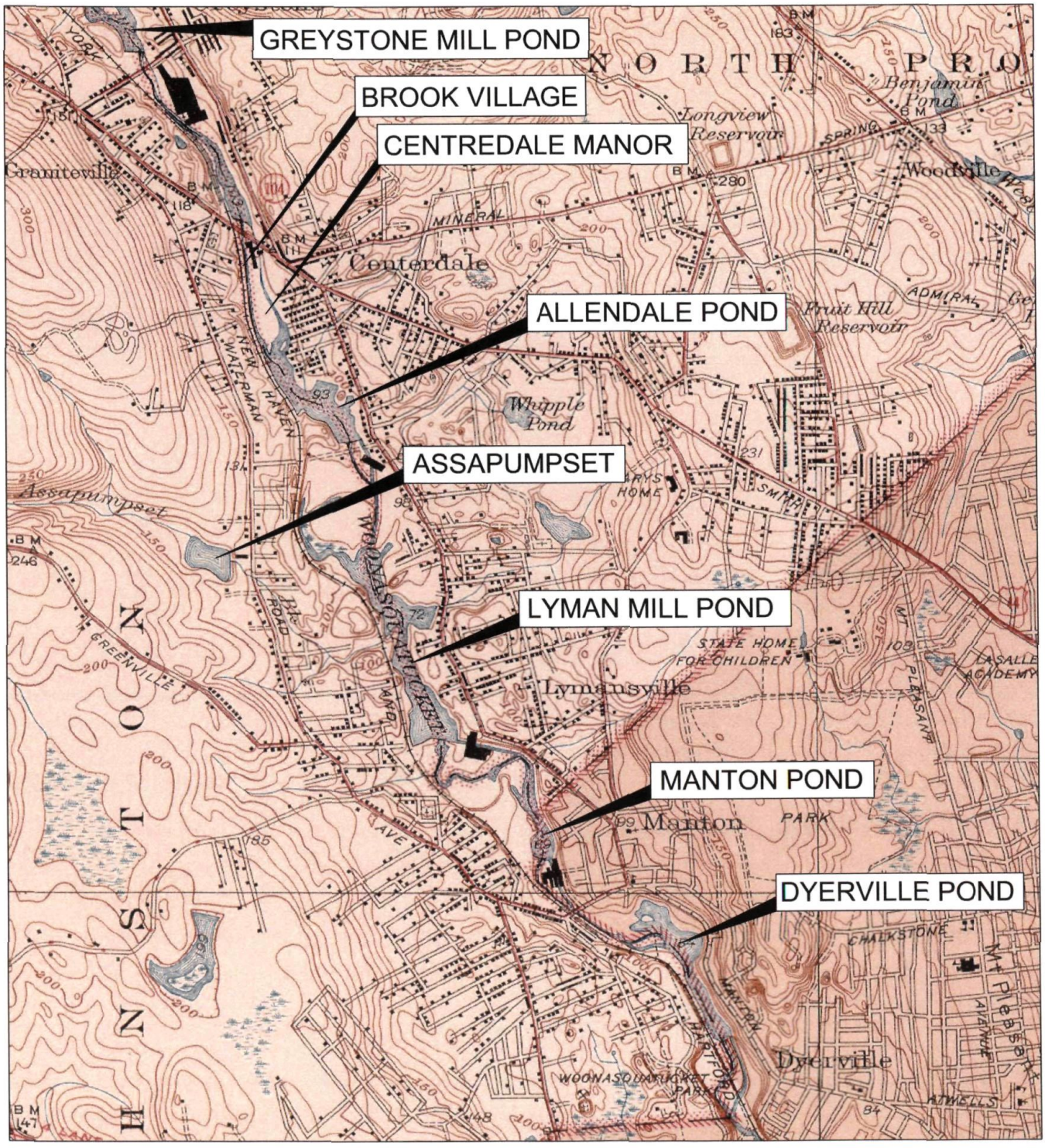

SOURCE: PORTION OF THE FOLLOWING U.S.G.S QUADRANGLE MAP: PROVIDENCE RI, 1957

GRAPHIC SCALE
0 0.5 MILE 1 MILE

N

FIGURE 1
SITE LOCATION MAP
CENTREDALE MANOR RESTORATION PROJECT
SUPERFUND SITE
NORTH PROVIDENCE, RHODE ISLAND

MACTEC

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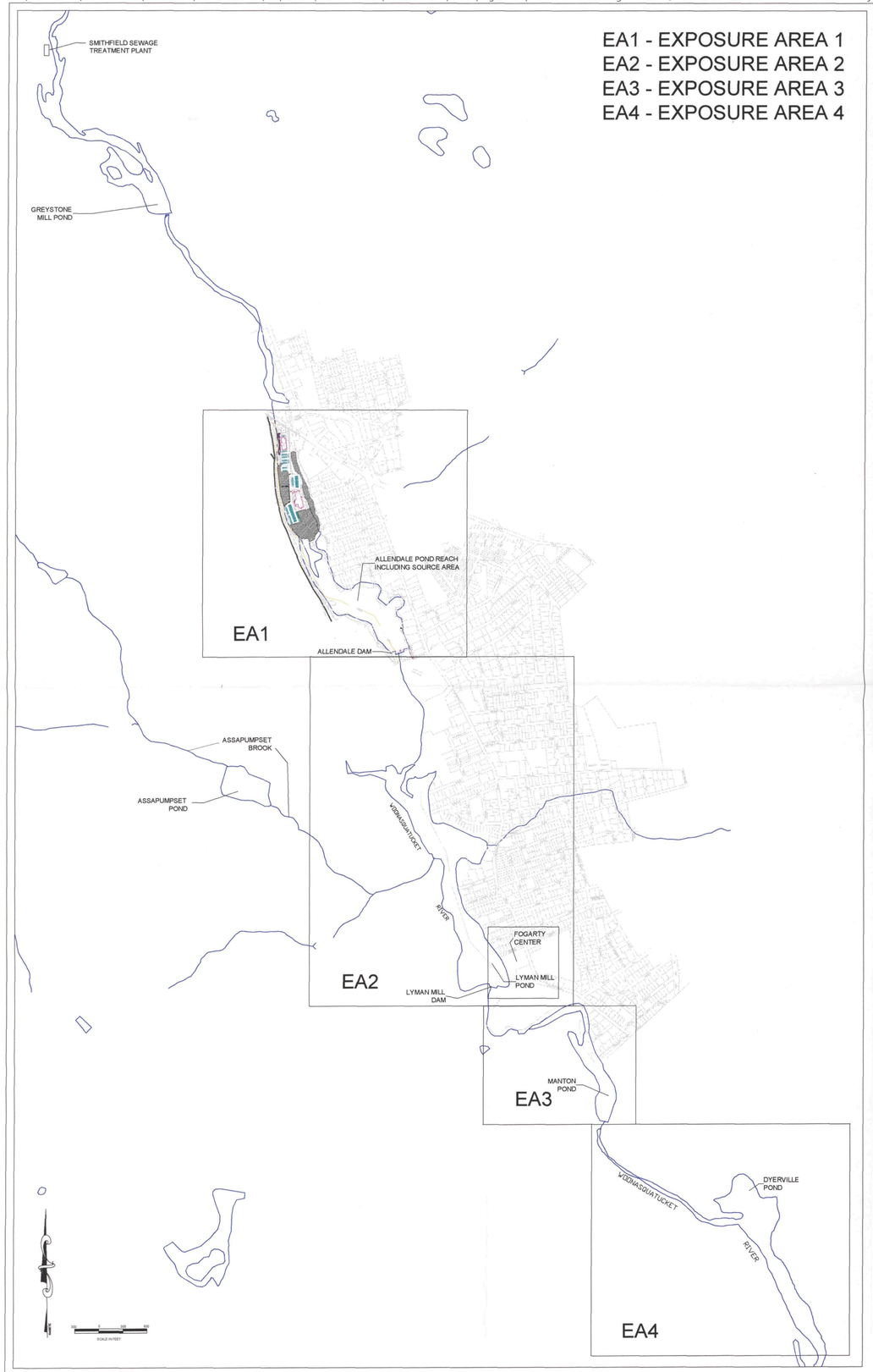
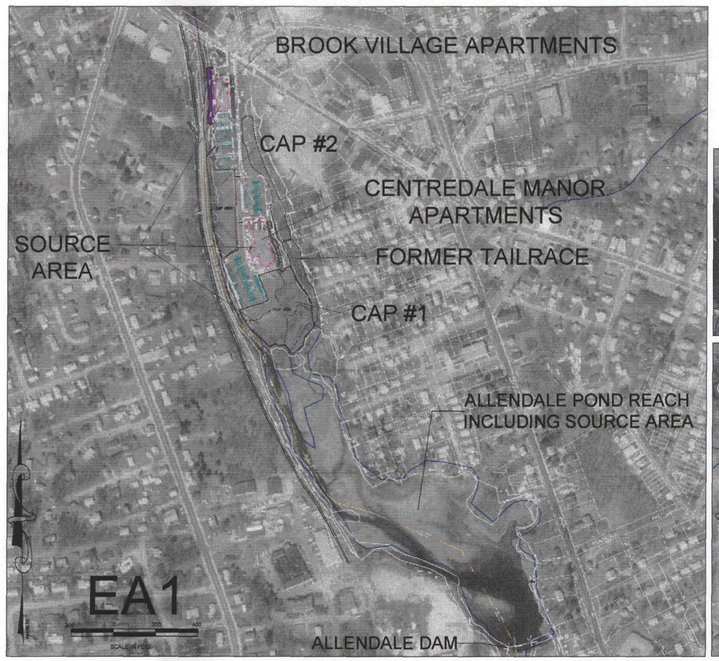
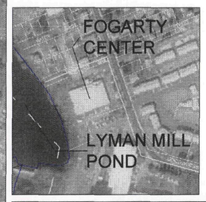
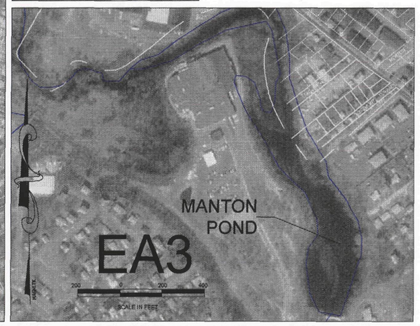
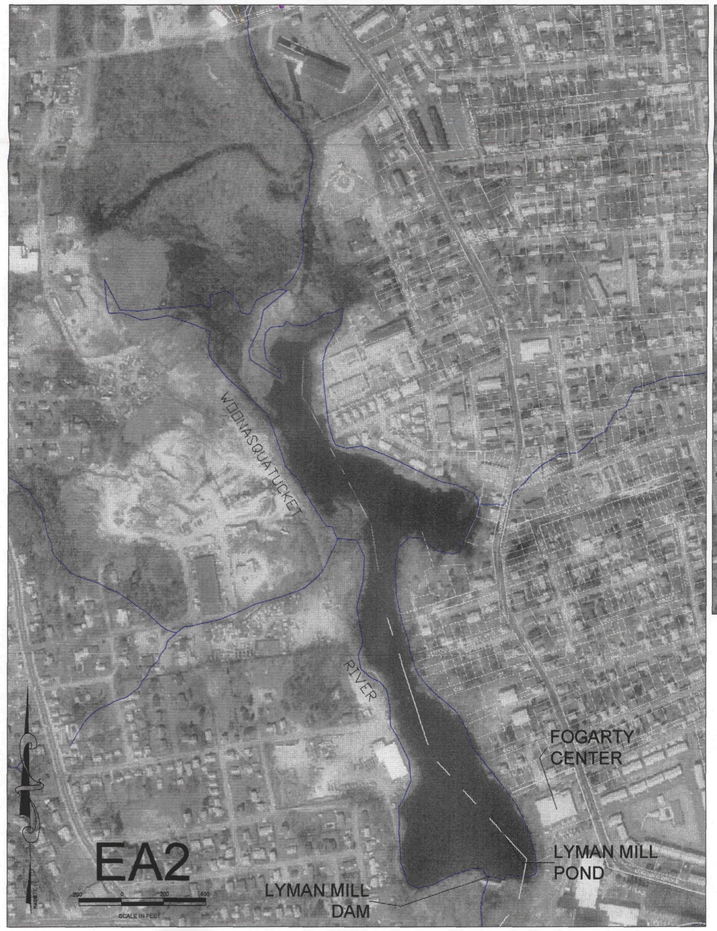
EA1 - EXPOSURE AREA 1
EA2 - EXPOSURE AREA 2
EA3 - EXPOSURE AREA 3
EA4 - EXPOSURE AREA 4











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107 Aububon Road
Building II, #301
Wakefield, MA 01880
781-245-6606

FIGURE 2
SITE LAYOUT AND SURROUNDING ENVIRONMENT
CENTREDALE MANOR RESTORATION PROJECT SUPERFUND SITE
NORTH PROVIDENCE, RHODE ISLAND

DRAWN	APPROVED	JOB NUMBER	DATE	REVISED DATE
KJA	MJM	51226	04/28/2004	

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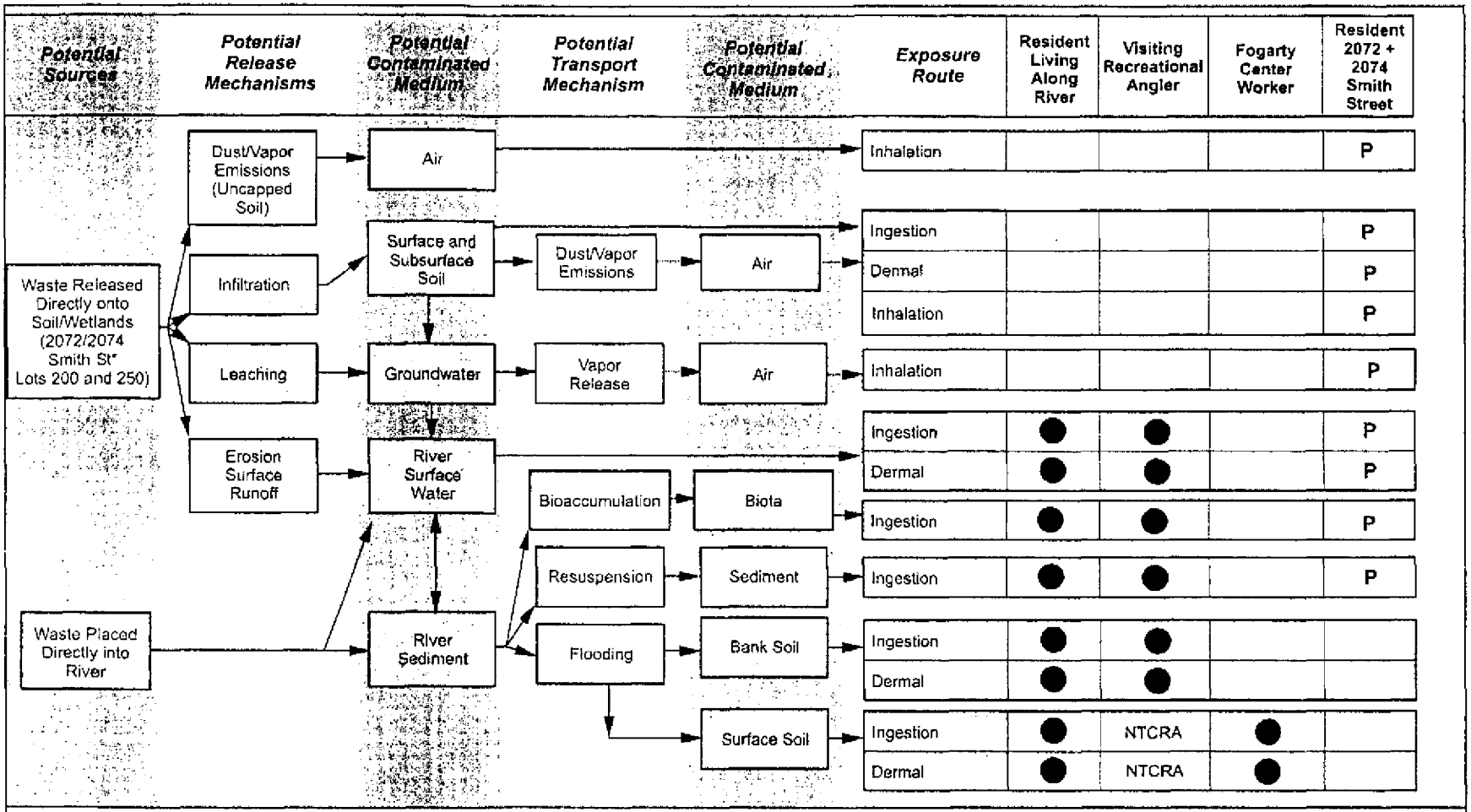

Legend

→ Potentially Significant Pathway
⋯► Unlikely to be a Significant Pathway
* Occupied by Brook Village and Centerdale Manor apartment buildings.

Notes:

Media, release/transport mechanisms, and exposure routes evaluated in the human health risk assessment are shaded
P=Previously evaluated
NTCRA=Evaluated as a component of Non-time Critical Removal Actions

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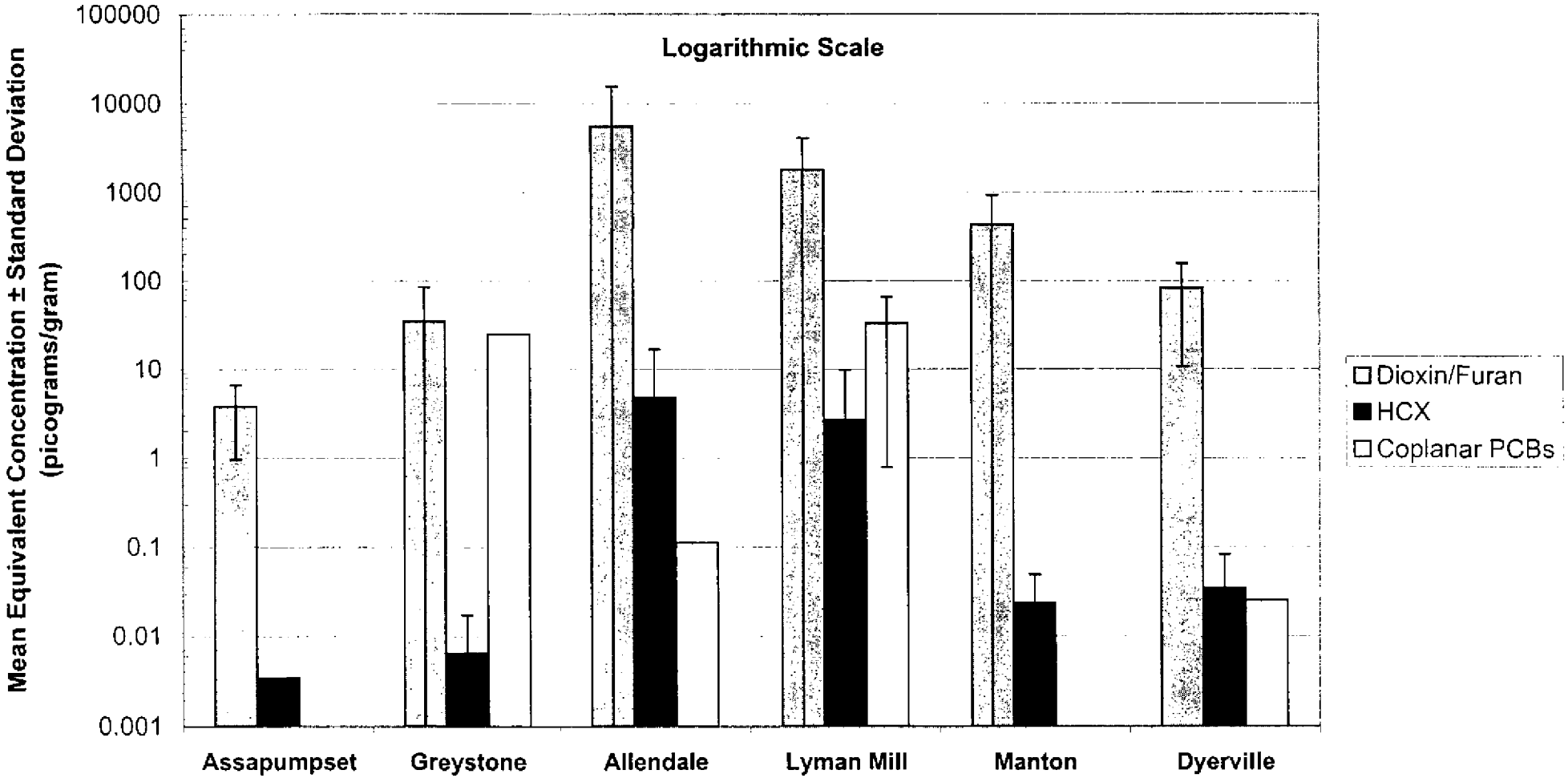
MACTEC Engineering and Consulting, Inc.
107 Audubon Road, Suite 301 - Wakefield, Massachusetts

Figure 3. Conceptual Site Model – Human Health
Baseline Human Health Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island

DRAWN:	JOB NUMBER:	FILE NUMBER:	DATE:	REVISED DATE:
BGF	51226-25.03	W2004020	5/04	5/13/04

Figure 4
Dioxin/Furan, HCX, and Coplanar PCB Equivalent Concentrations in Sediment

Baseline Ecological and Human Health Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island

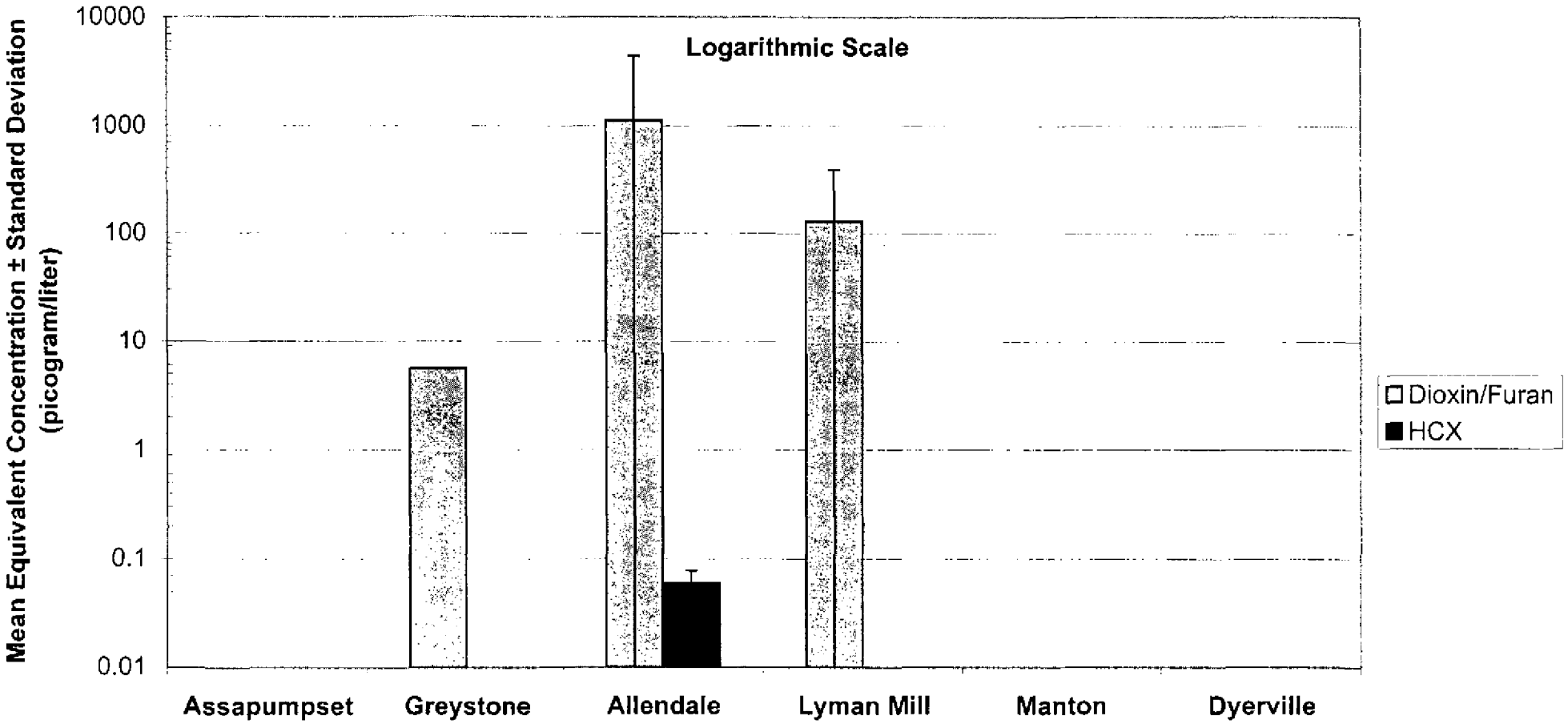


Mammalian toxicity equivalence factors (TEFs) per Van den Berg et al. 1998 were used to calculate equivalent concentrations for dioxins, furans, and coplanar PCBs, for consistency in comparing nature and extent of contamination only. Hexachloroxanthene (HCX) equivalent concentrations were calculated using a TEF of 0.0002. These equivalent concentrations are not intended to be representative of ecological risks for non-mammalian receptors.

Picograms/gram is equivalent to parts per trillion.

Figure 5
Dioxin/Furan and HCX Equivalent Concentrations in Surface Water

Baseline Ecological and Human Health Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



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Coplanar PCB analysis was not conducted for surface water.
Picograms per liter is equivalent to parts per quadrillion.

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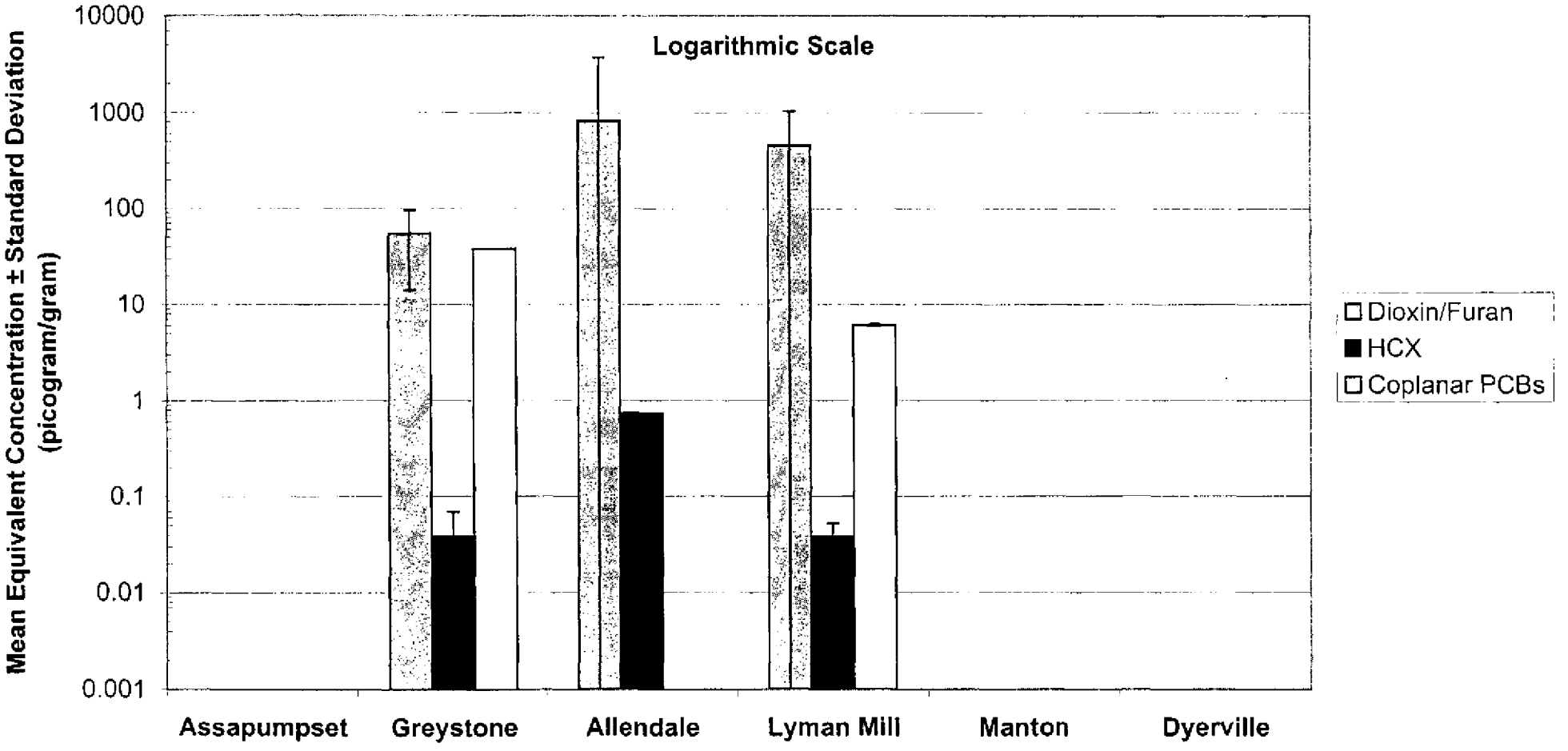
Figure 6
Dioxin/Furan, HCX, and Coplanar PCB Equivalent Concentrations in Floodplain and Bank Soil

Baseline Ecological and Human Health Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, RI



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Picogram/gram is equivalent to parts per trillion.

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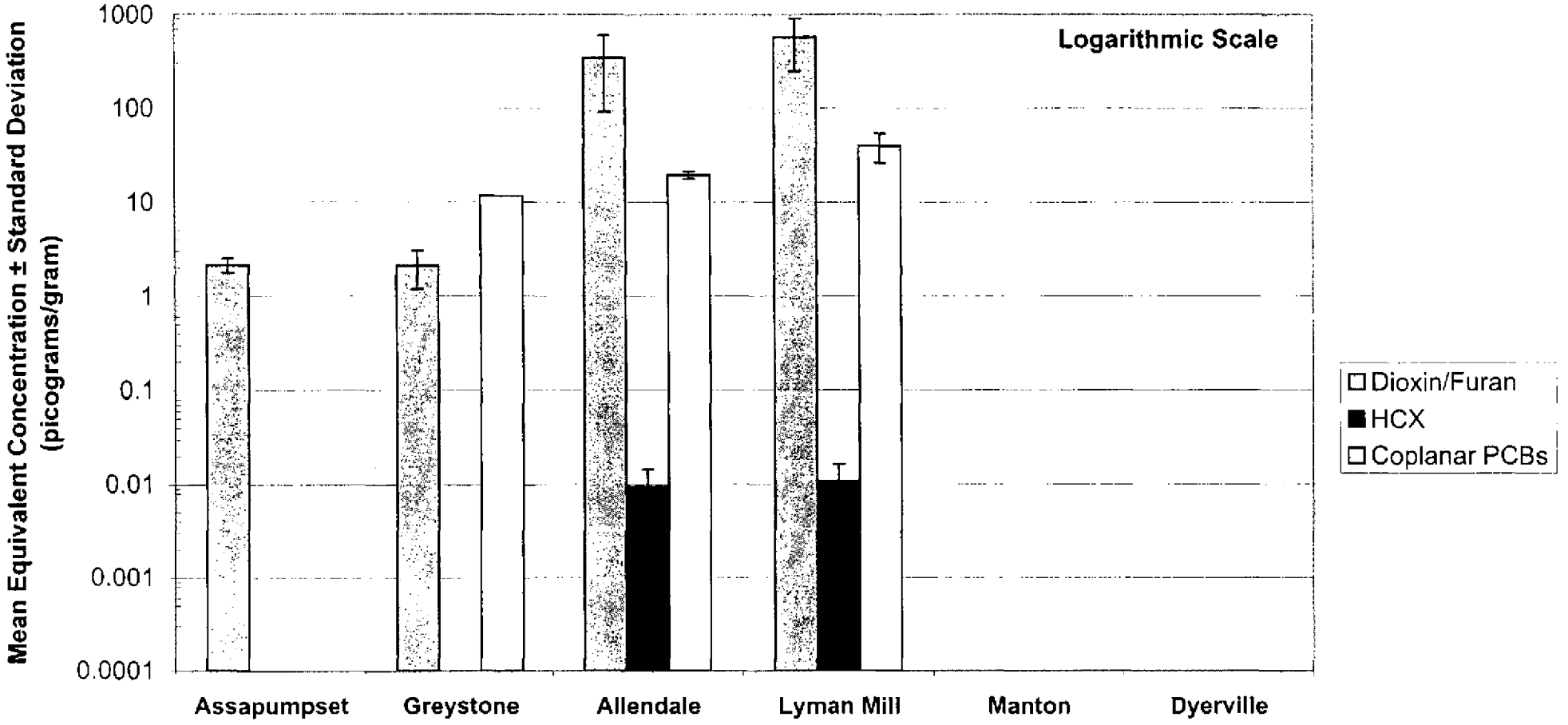
Figure 7
Dioxin/Furan, HCX, and Coplanar PCB Equivalent Concentrations in White Sucker

Baseline Ecological and Human Health Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



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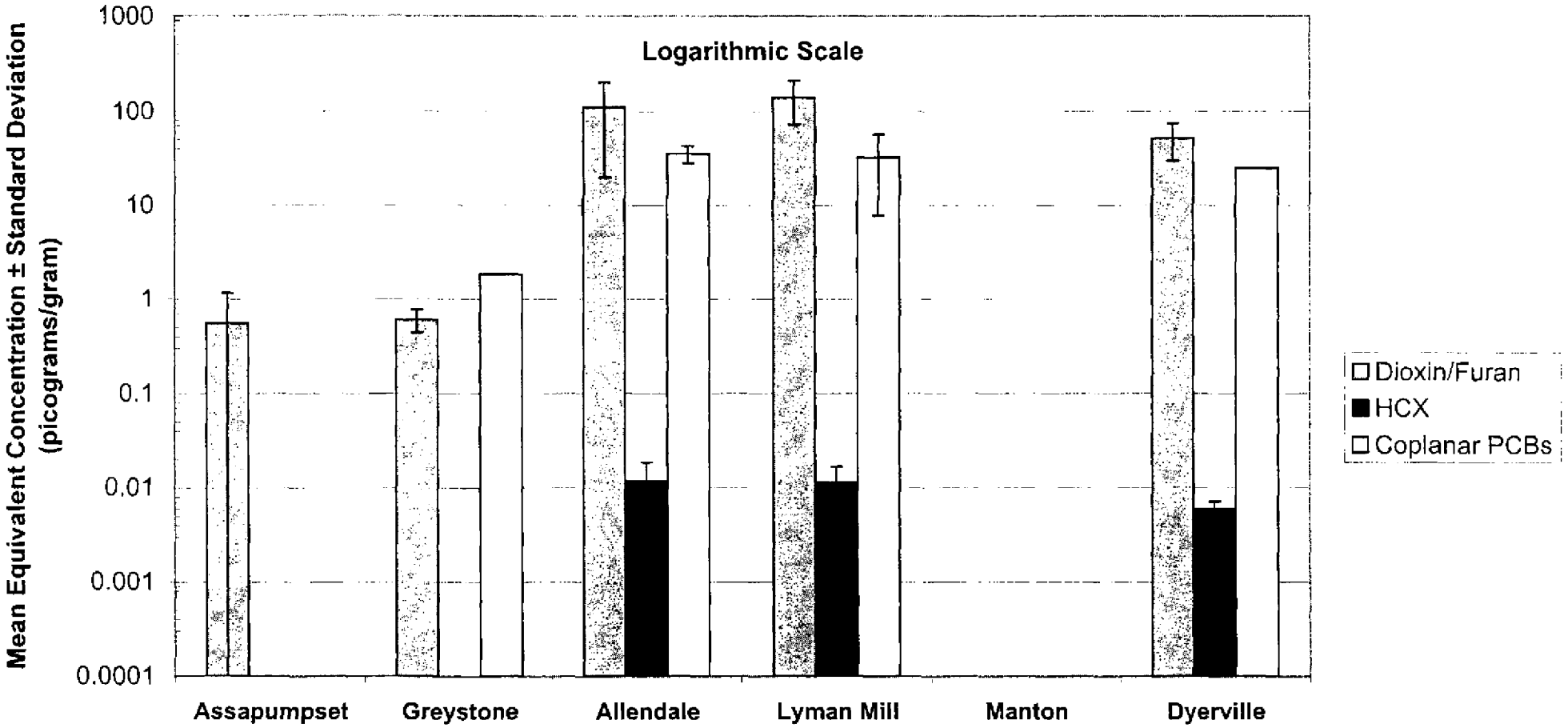
Picograms/gram is equivalent to parts per trillion.

Figure 8
Dioxin/Furan, HCX, and Coplanar PCB Equivalent Concentrations in American Eel

Baseline Ecological and Human Health Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



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Picograms/gram is equivalent to parts per trillion.

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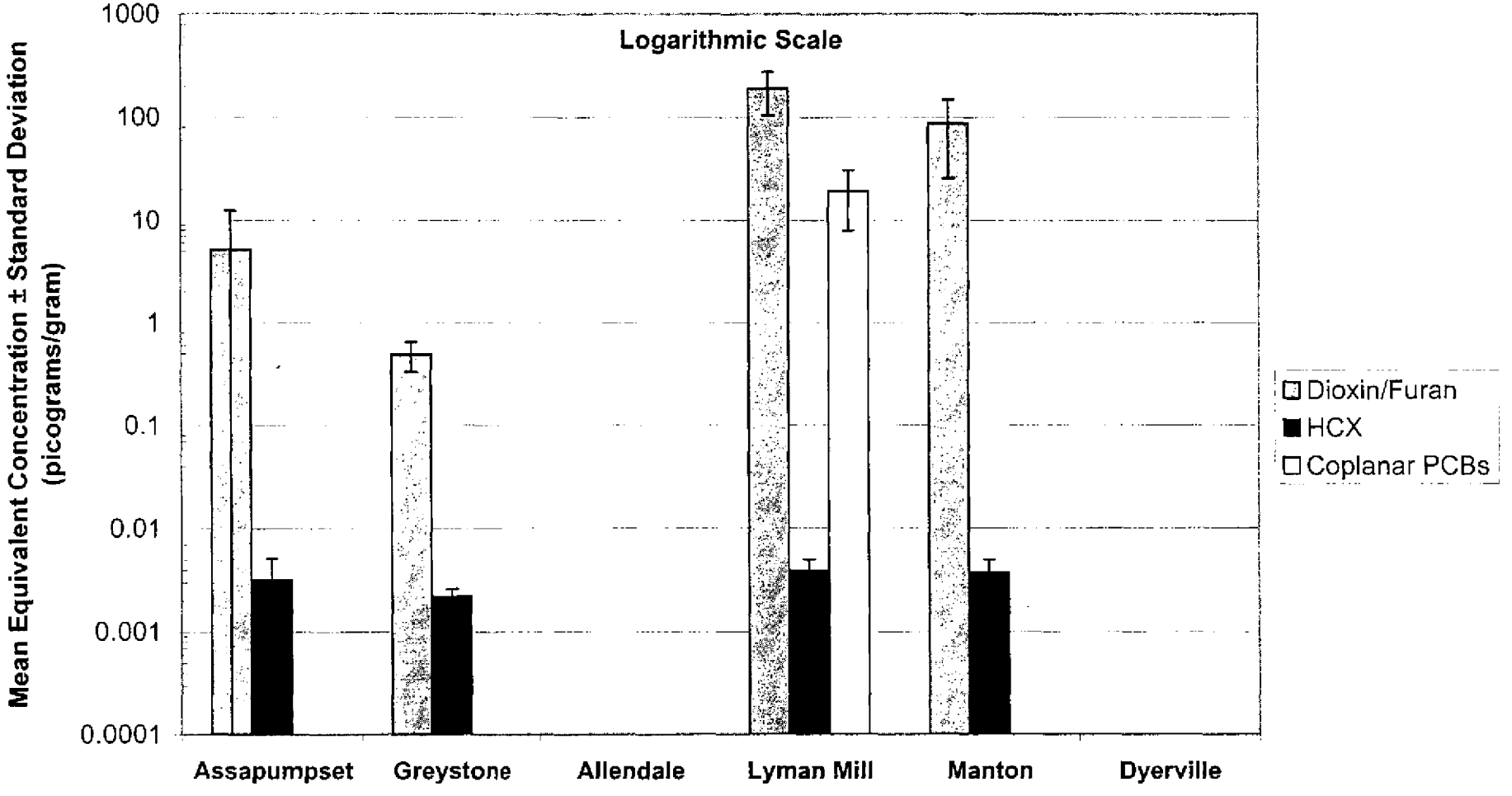
Figure 9

Dioxin/Furan, HCX, and Coplanar PCB Equivalent Concentrations in Largemouth Bass

Baseline Ecological and Human Health Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, RI



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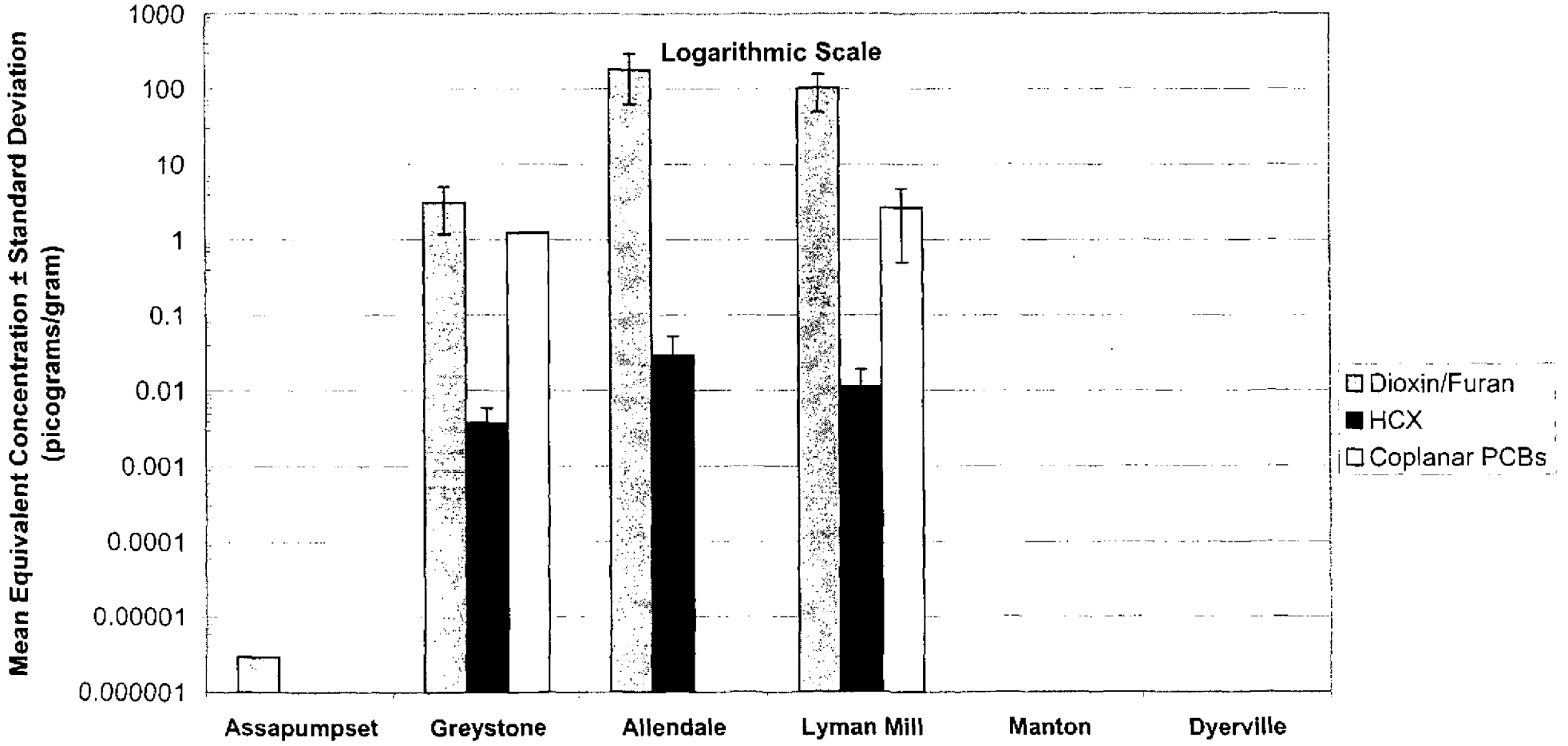
Reconstituted whole body data for Largemouth Bass used in developing histograms.

Picograms/gram is equivalent to parts per trillion.

Figure 10
Dioxin/Furan, HCX, and Coplanar PCB Equivalent Concentrations in Crayfish

Baseline Ecological and Human Health Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



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Picograms/gram is equivalent to parts per trillion.

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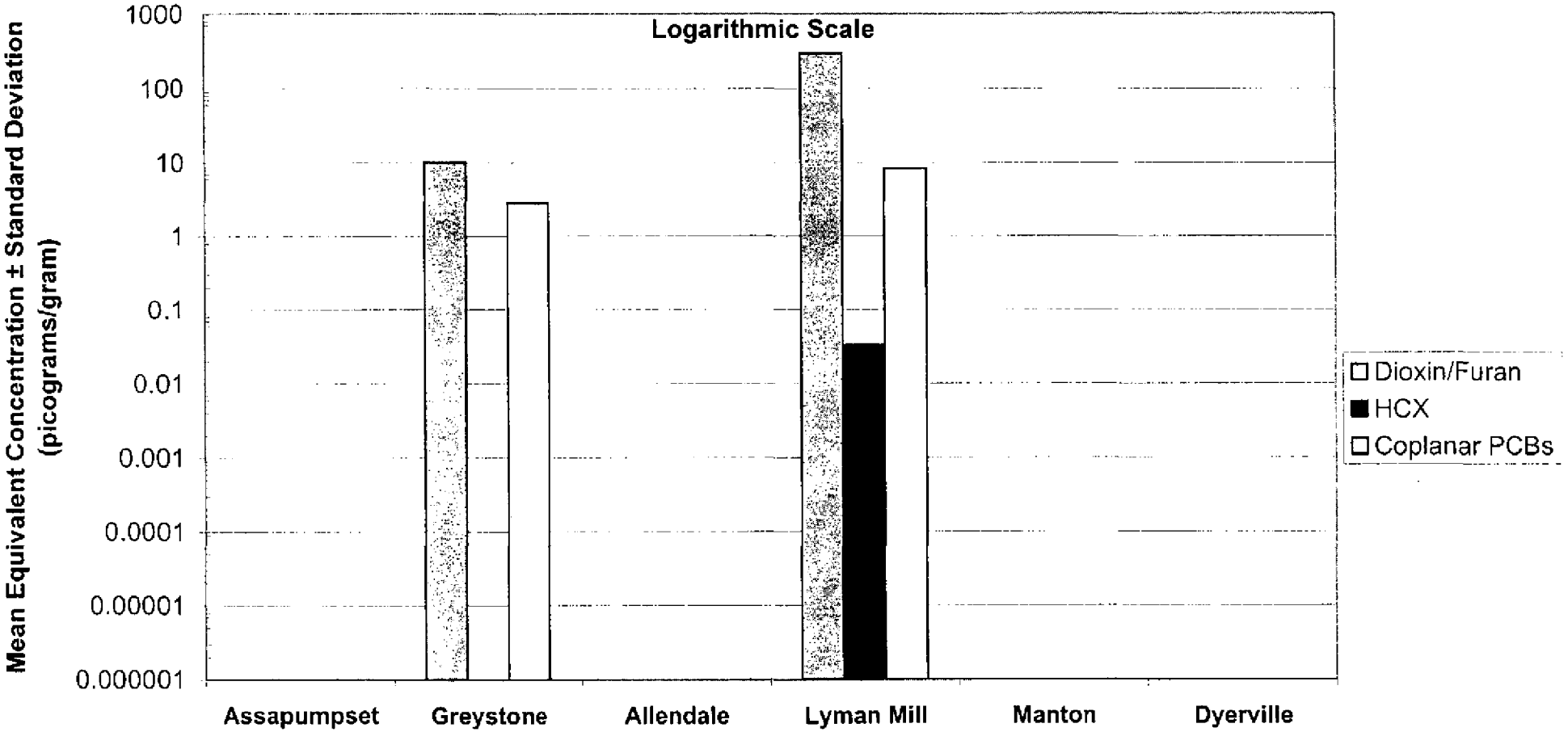
Figure 11
Dioxin/Furan, HCX, and Coplanar PCB Equivalent Concentrations in Emerging Insects

Baseline Ecological and Human Health Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, RI



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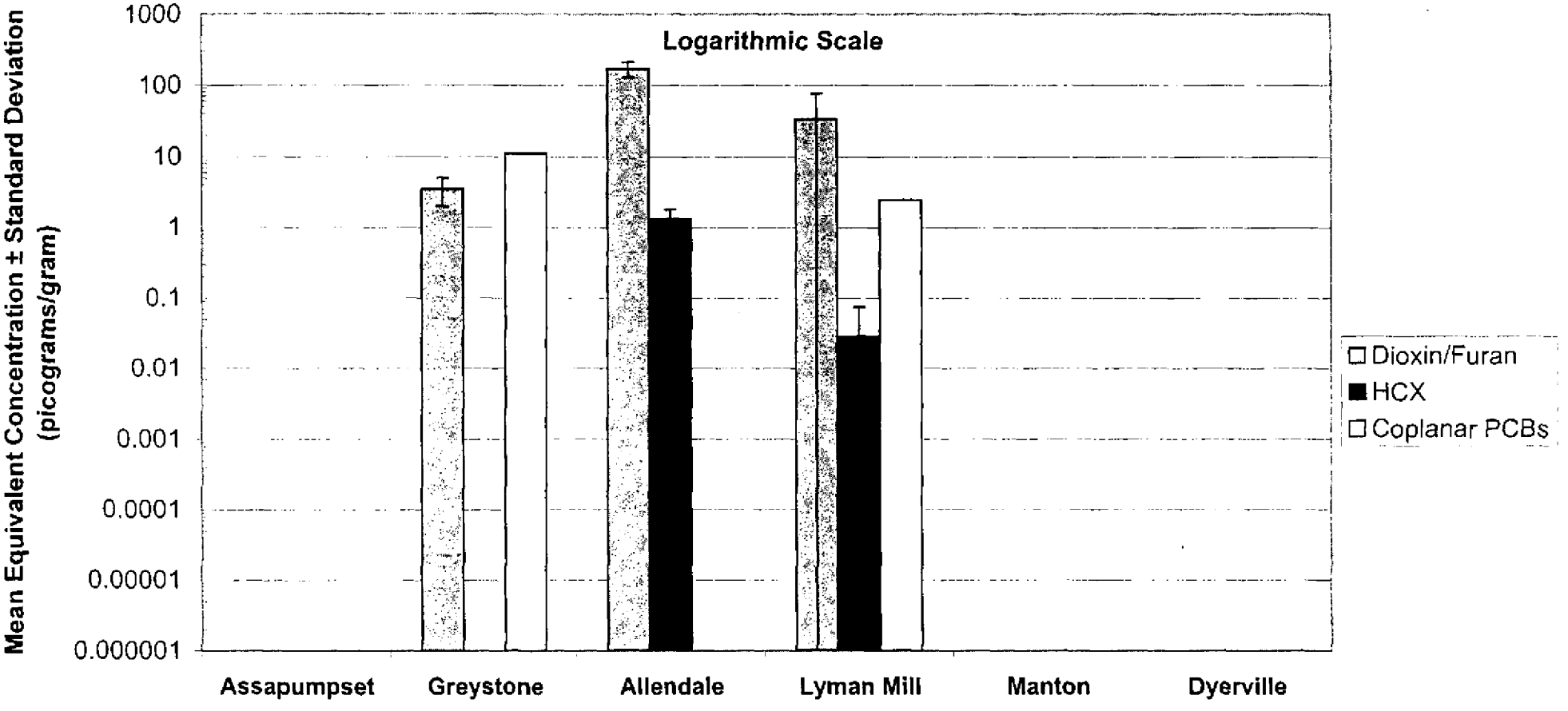
Picograms/gram is equivalent to parts per trillion.

Figure 12
Dioxin/Furan, HCX, and Coplanar PCB Equivalent Concentrations in Earthworms

Baseline Ecological and Human Health Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



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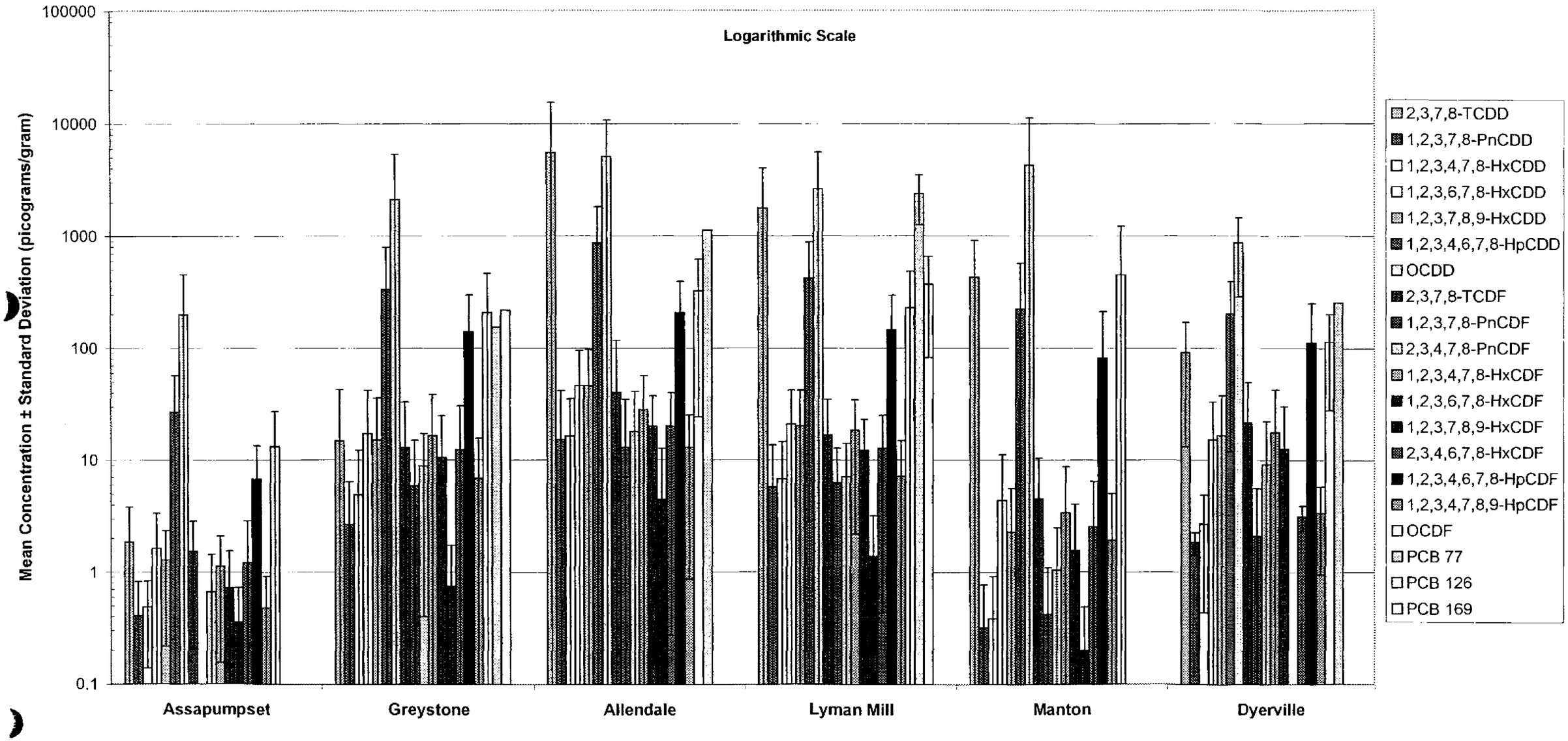
Picograms/gram is equivalent to parts per trillion.

Figure 13
Select Dioxin and PCB Congener Concentrations in Sediment

Baseline Ecological and Human Health Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



Picograms/gram is equivalent to parts per trillion.

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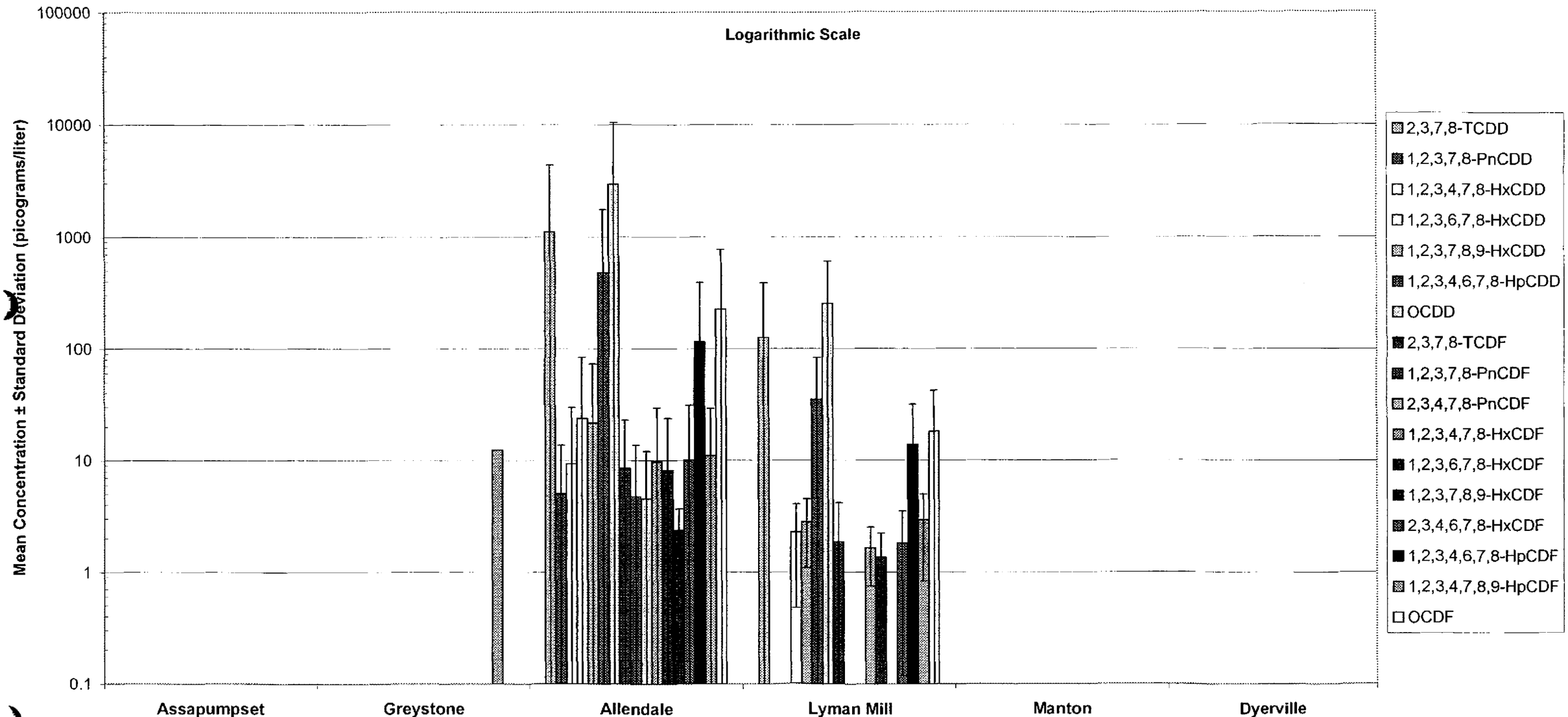
US EPA ARCHIVE DOCUMENT

Figure 14
Select Dioxin Concentrations in Surface Water

Baseline Ecological and Human Health Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



Coplanar PCB analysis was not conducted for surface water.
Picograms per liter is equivalent to parts per quadrillion.

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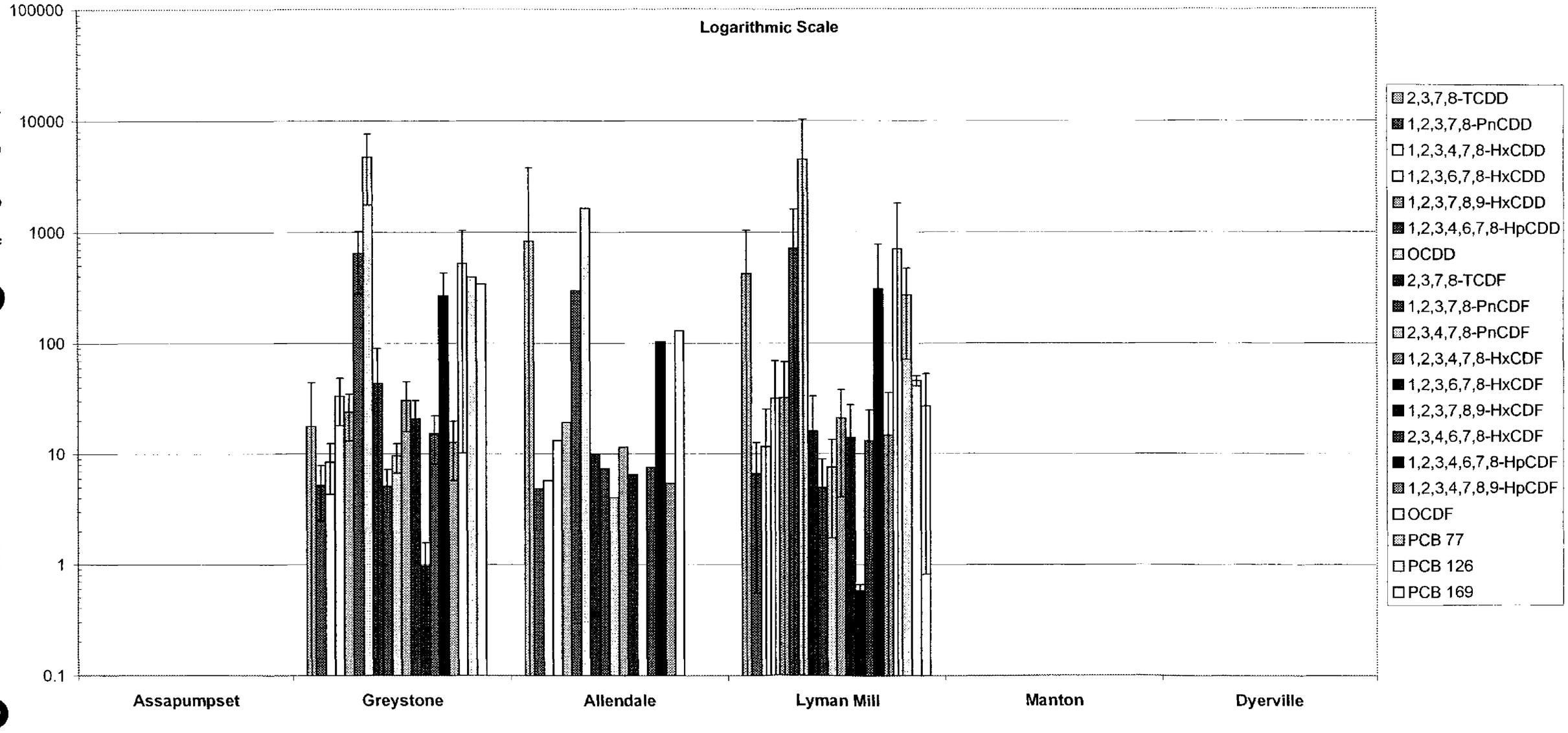
Figure 15
Select Dioxin and PCB Congener Concentrations in Floodplain and Bank Soil

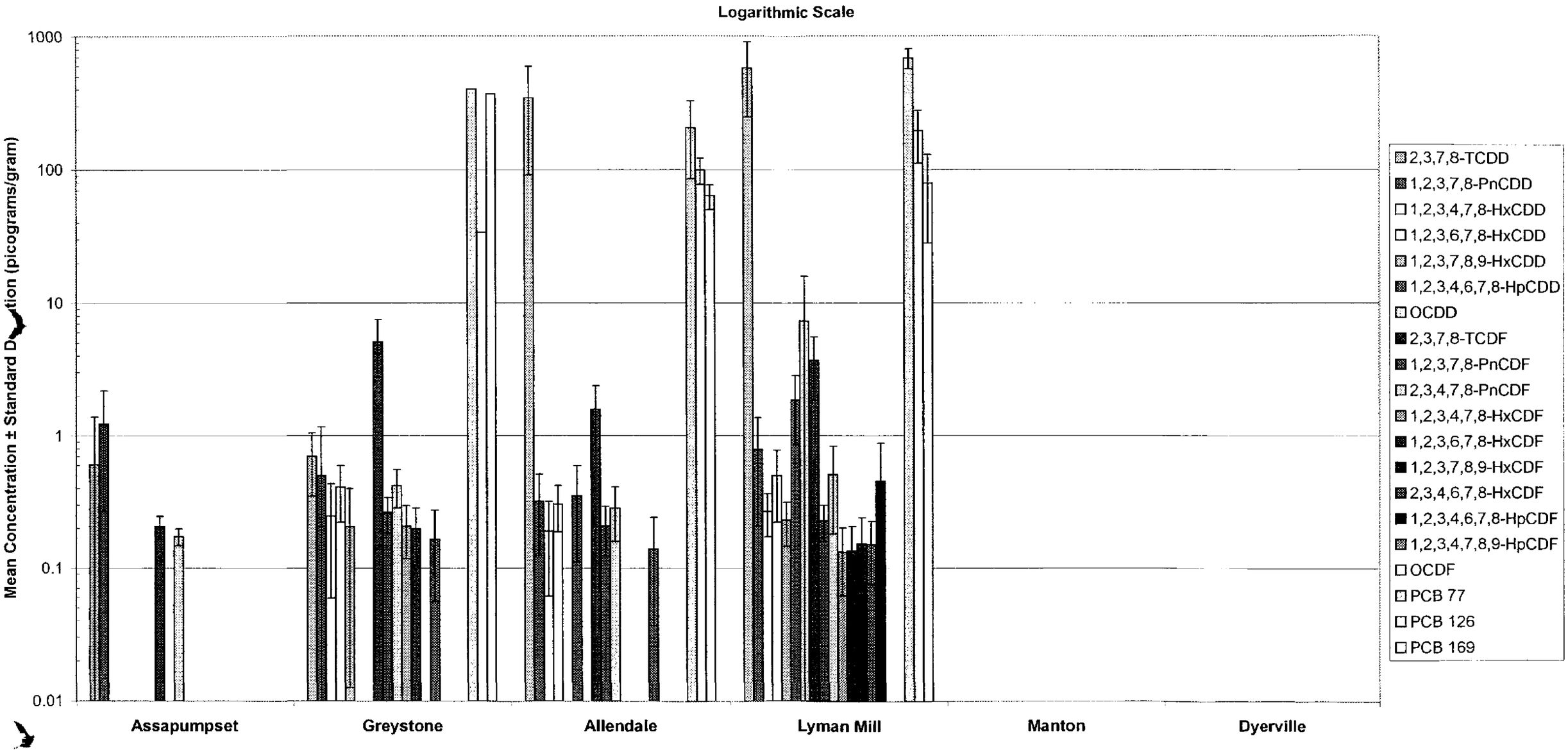
Baseline Ecological and Human Health Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



Picogram/gram is equivalent to parts per trillion.

Figure 16
Select Dioxin and PCB Congener Concentrations in White Sucker

Baseline Ecological and Human Health Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



Note: Brown Bullhead were used as a surrogate for White Sucker in Assapumpset Brook
Picograms/gram is equivalent to parts per trillion.

US EPA ARCHIVE DOCUMENT

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Figure 17
Select Dioxin and PCB Congener Concentrations in American Eel

Baseline Ecological and Biota Consumption Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island

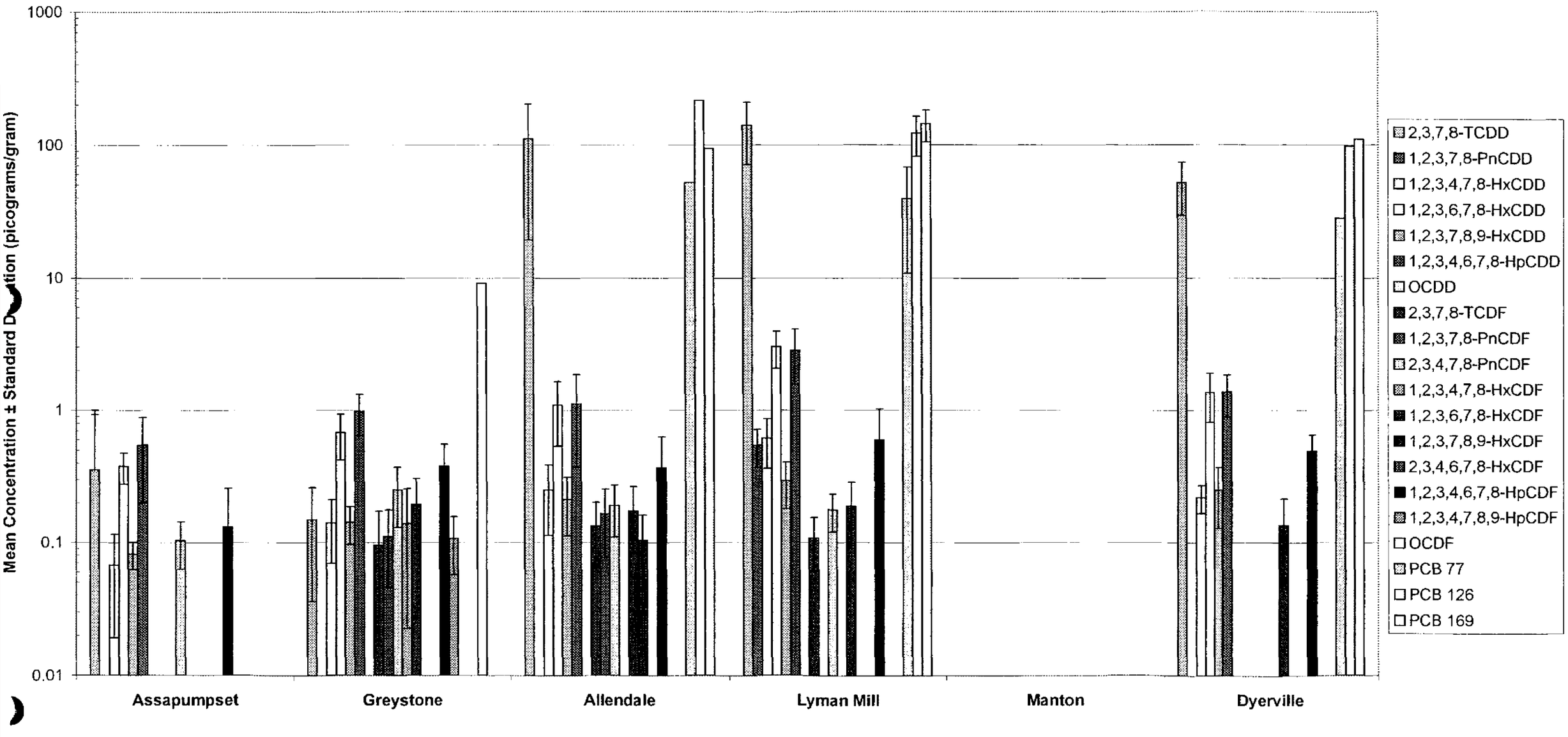
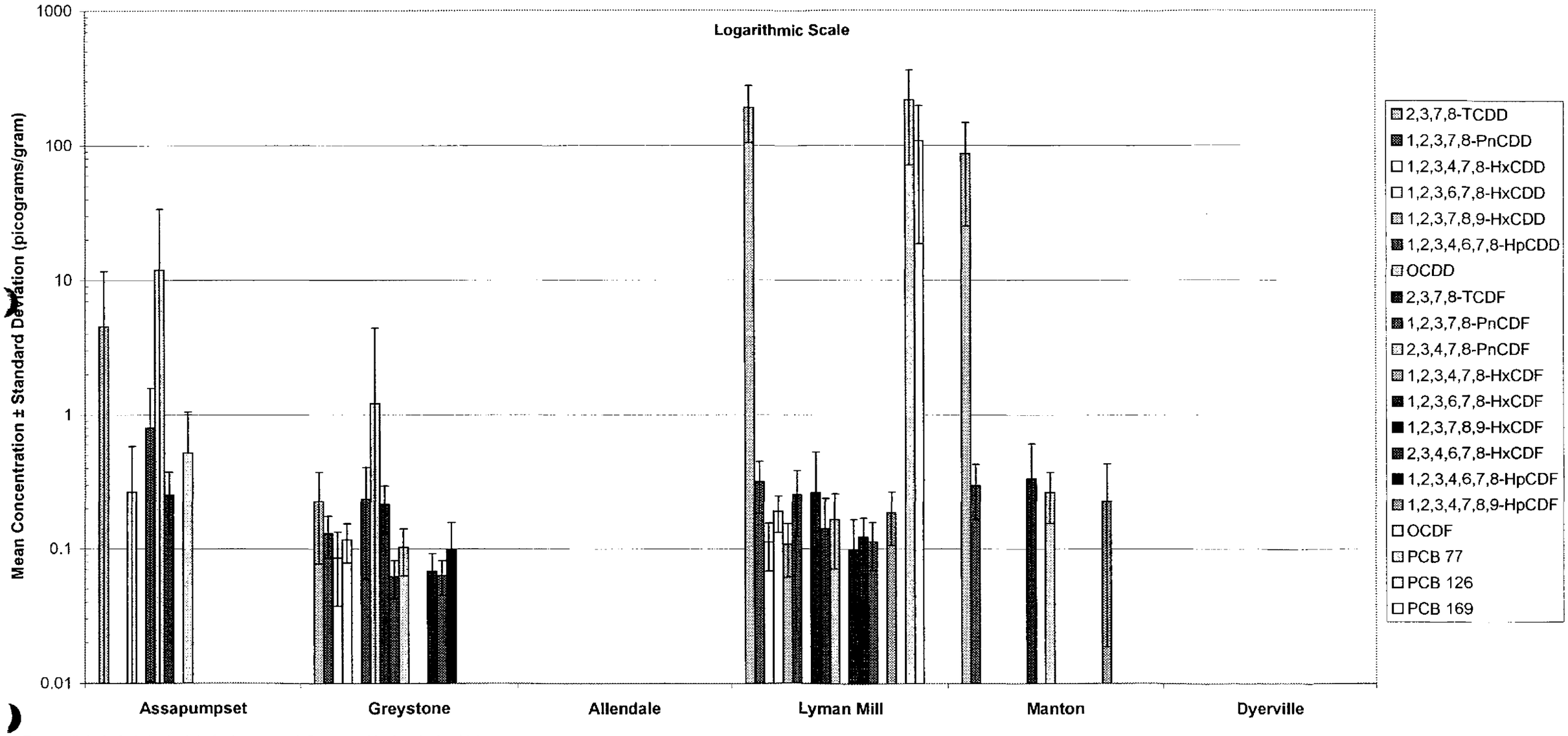


Figure 18
Select Dioxin and PCB Congener Concentrations in Largemouth Bass

Baseline Ecological and Human Health Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



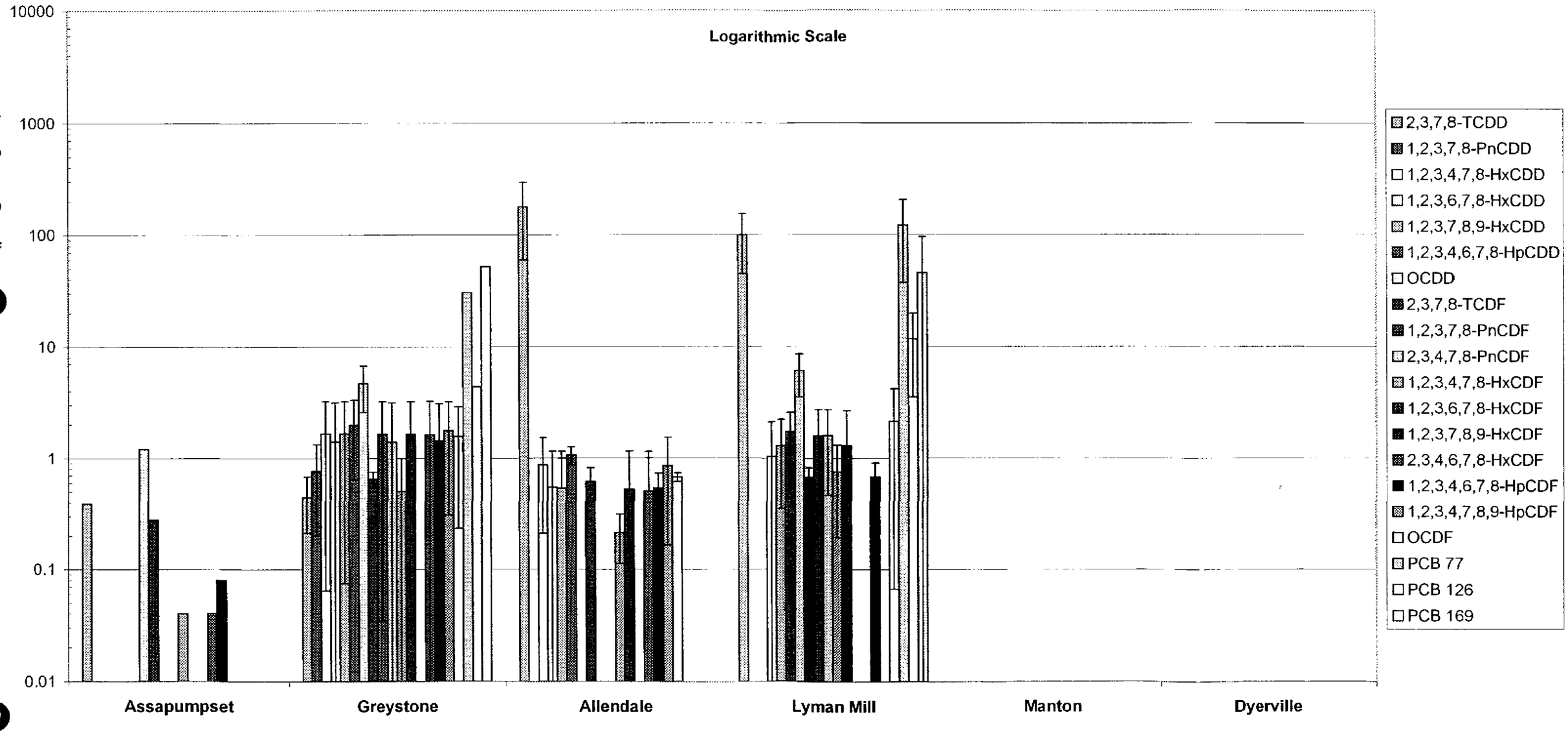
Reconstituted whole body data for Largemouth Bass used in developing histograms.
Picogram/gram is equivalent to parts per trillion.

US EPA ARCHIVE DOCUMENT

US EPA ARCHIVE DOCUMENT

Figure 19
Select Dioxin and PCB Congener Concentrations in Crayfish

Baseline Ecological and Human Health Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island

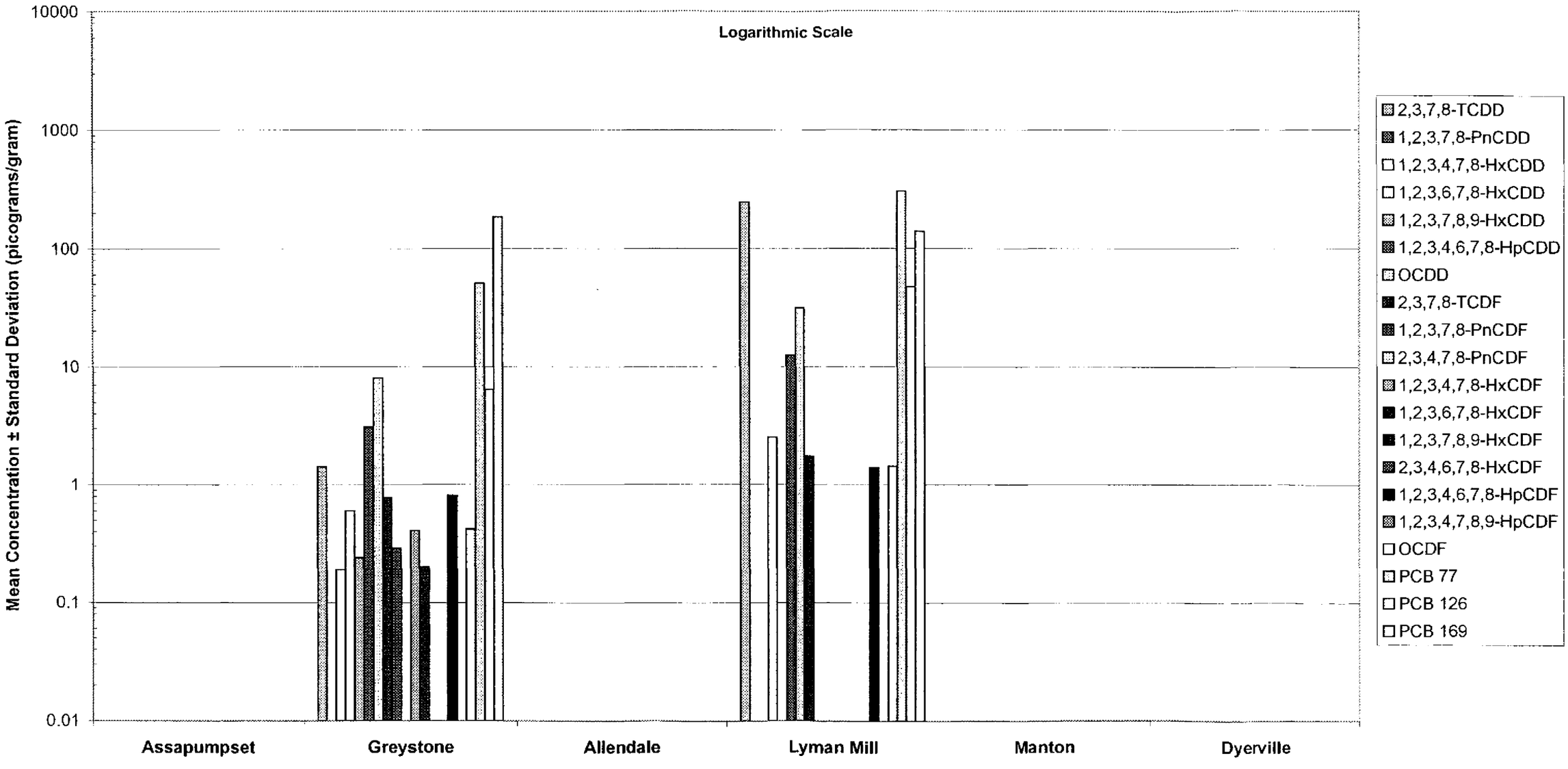


Picogram/gram is equivalent to parts per trillion.

US EPA ARCHIVE DOCUMENT

Figure 20
Select Dioxin and PCB Congener Concentrations in Emerging Insects

Baseline Ecological and Human Health Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island

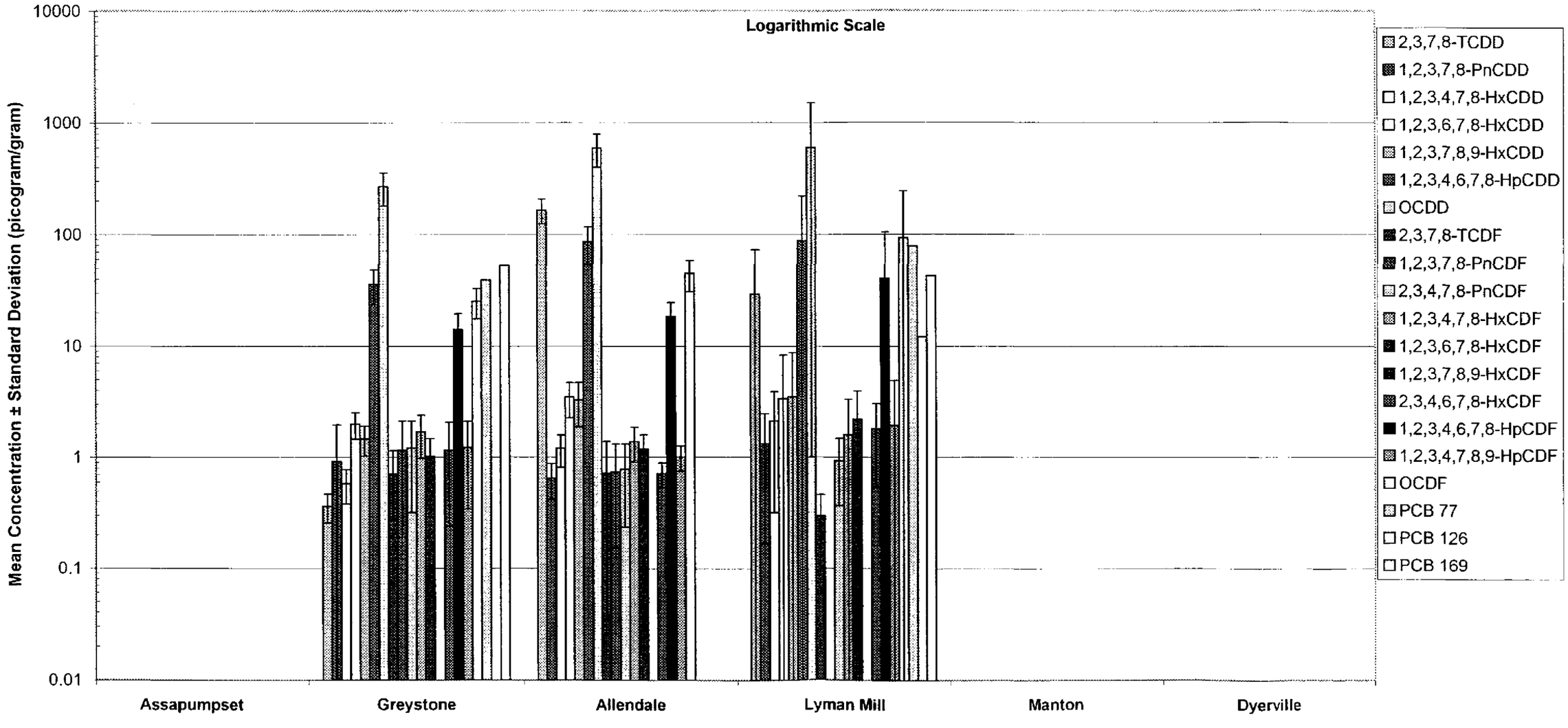


Picogram/gram is equivalent to parts per trillion.

US EPA ARCHIVE DOCUMENT

Figure 21
Select Dioxin and PCB Congener Concentrations in Earthworms

Baseline Ecological and Human Health Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island

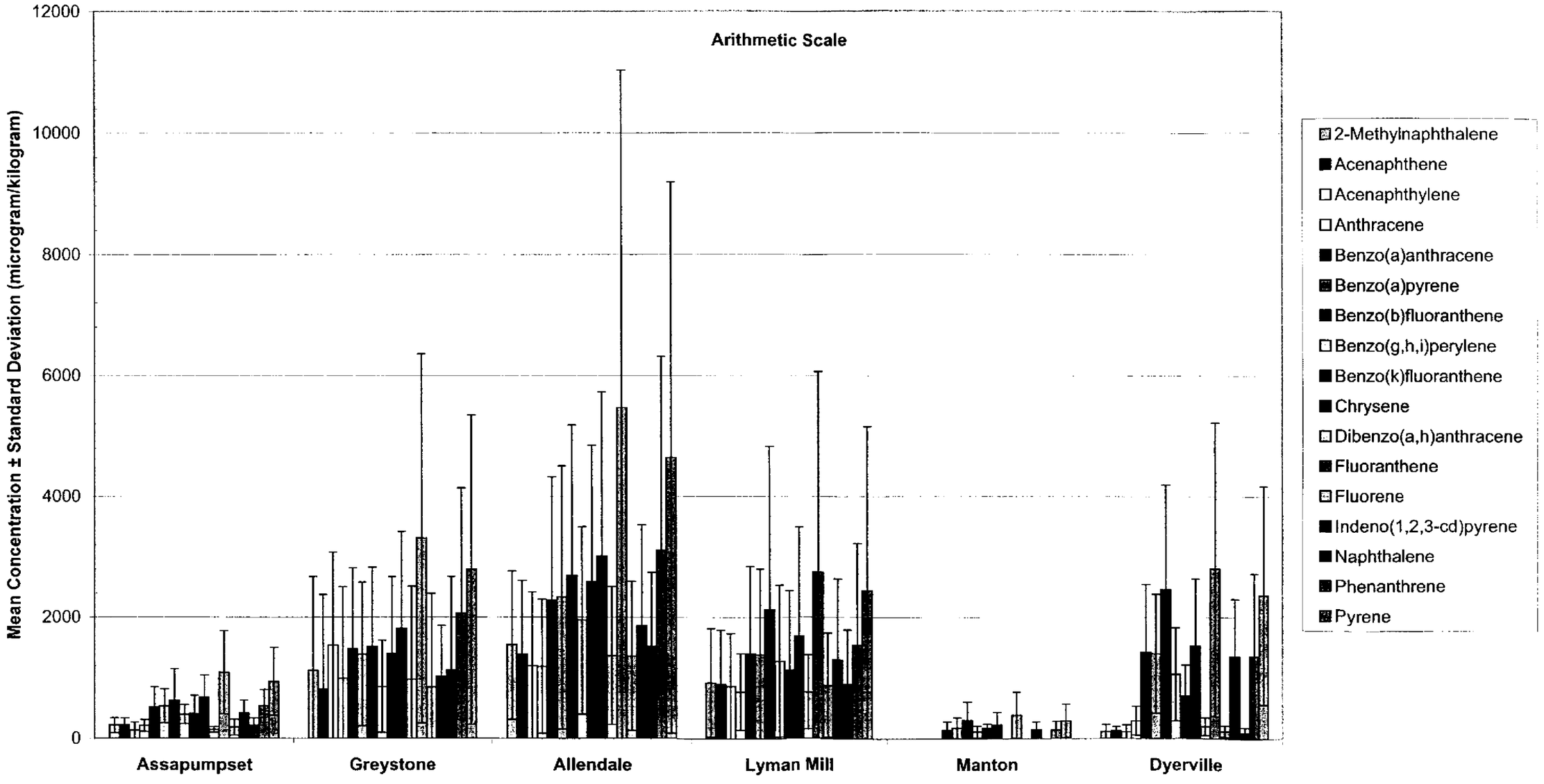


Picogram/gram is equivalent to parts per trillion.

US EPA ARCHIVE DOCUMENT

Figure 22
Select PAH Concentrations in Sediment

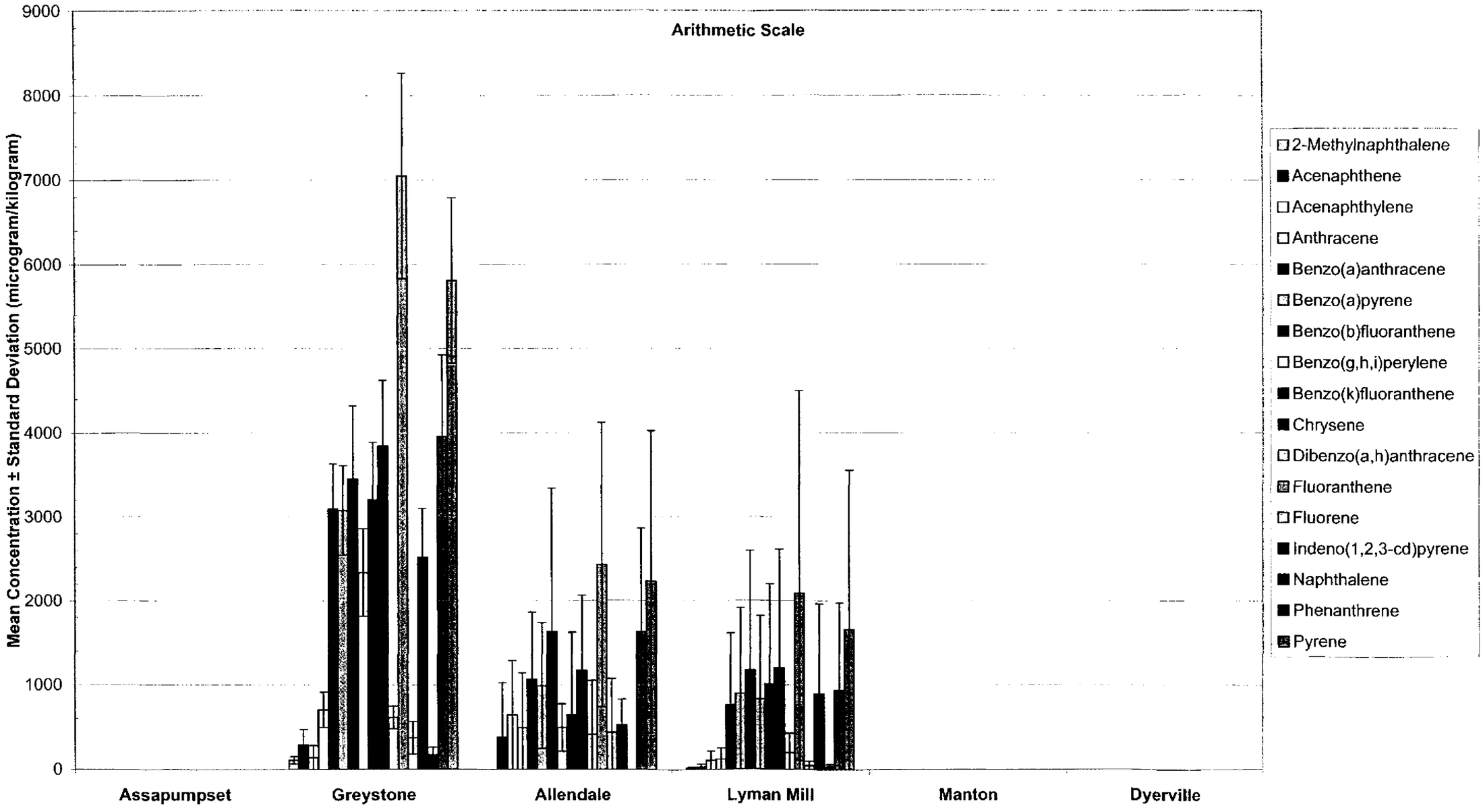
Baseline Ecological and Human Health Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



US EPA ARCHIVE DOCUMENT

Figure 23
Select PAH Concentrations in Floodplain and Bank Soil

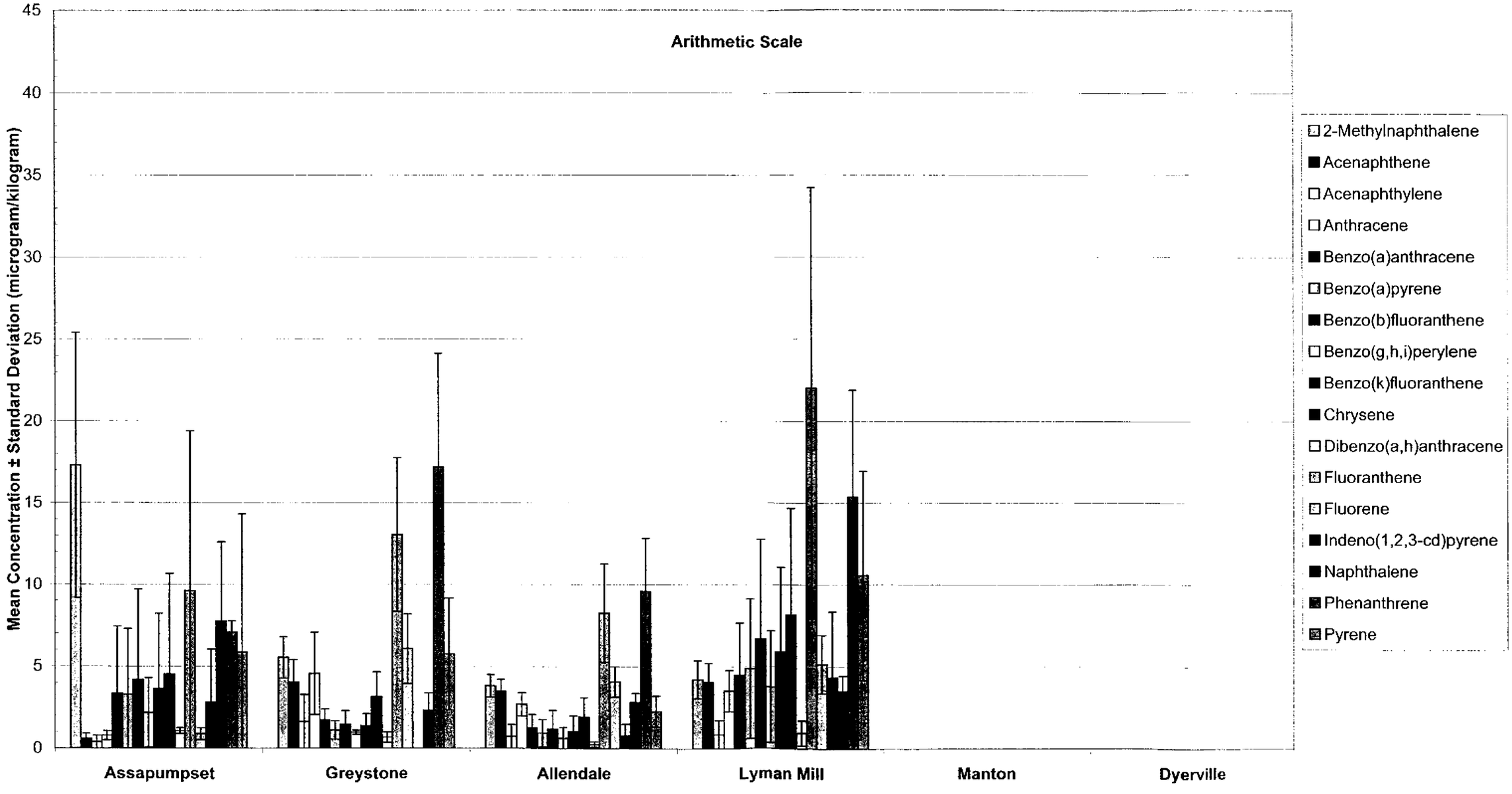
Baseline Ecological and Human Health Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



US EPA ARCHIVE DOCUMENT

Figure 24
Select PAH Concentrations in White Sucker

Baseline Ecological and Human Health Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island

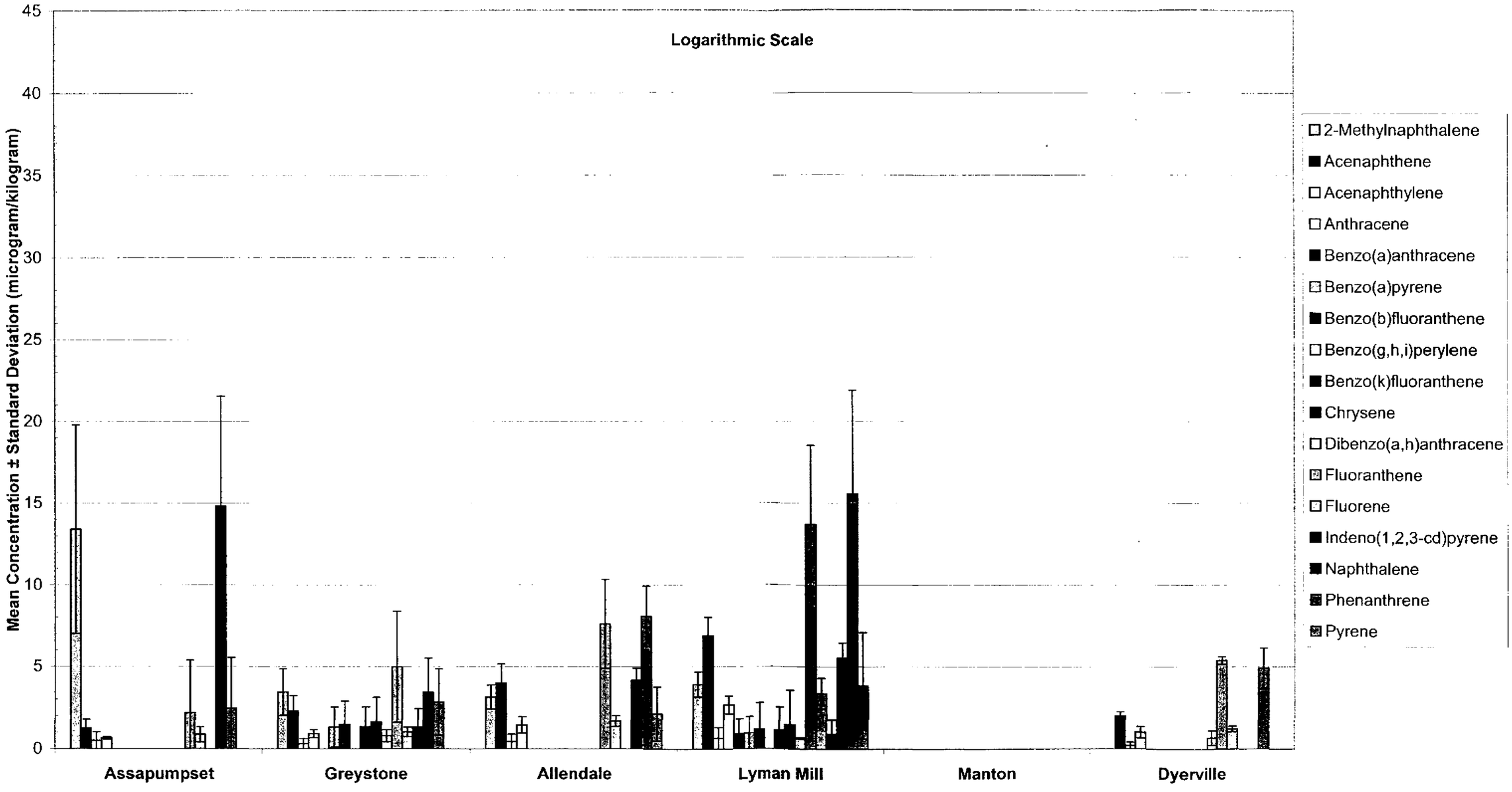


Note: Brown Bullhead were used as a surrogate for White Sucker in Assapumpset Brook

US EPA ARCHIVE DOCUMENT

Figure 25
Select PAH Concentrations in American Eel

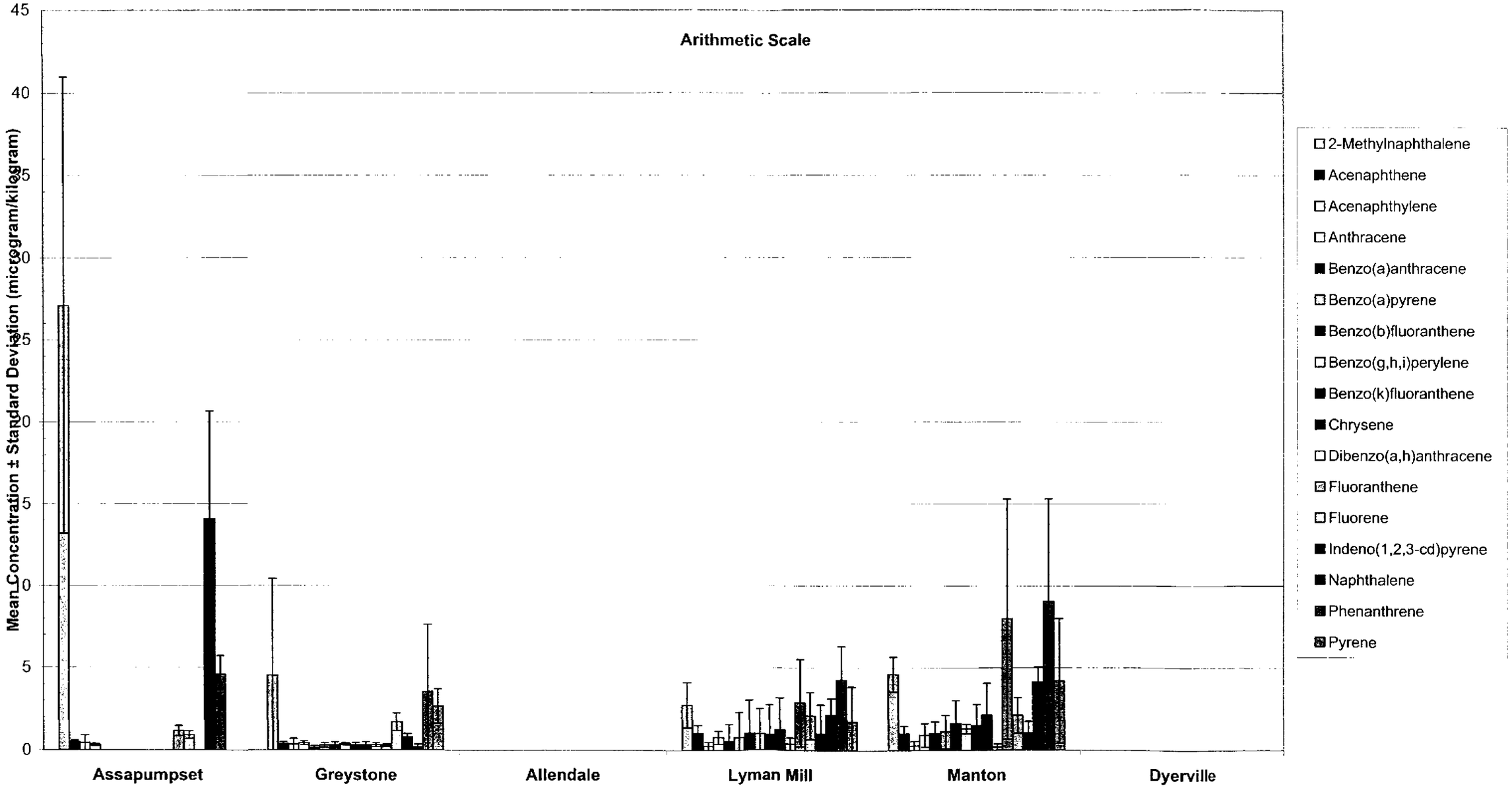
Baseline Ecological and Human Health Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



US EPA ARCHIVE DOCUMENT

Figure 26
Select PAH Concentrations in Largemouth Bass

Baseline Ecological and Human Health Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island

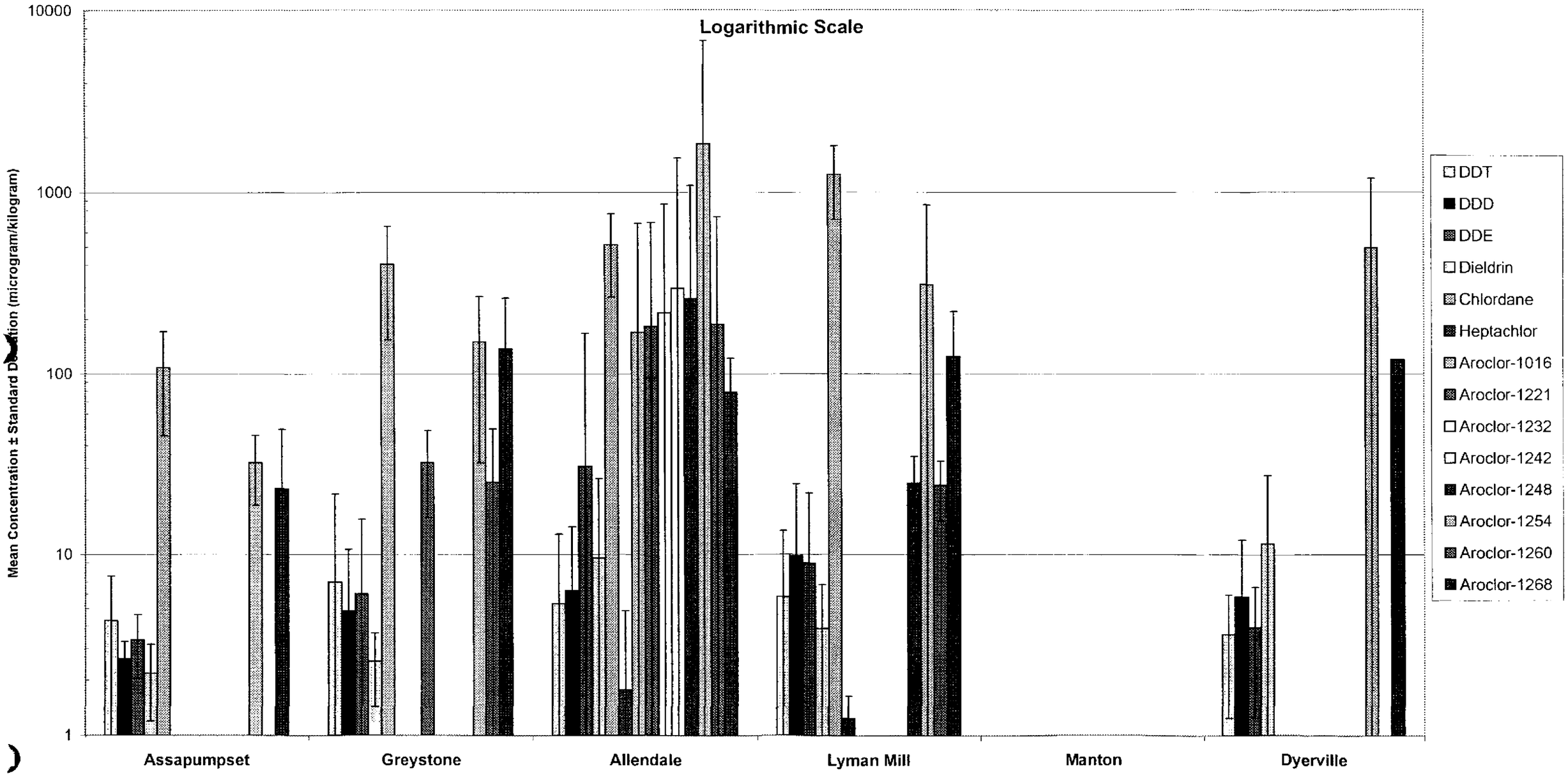


Reconstituted whole body data for Largemouth Bass used in developing histograms.

US EPA ARCHIVE DOCUMENT

Figure 27
Select Pesticide and PCB Aroclor Concentrations in Sediment

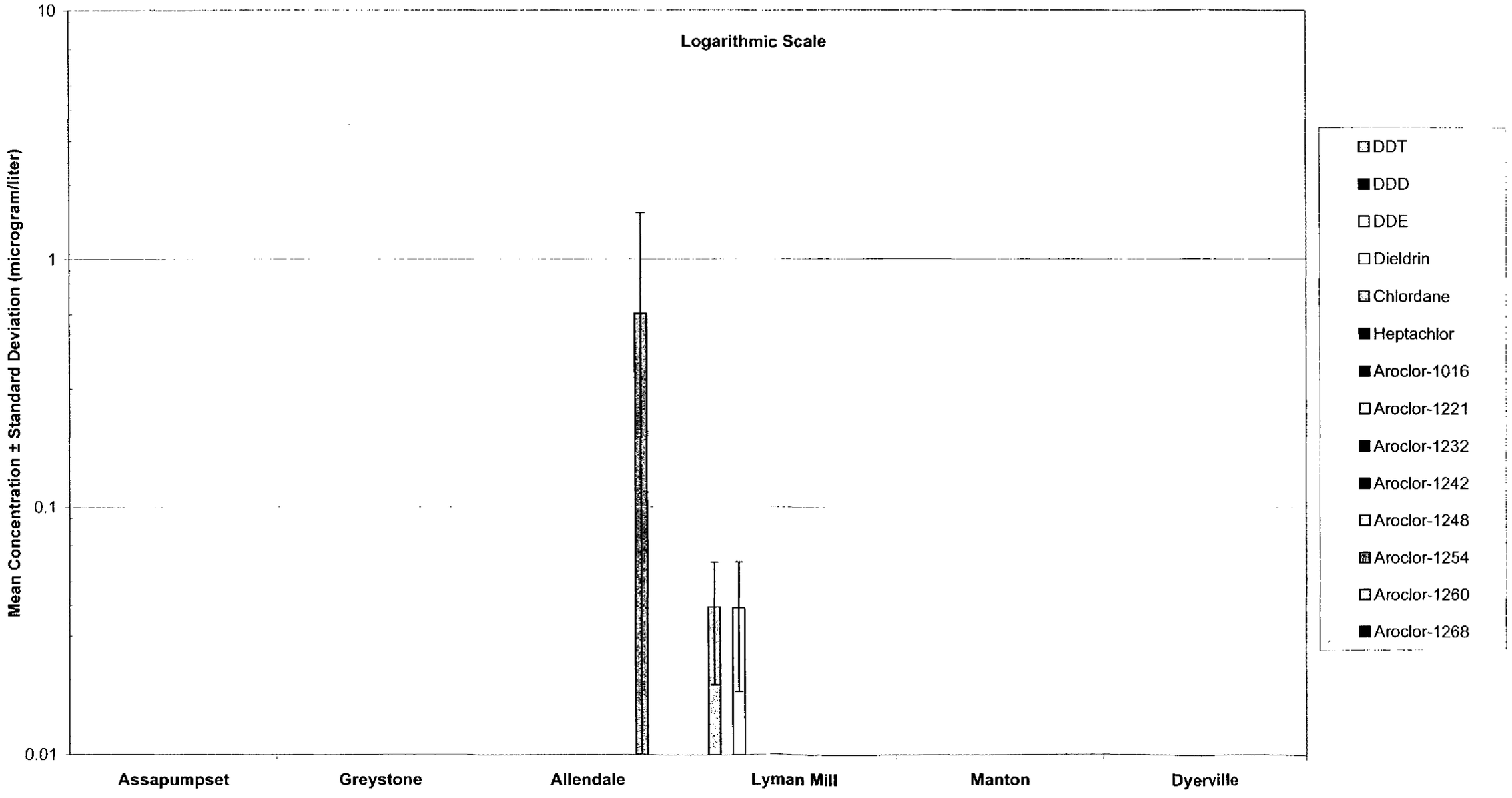
Baseline Ecological and Human Health Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



US EPA ARCHIVE DOCUMENT

Figure 28
Select Pesticide and PCB Aroclor Concentrations in Surface Water

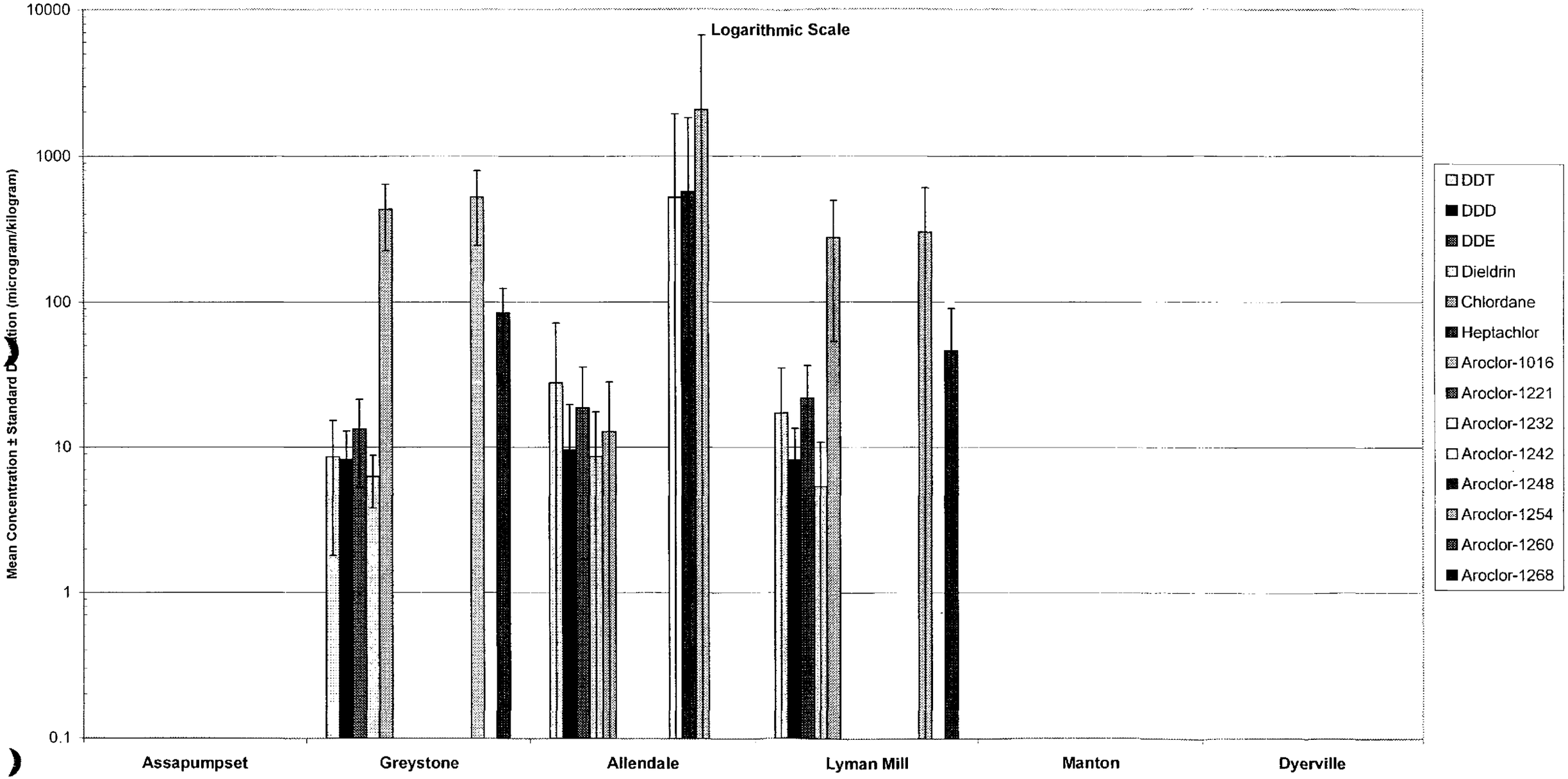
Baseline Ecological and Human Health Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



US EPA ARCHIVE DOCUMENT

Figure 29
Select Pesticide and PCB Aroclor Concentrations in Floodplain and Bank Soil

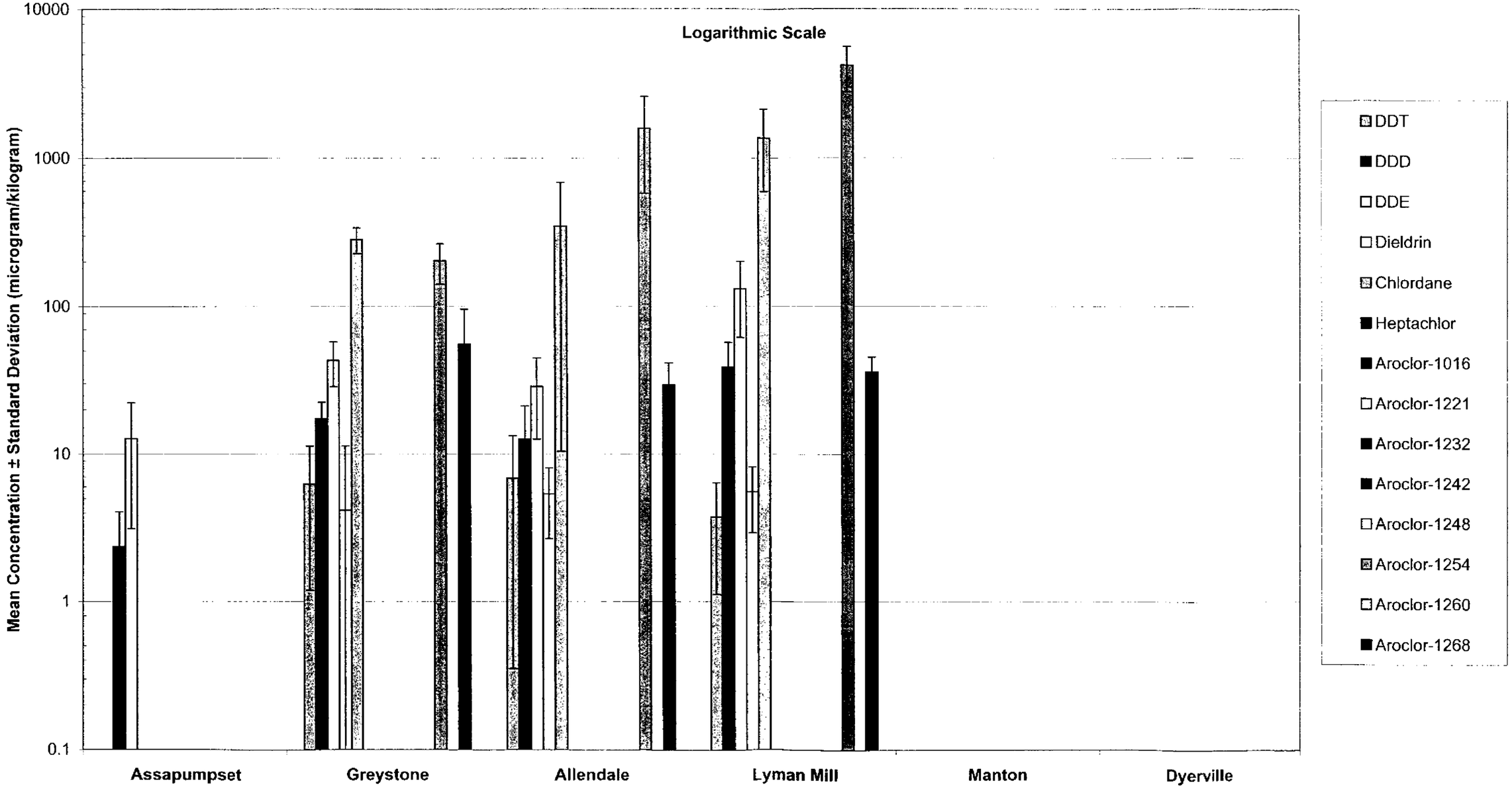
Baseline Ecological and Human Health Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



US EPA ARCHIVE DOCUMENT

Figure 30
Select Pesticide and PCB Aroclor Concentrations in White Sucker

Baseline Ecological and Human Health Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island

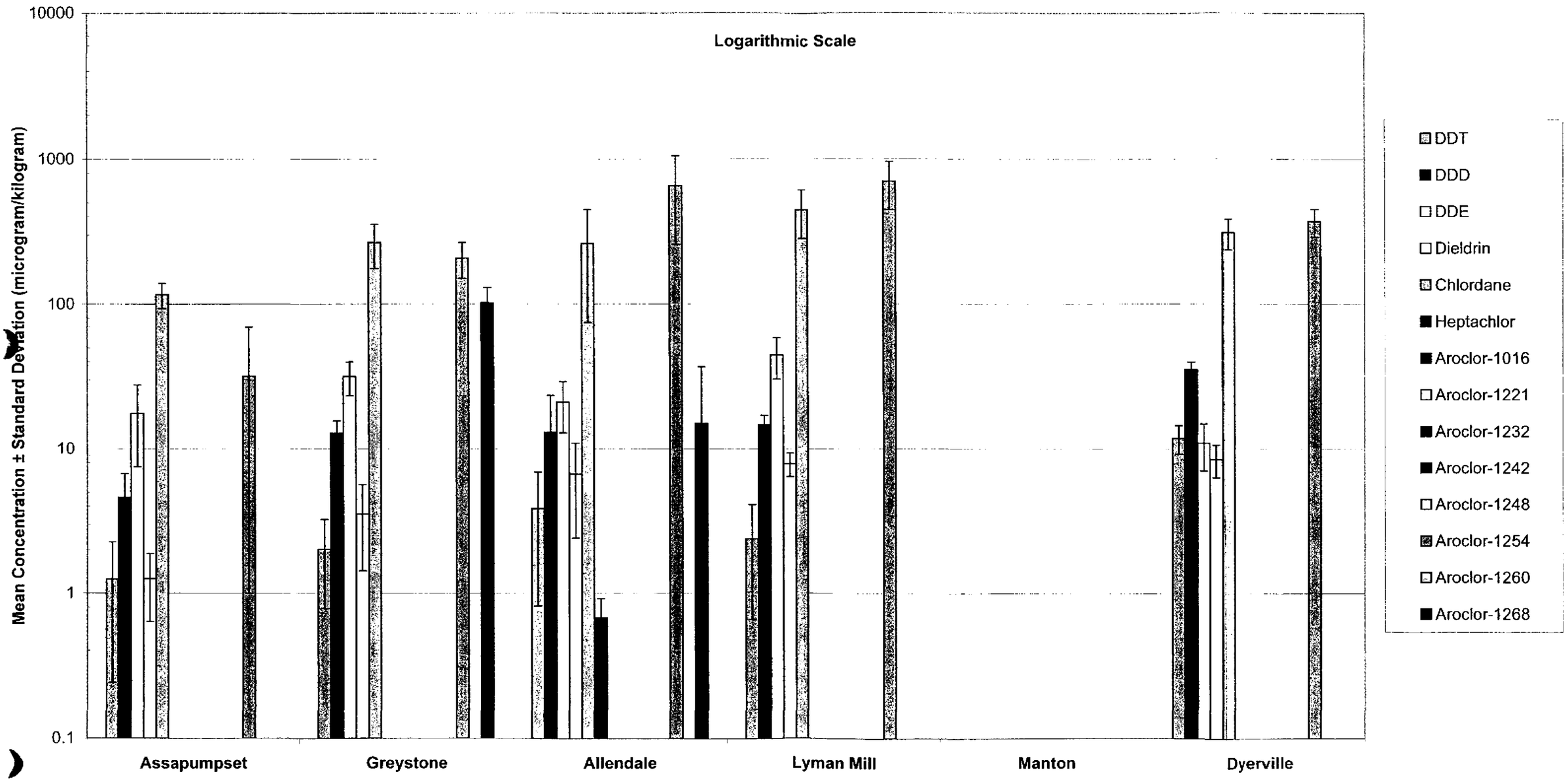


Note: Brown Bullhead used as a surrogate for White Sucker in Assapumpset Brook

US EPA ARCHIVE DOCUMENT

Figure 31
Select Pesticide and PCB Aroclor Concentrations in American Eel

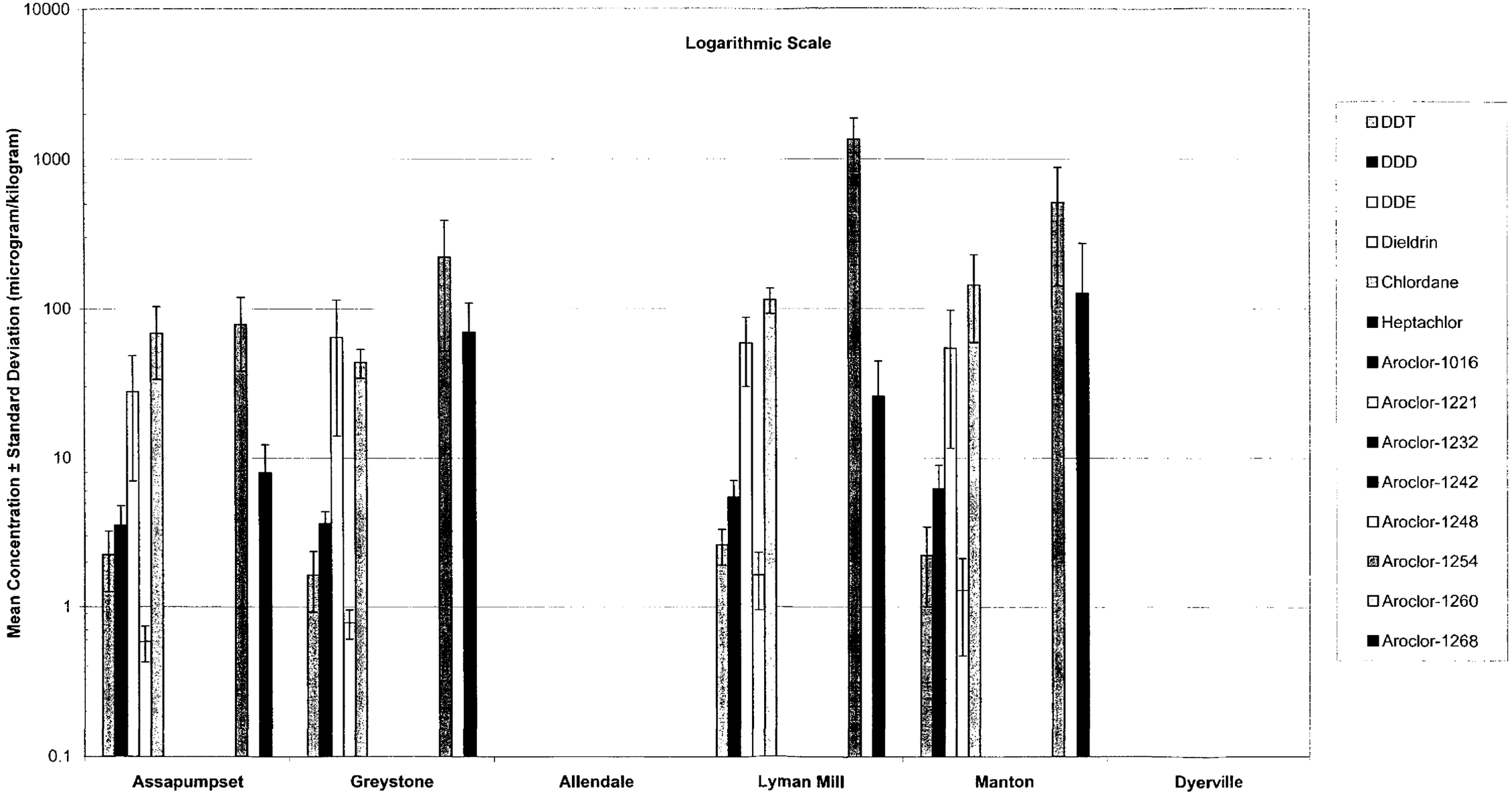
Baseline Ecological and Human Health Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



US EPA ARCHIVE DOCUMENT

Figure 32
Select Pesticide and PCB Aroclor Concentrations in Largemouth Bass

Baseline Ecological and Human Health Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island

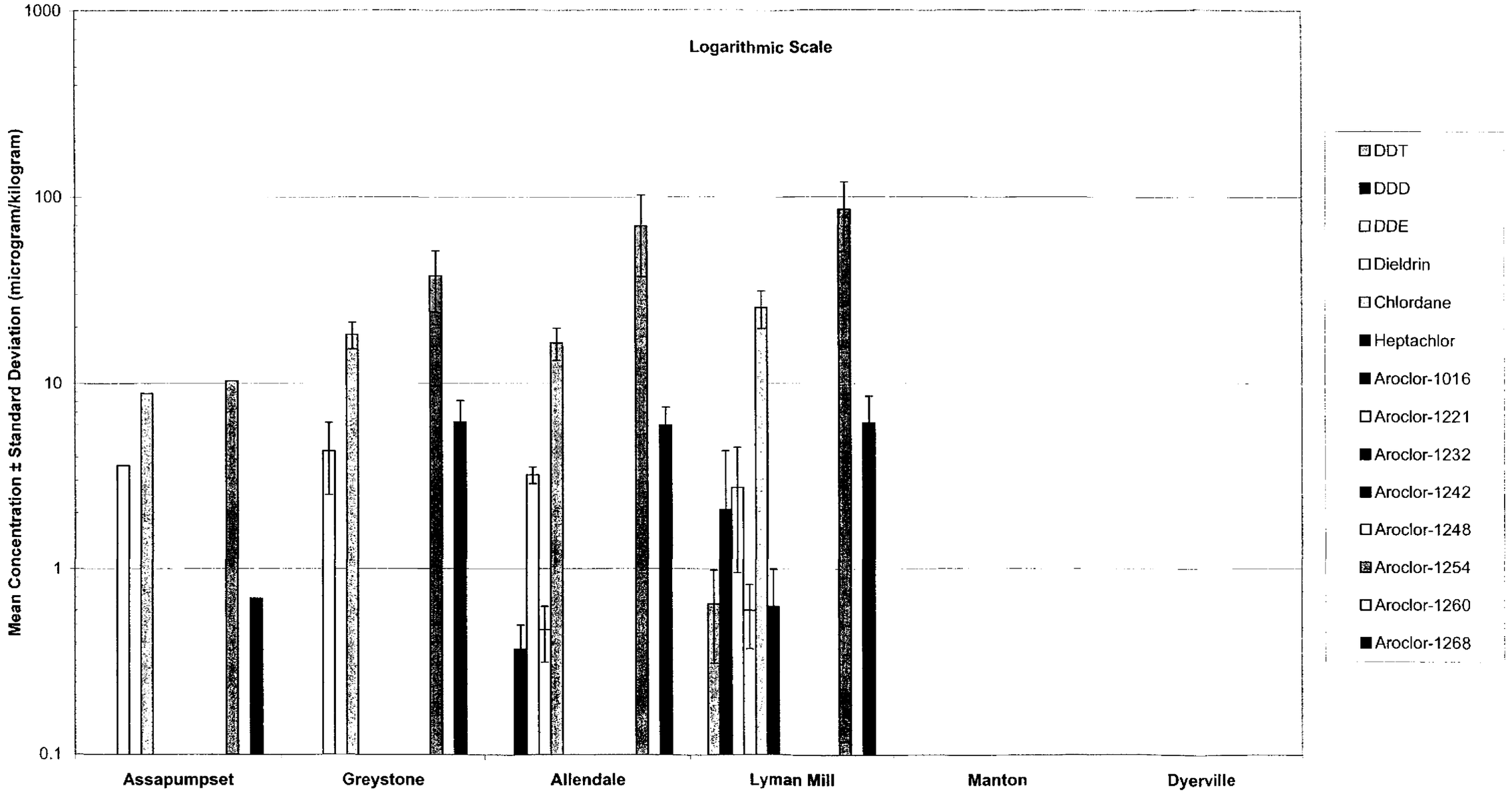


Reconstituted whole body data for Largemouth Bass used in developing histograms.

US EPA ARCHIVE DOCUMENT

Figure 33
Select Pesticide and PCB Aroclor Concentrations in Crayfish

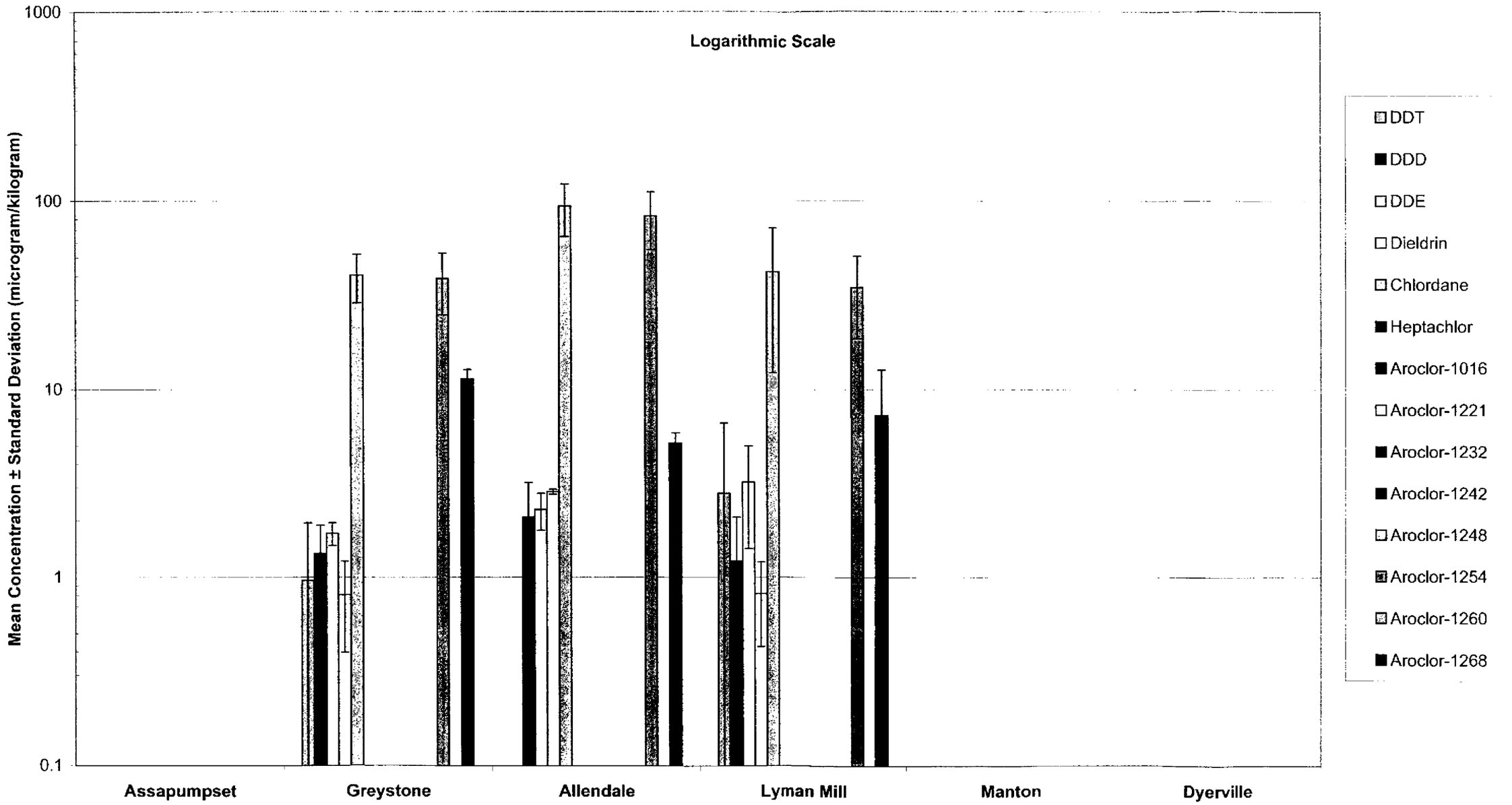
Baseline Ecological and Human Health Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



US EPA ARCHIVE DOCUMENT

Figure 34
Select Pesticide and PCB Aroclor Concentrations in Earthworms

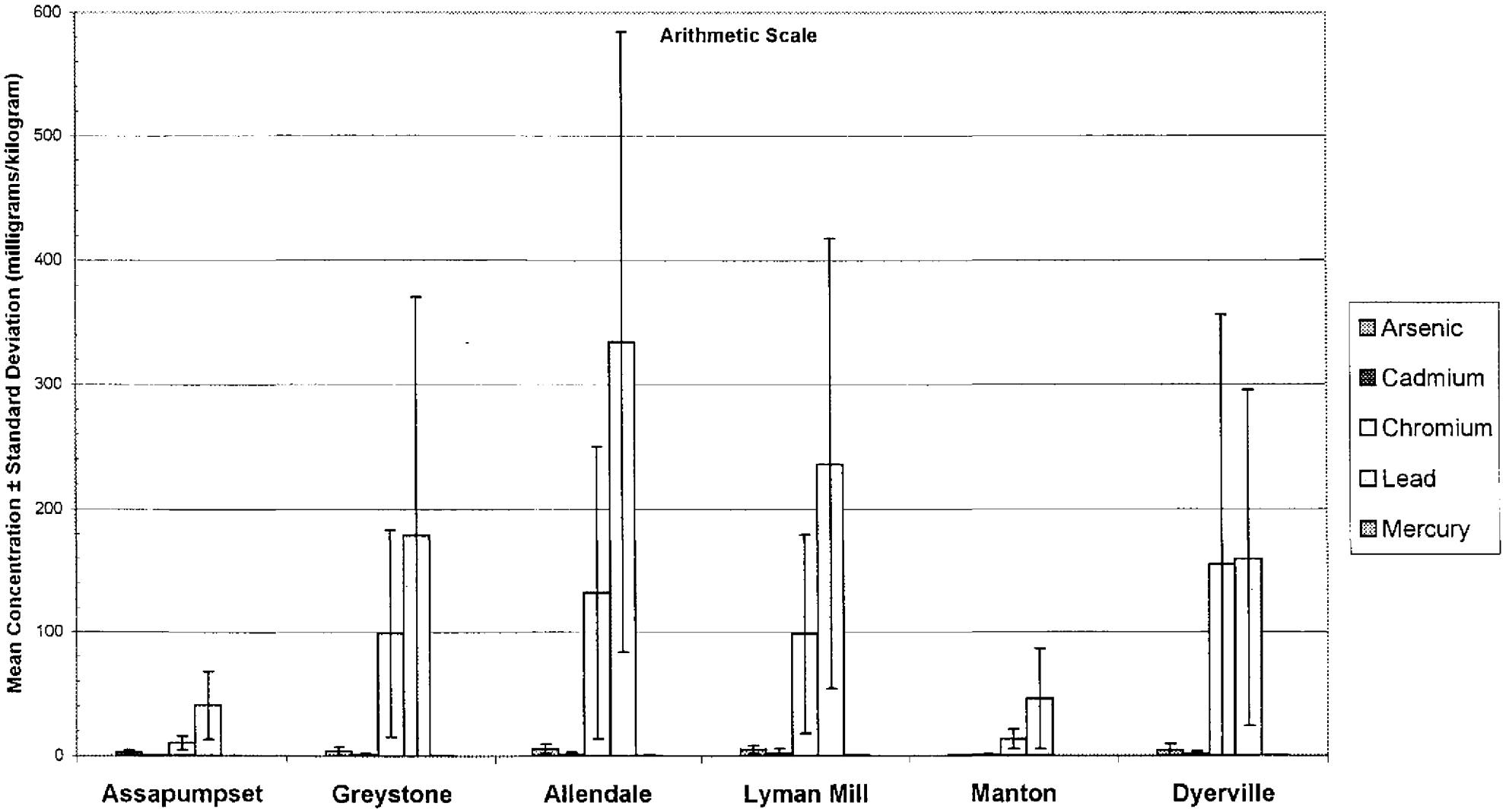
Baseline Ecological and Human Health Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



US EPA ARCHIVE DOCUMENT

Figure 35
Select Inorganic Analyte Concentrations in Sediment

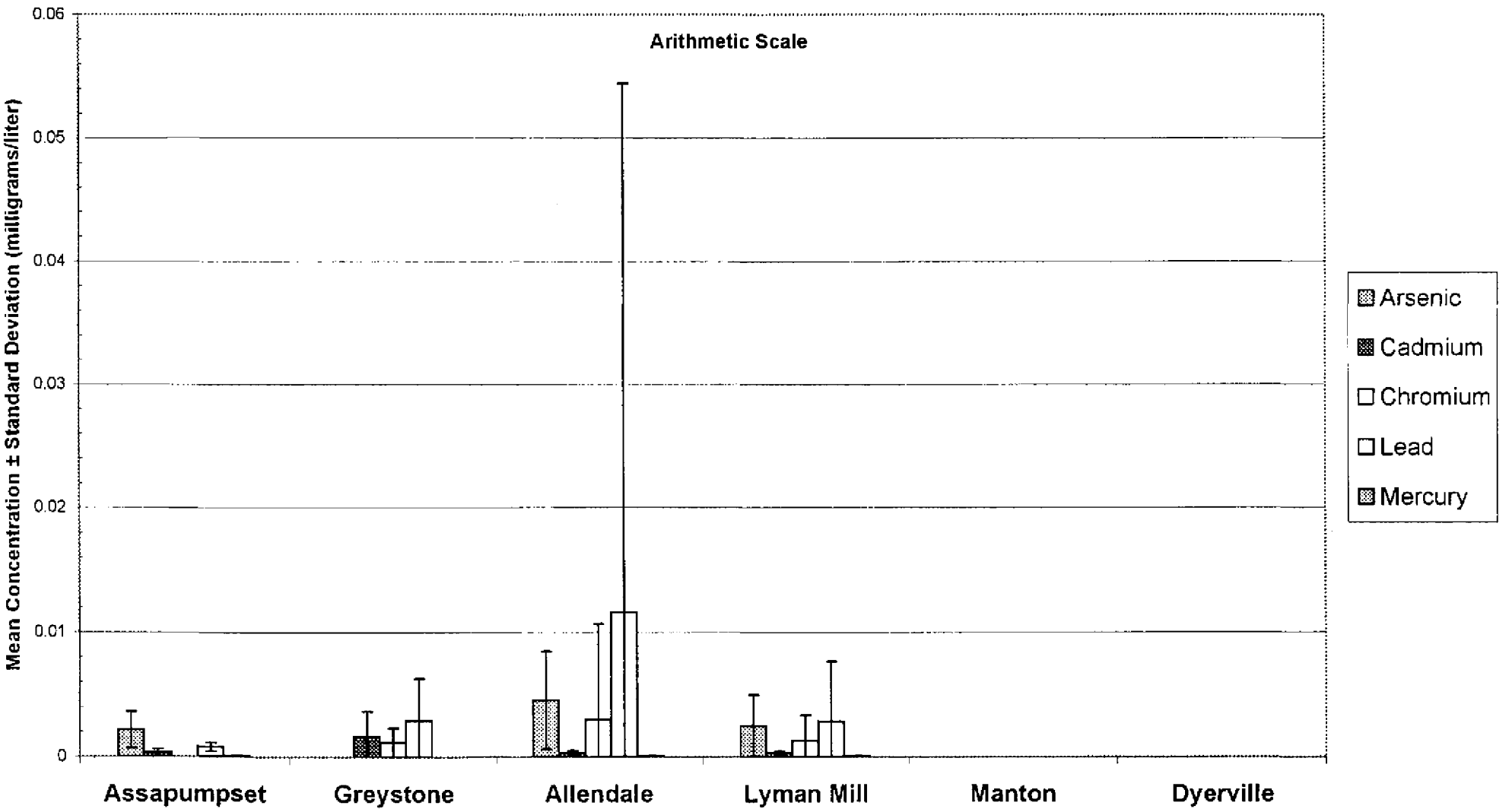
Baseline Ecological and Human Health Risk Assessments
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



US EPA ARCHIVE DOCUMENT

Figure 36
Select Inorganic Analyte Concentrations in Surface Water

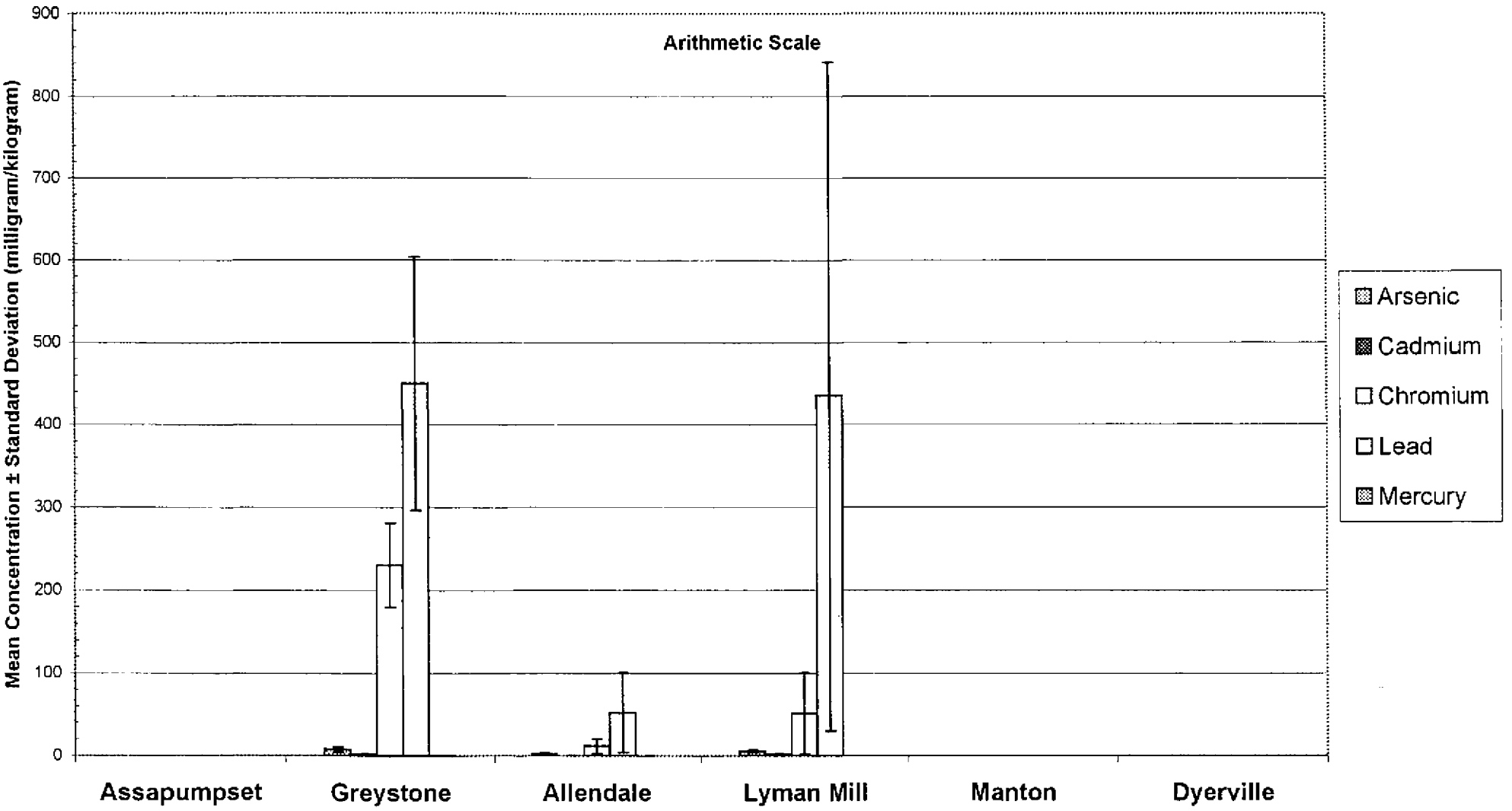
Baseline Ecological and Human Health Risk Assessments
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



US EPA ARCHIVE DOCUMENT

Figure 37
Select Inorganic Analyte Concentrations in Floodplain and Bank Soil

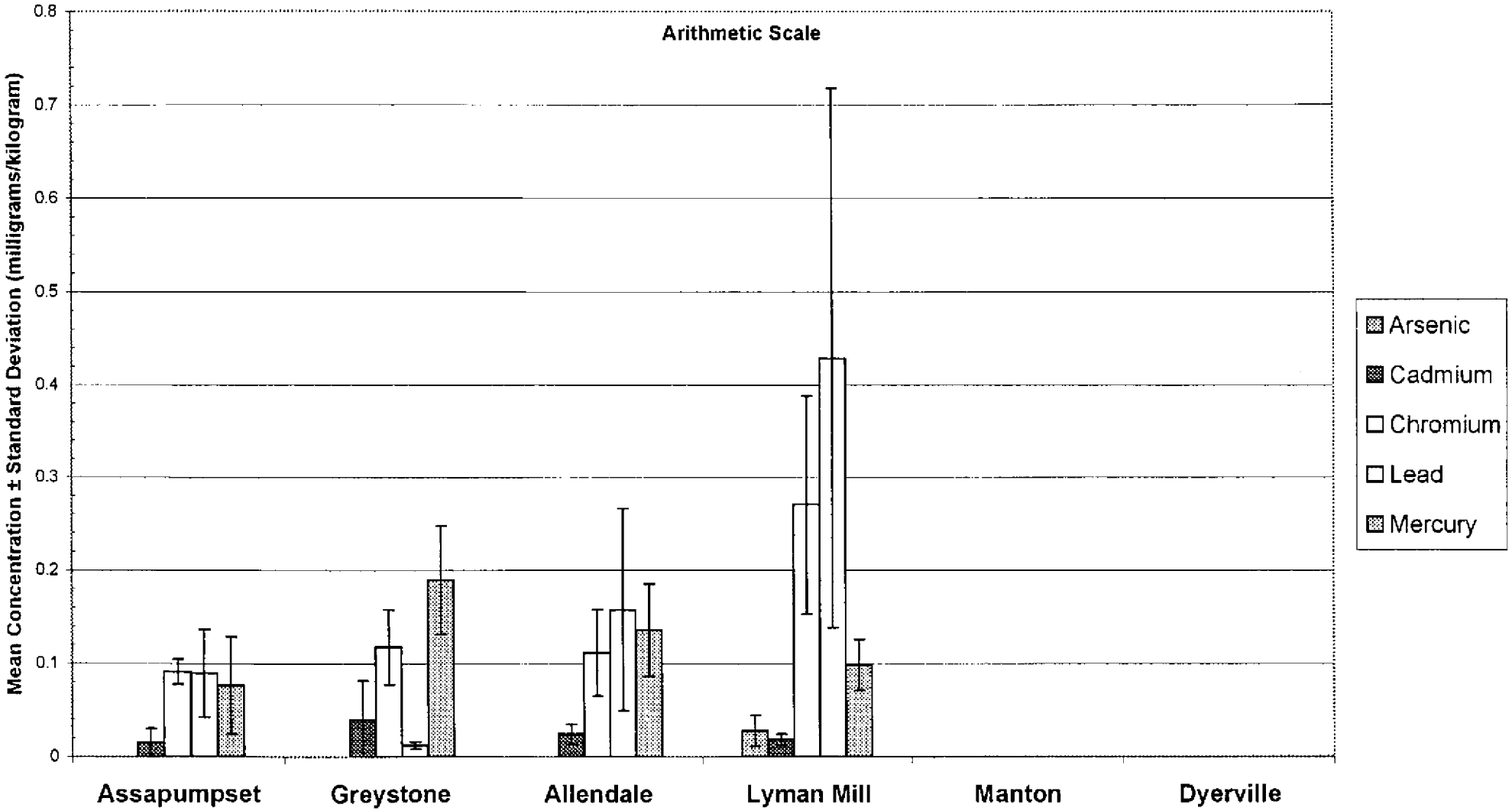
Baseline Ecological and Human Health Risk Assessments
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



US EPA ARCHIVE DOCUMENT

Figure 38
Select Inorganic Analyte Concentrations in White Sucker

Baseline Ecological and Human Health Risk Assessments
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island

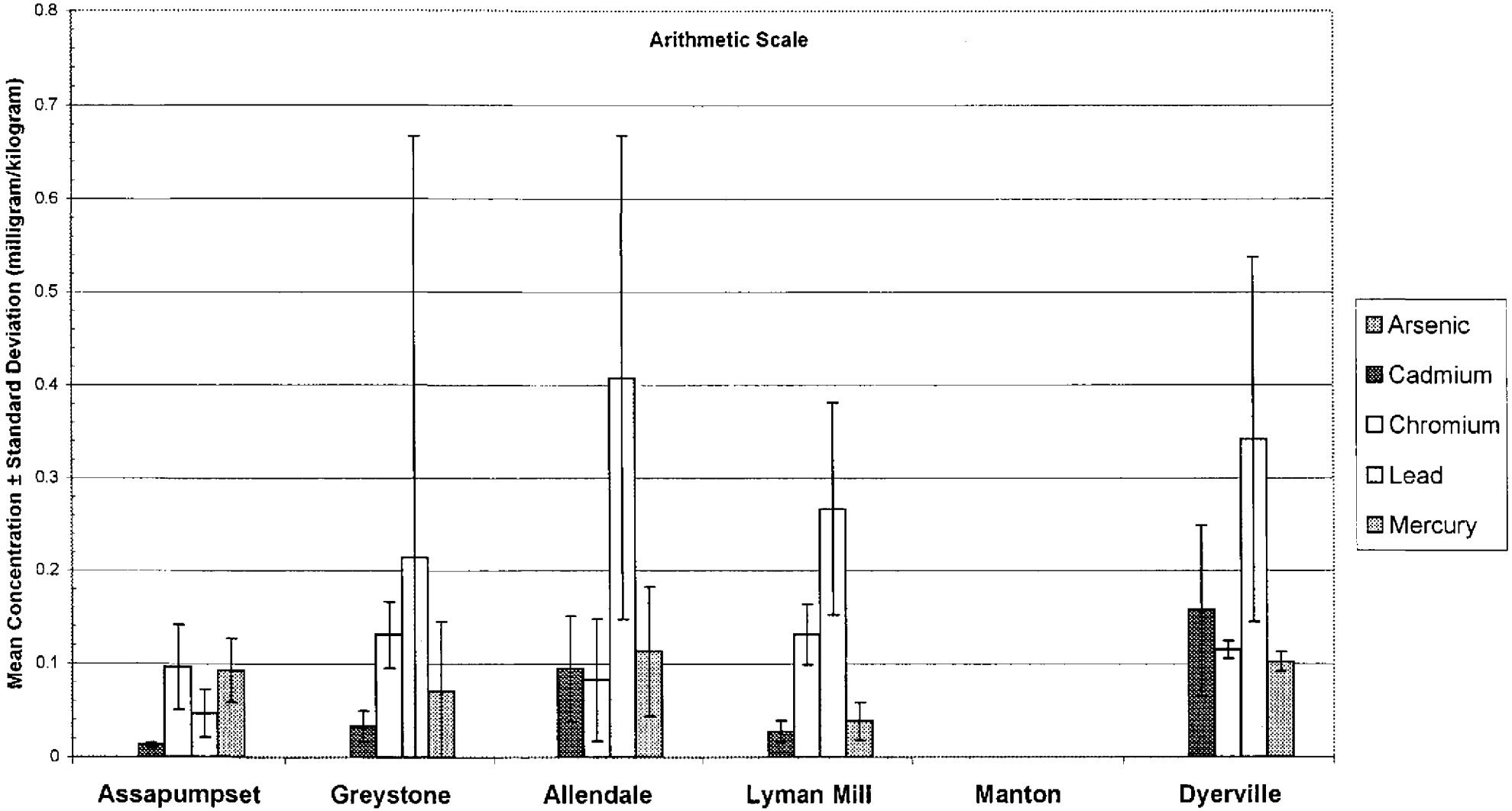


Note: Brown Bullhead used as a surrogate for White Sucker in Assapumpset Brook

US EPA ARCHIVE DOCUMENT

Figure 39
Select Inorganic Analyte Concentrations in American Eel

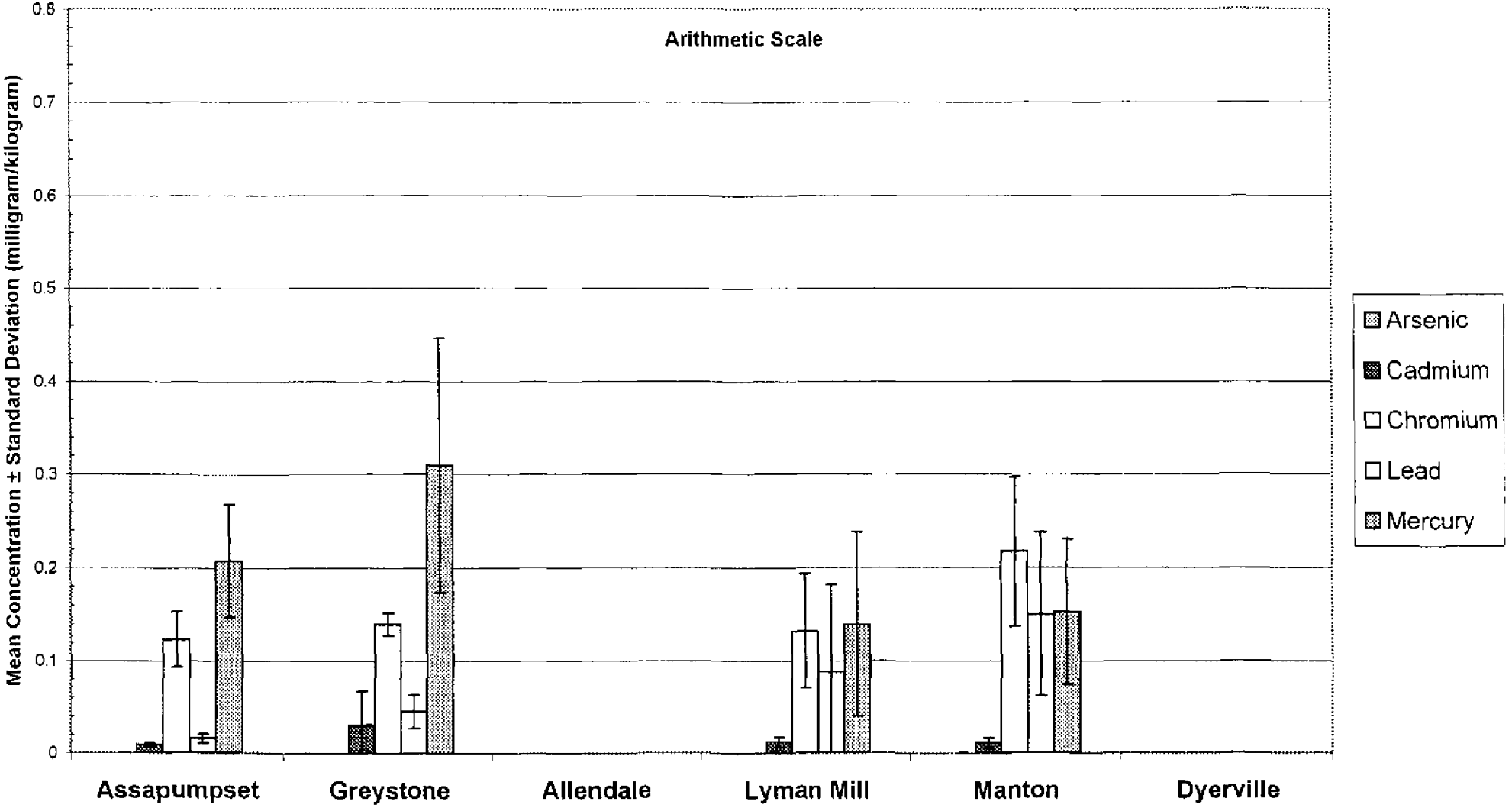
Baseline Ecological and Human Health Risk Assessments
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



US EPA ARCHIVE DOCUMENT

Figure 40
Select Inorganic Analyte Concentrations in Largemouth Bass

Baseline Ecological and Human Health Risk Assessments
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island

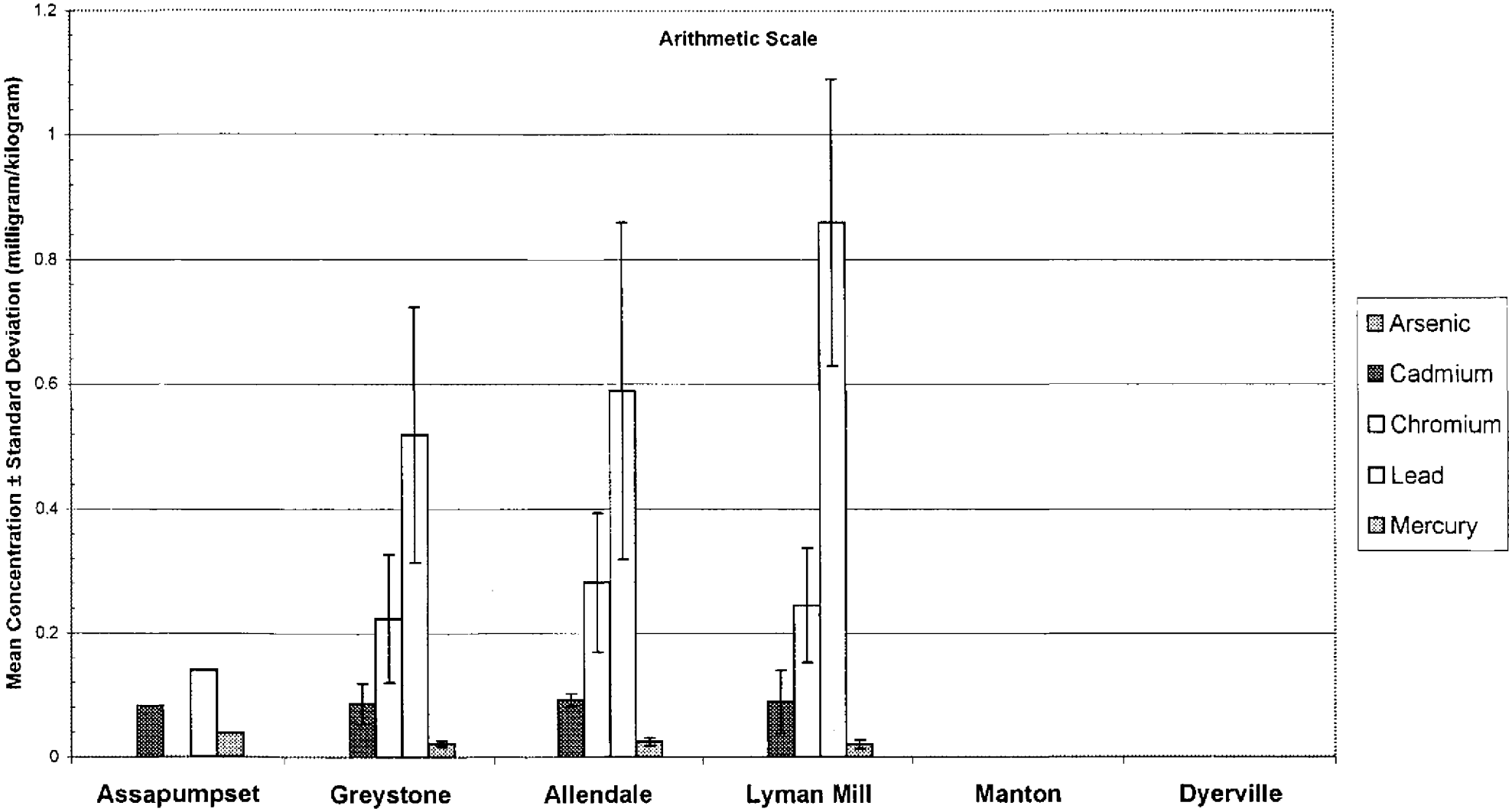


Reconstituted whole body data for Largemouth Bass used in developing histograms.

US EPA ARCHIVE DOCUMENT

Figure 41
Select Inorganics Concentrations in Crayfish

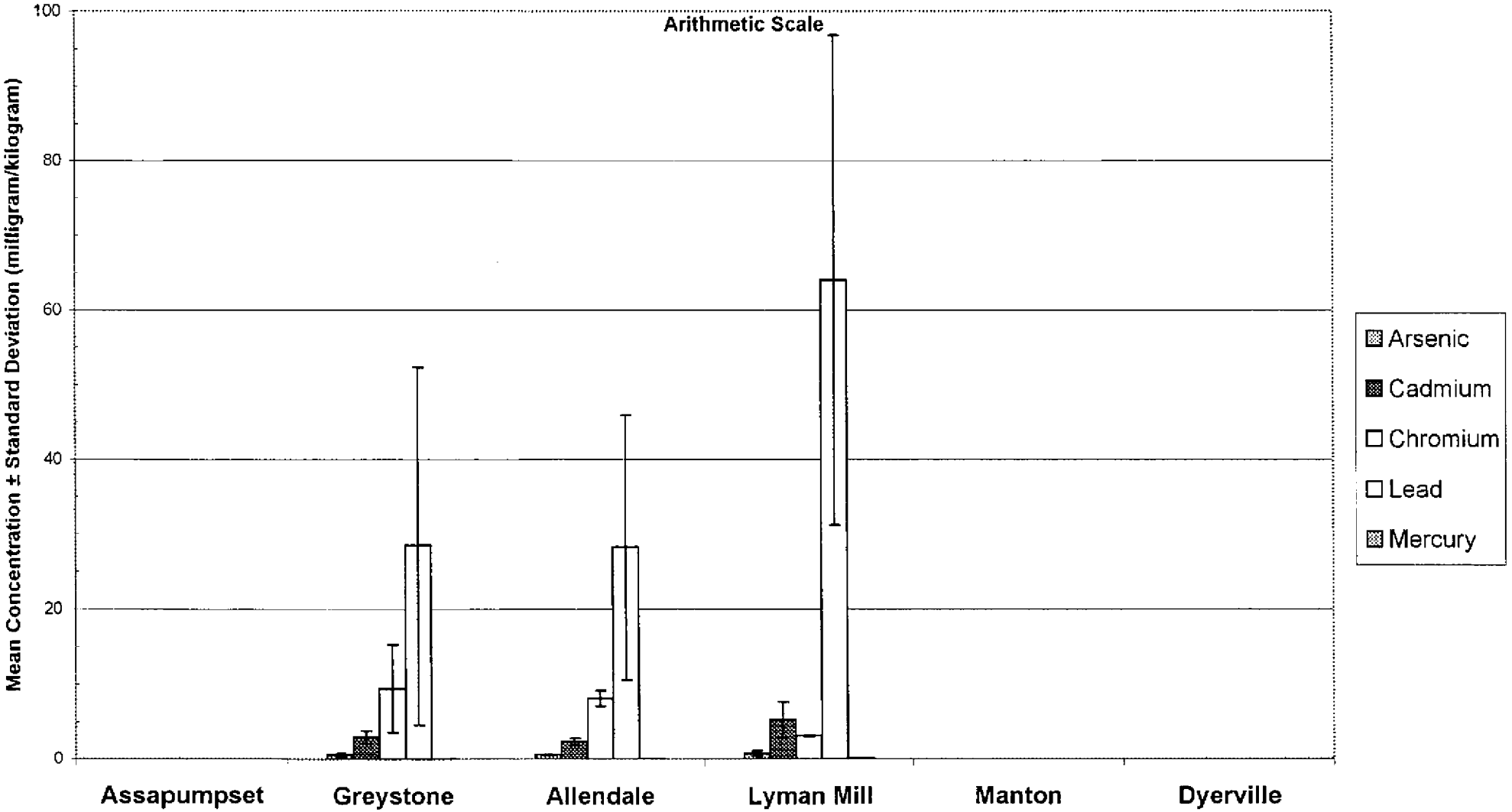
Baseline Ecological and Human Health Risk Assessments
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



US EPA ARCHIVE DOCUMENT

Figure 42
Select Inorganic Analyte Concentrations in Earthworms

Baseline Ecological and Human Health Risk Assessments
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



GREYSTONE MILL POND DAM

ALLENDALE POND

ALLENDALE DAM

ASSAPUMPSET BROOK

WOONASQUATUCKET RIVER

LYMAN MILL POND

LYMAN MILL DAM

MANTON POND

MANTON DAM

WOONASQUATUCKET RIVER

DYERVILLE POND

DYERVILLE DAM

LEGEND

- Constituent Detected
  Symbol Size Proportional to Concentration
- Constituent Not Detected
- Constituent Not Sampled

SCALE IN FEET:

0 300 600

MAGNETIC

Range of Detected Concentrations: 0.422 to 110,000 pg/g

- Concentration < 80 pg/g
- Concentration between 80 and 430 pg/g
- Concentration between 430 and 1800 pg/g
- Concentration >1800 pg/g

MACTEC

Figure 43
Distribution of Dioxin TEQ Concentrations in Sediment
Baseline Human Health Risk Assessment

Prepared By: AWS
Checked By: KJA
Approved By: MJM

P:/W9-GVT/COE-NAE/BATTELLE/CENTREDALE/TASK25-BCRA/FIGURES/SURFER/SD-TEQ.SRF

US EPA ARCHIVE DOCUMENT

MACTEC

Figure 44
Distribution of HCX Concentrations in Sediment
Baseline Human Health Risk Assessment

Prepared By: AWS
Checked By: KJA
Approved By: MJM

P:/W9-GVT/COE-NAE/BATTELLE/CENTREDALE/TASK25-BCRA/FIGURES/SURFER/SD-HCX.SRF

US EPA ARCHIVE DOCUMENT

LEGEND

Constituent Detected
Symbol Size Proportional to Concentration

Constituent Not Detected

Constituent Not Sampled

SCALE IN FEET:

0 300 600

MAGNETIC

Range of Detected Concentrations: 11 to 27,000 ug/Kg

GREYSTONE MILL POND DAM

ALLENDALE POND

ALLENDALE DAM

ASSAPUMPSET BROOK

WOONASQUATUCKET RIVER

LYMAN MILL POND

LYMAN MILL DAM

MANTON POND

MANTON DAM

WOONASQUATUCKET RIVER

DYERVILLE POND

DYERVILLE DAM

MACTEC

Figure 45
Distribution of Total PCB Concentrations in Sediment
Baseline Human Health Risk Assessment

Prepared By: AWS
Checked By: KJA
Approved By: MJM

P:/W9-GVT/COE-NAE/BATTELLE/CENTREDALE/TASK25-BCRA/FIGURES/SURFER/SD-PCB.SRF

US EPA ARCHIVE DOCUMENT

GREYSTONE MILL POND DAM

ALLENDALE POND

ALLENDALE DAM

ASSAPUMPSET BROOK

WOONASQUATUCKET RIVER

LYMAN MILL POND

LYMAN MILL DAM

MANTON POND

MANTON DAM

WOONASQUATUCKET RIVER

DYERVILLE POND

DYERVILLE DAM

LEGEND

- Constituent Detected (Relative Concentration)
- Constituent Not Detected
- Constituent Not Sampled

SCALE IN FEET:

0 300 600

MAGNETIC

Range of Detected Concentrations: 49 to 113,110 ug/Kg

US EPA ARCHIVE DOCUMENT

MACTEC

Figure 46
Distribution of Total PAH Concentrations in Sediment
Baseline Human Health Risk Assessment

Prepared By: AWS
Checked By: KJA
Approved By: MJM

P:/W9-GVT/COE-NAE/BATTELLE/CENTREDALE/TASK25-BCRA/FIGURES/SURFER/SD-PAH.SRF

GREYSTONE MILL POND DAM

ALLENDALE POND

ALLENDALE DAM

ASSAPUMPSET BROOK

WOONASQUATUCKET RIVER

LYMAN MILL POND

LYMAN MILL DAM

MANTON POND

MANTON DAM

WOONASQUATUCKET RIVER

DYERVILLE POND

DYERVILLE DAM

LEGEND

- Constituent Detected
  Symbol Size Proportional to Concentration
- Constituent Not Detected
- Constituent Not Sampled

SCALE IN FEET:

0 300 600

MAGNETIC

Range of Detected Concentrations: 0.31 to 2,305.88 ug/Kg

US EPA ARCHIVE DOCUMENT

MACTEC

Figure 47
Distribution of Total Pesticide Concentrations in Sediment
Baseline Human Health Risk Assessment

Prepared By: AWS
Checked By: KJA
Approved By: MJM

P:/W9-GVT/COE-NAE/BATTELLE/CENTREDALE/TASK25-BCRA/FIGURES/SURFER/SD-PEST.SRF

GREYSTONE MILL POND DAM

ALLENDALE POND

ALLENDALE DAM

ASSAPUMPSET BROOK

WOONASQUATUCKET RIVER

LYMAN MILL POND

LYMAN MILL DAM

MANTON POND

MANTON DAM

DYERVILLE POND

DYERVILLE DAM

LEGEND

Constituent Detected
Symbol Size Proportional to Concentration

Constituent Not Detected

Constituent Not Sampled

SCALE IN FEET:

0 300 600

MAGNETIC

Range of Detected Concentrations: 0.938 to 333 Metals Toxic Units

US EPA ARCHIVE DOCUMENT

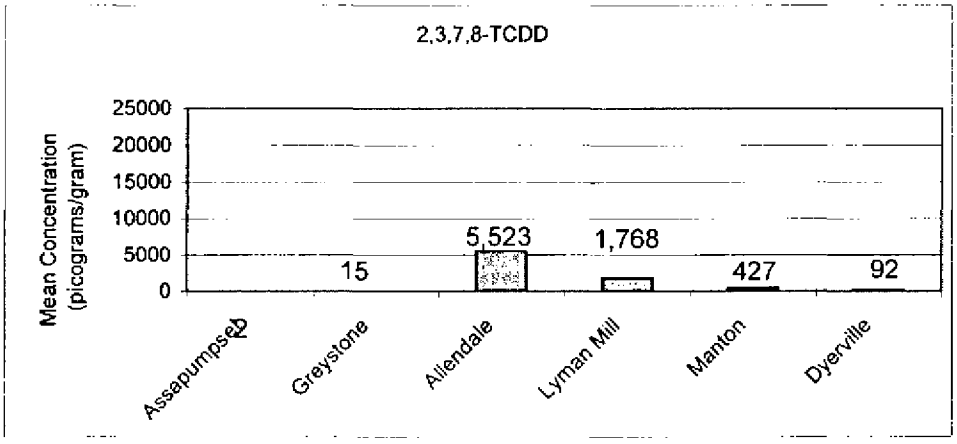
MACTEC

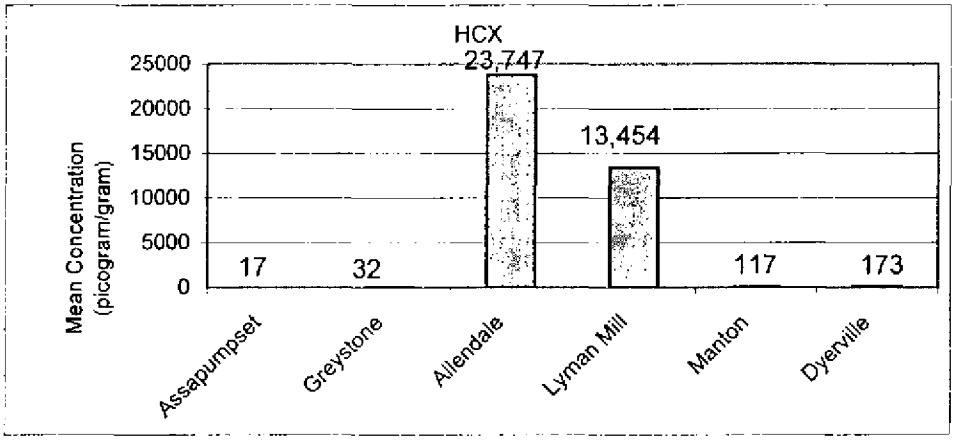
Figure 48
Distribution of Metals (Toxic Units) Concentrations in Sediment
Baseline Human Health Risk Assessment

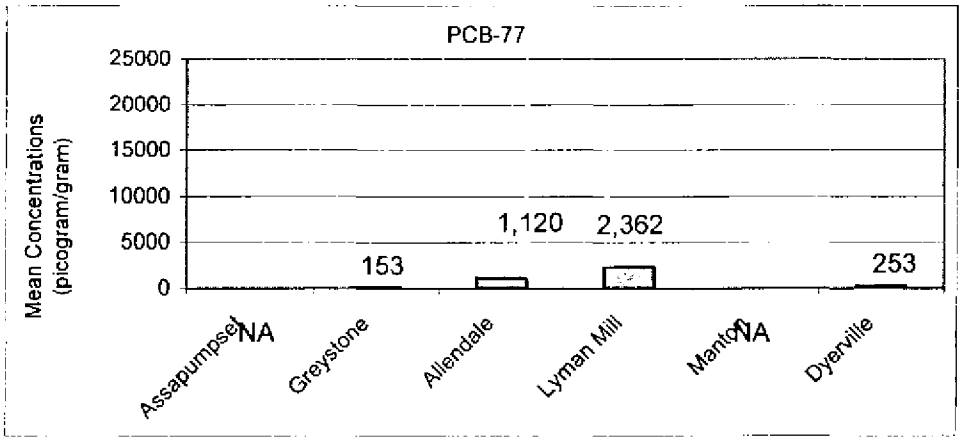
Prepared By: AWS
Checked By: KJA
Approved By: MJM

P:/W9-GVT/COE-NAE/BATTELLE/CENTREDALE/TASK25-BCRA/FIGURES/SURFER/SD-MTU.SRF

Figure 49
2,3,7,8-TCDD, HCX, and PCB-77 Concentrations in Sediment
Baseline Human Health Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island





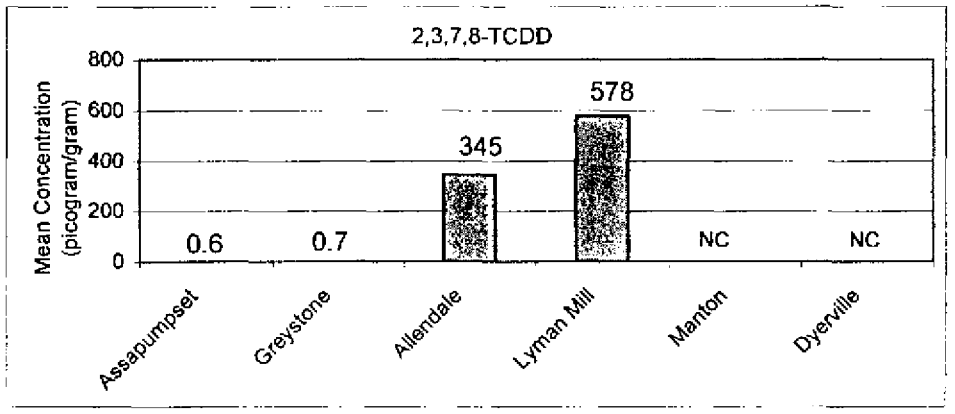


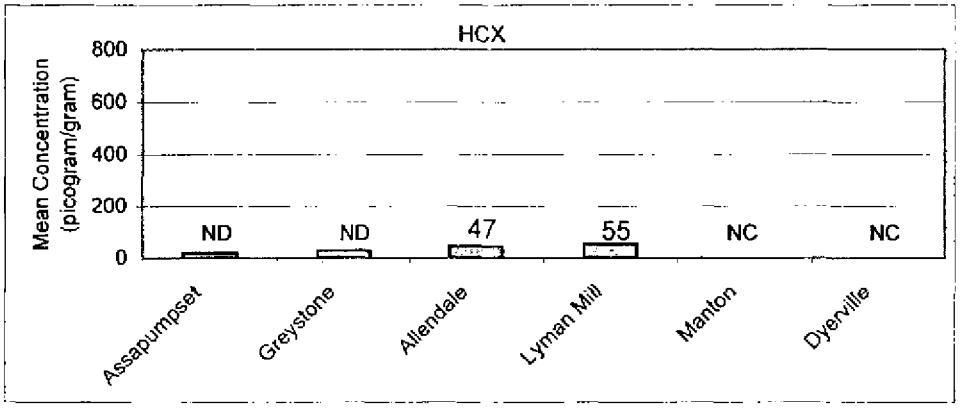
NA = Not Analyzed

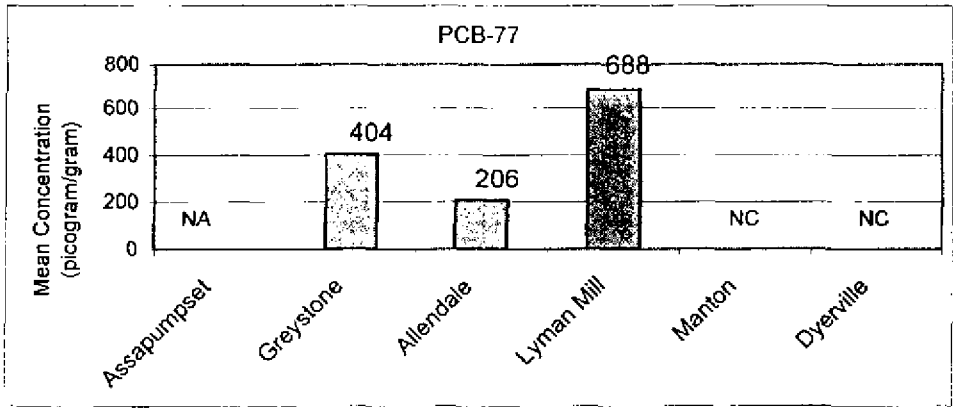
Created By: LAS
Checked By: MJM

US EPA ARCHIVE DOCUMENT

Figure 50
2,3,7,8-TCDD, HCX, and PCB-77 Concentrations in White Sucker
Baseline Human Health Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island







NC = Not Collected
ND = Not Detected
NA = Not Analyzed
White Sucker was not collected from Assapumpset Pond.
Data for Assapumpset Pond represents Brown Bullhead.

Created By: LAS
Checked By: MJM

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Figure 51
Summary of RME Human Health Noncancer Risks - Resident Living Along the Woonasquatucket River

Baseline Human Health Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island

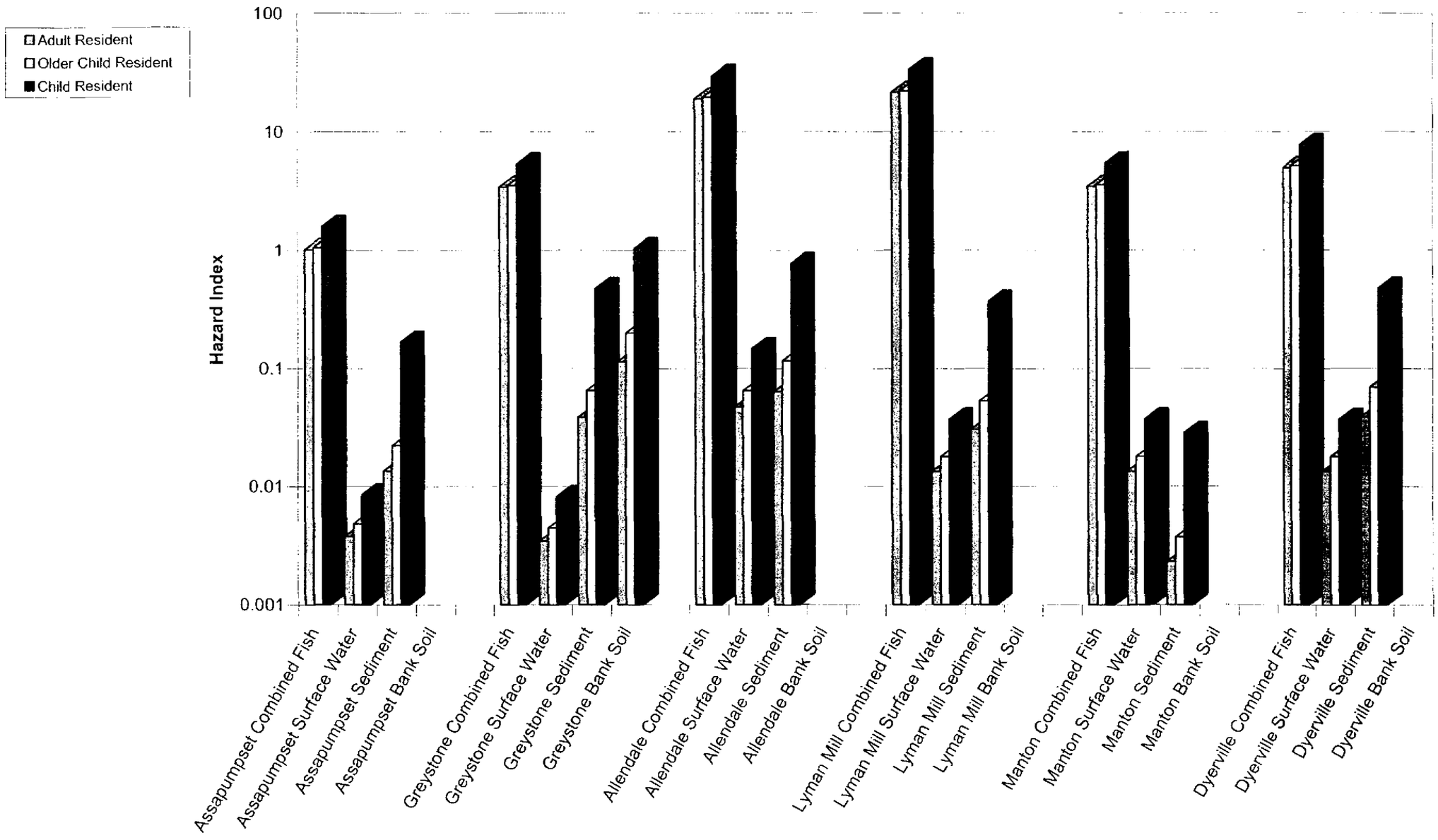

Figure 52
Summary of RME Human Health Cancer Risks - Resident Living Along the Woonasquatucket River

Baseline Human Health Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island

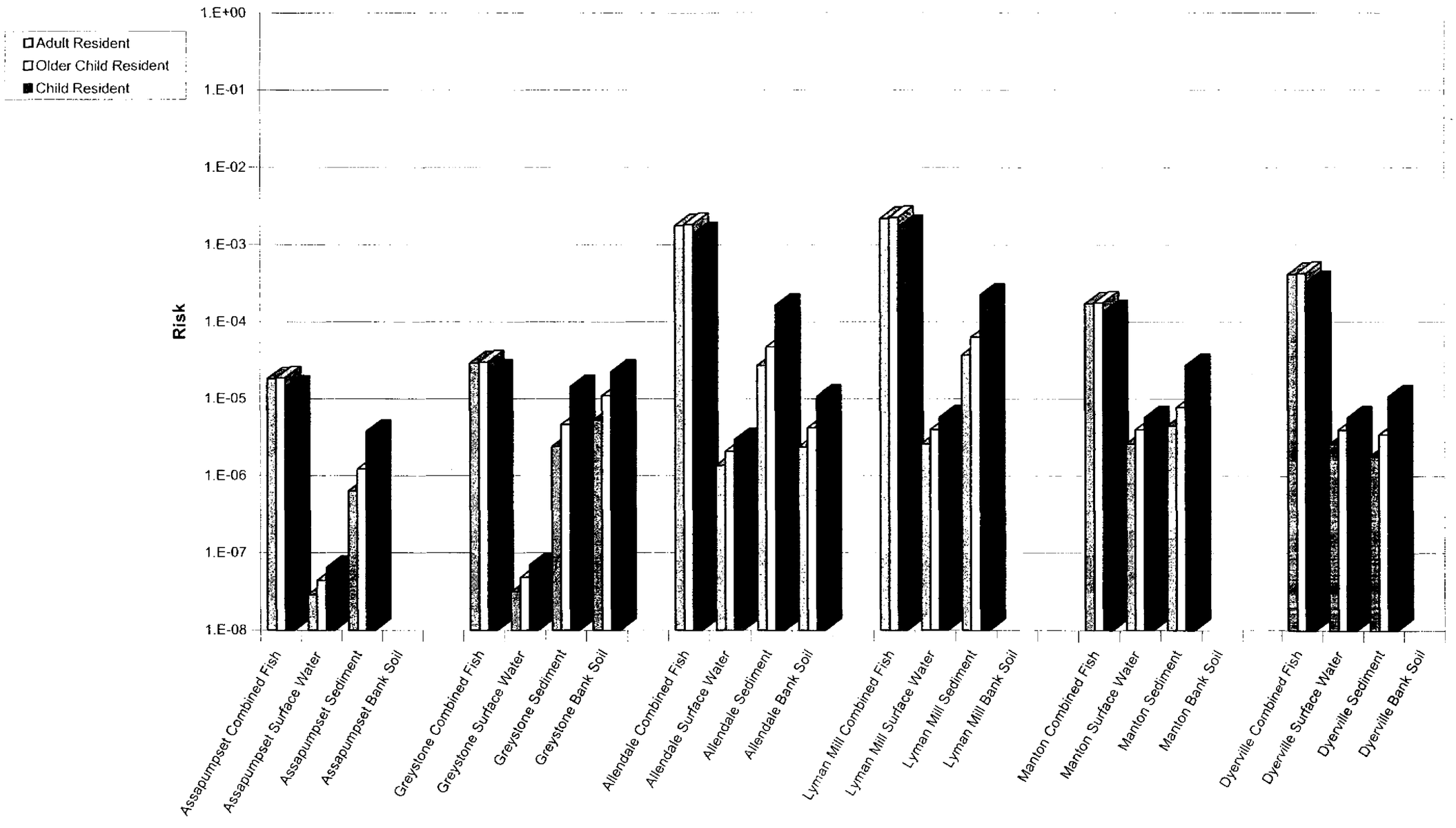

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Figure 53
Summary of RME Human Health Noncancer Risks - Visiting Recreational Angler

Baseline Human Health Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island

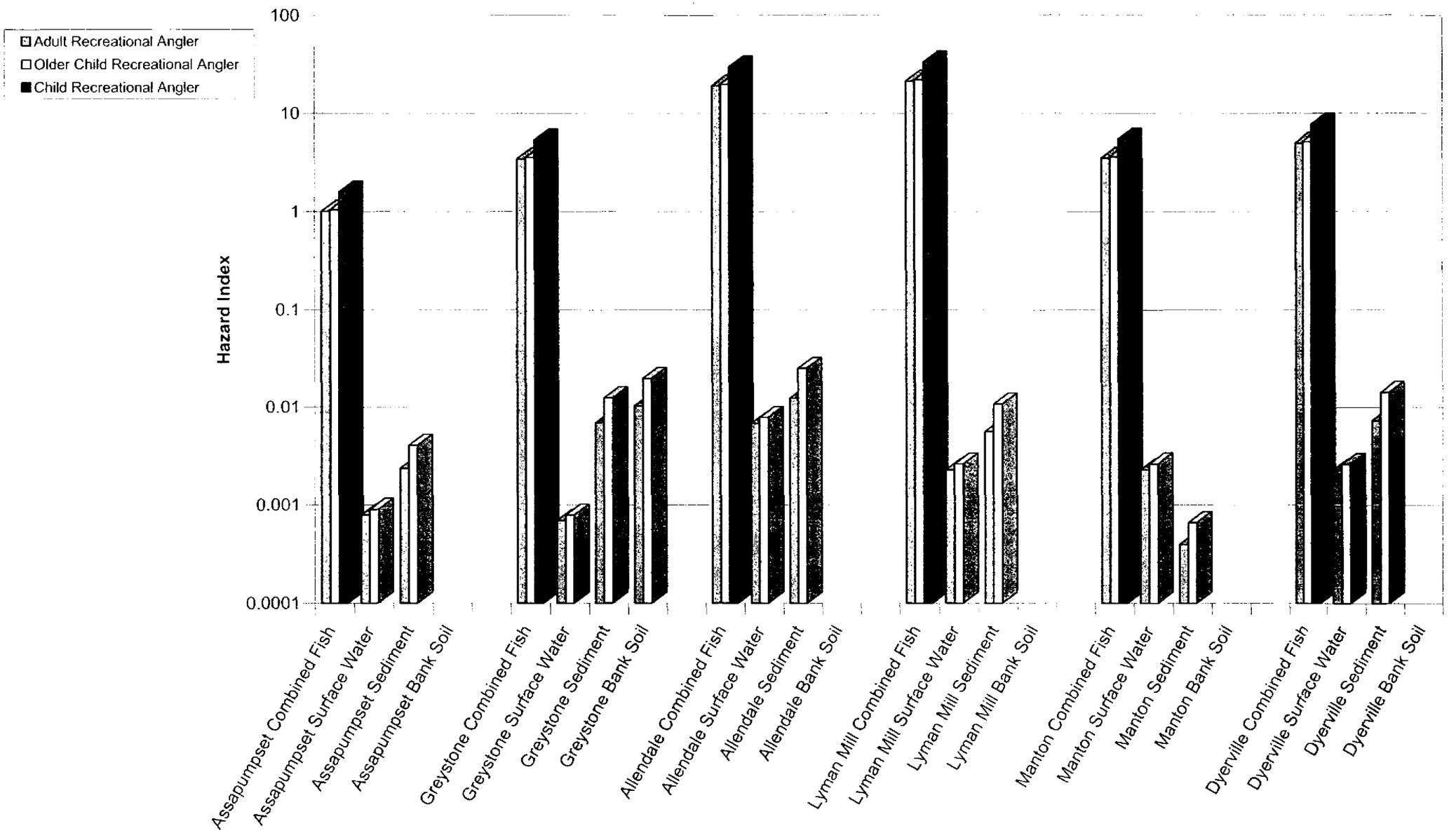

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Figure 54
Summary of RME Human Health Cancer Risks - Visiting Recreational Angler

Baseline Human Health Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island

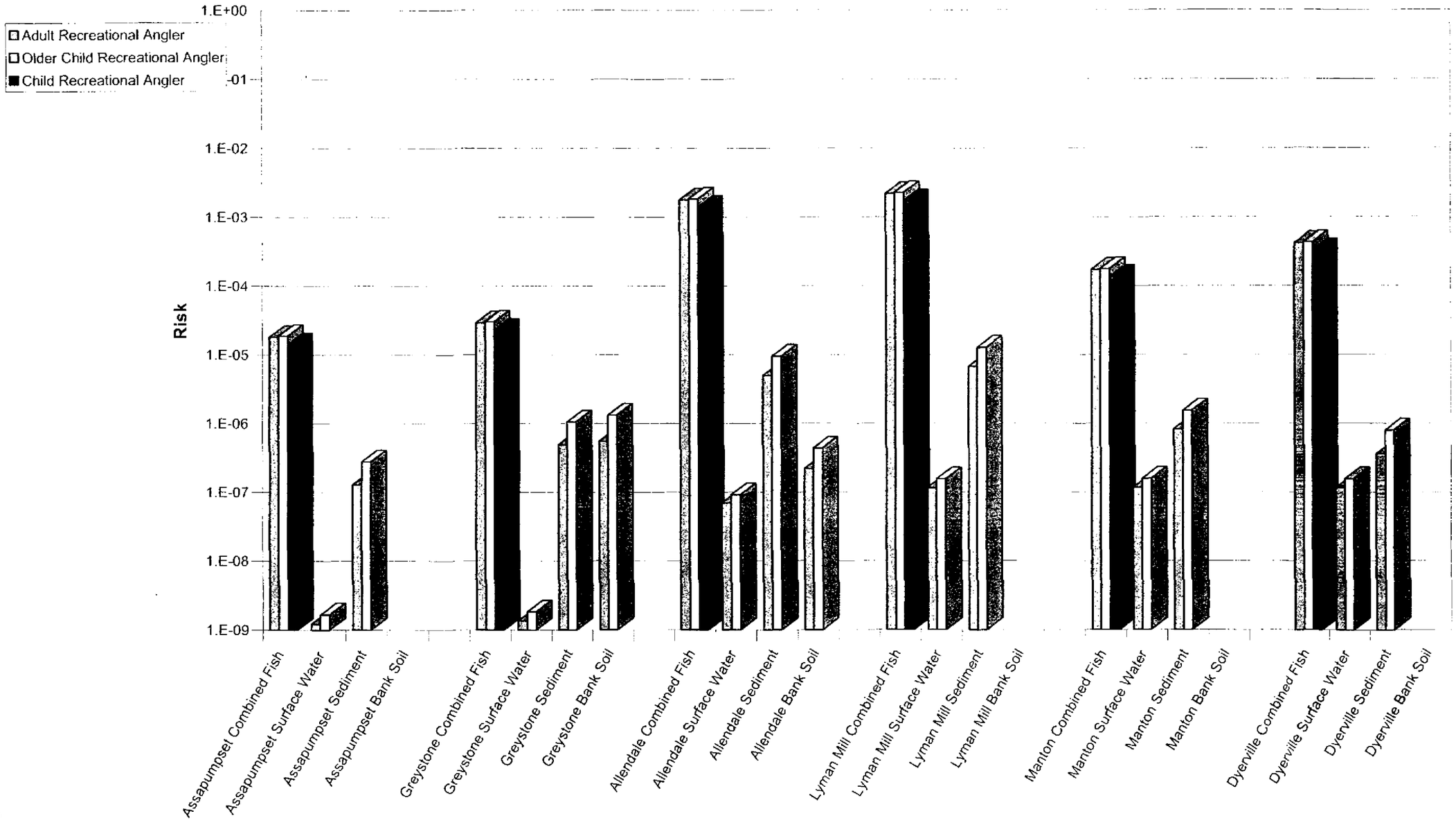

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Figure 55
Summary of CT Human Health Noncancer Risks - Resident Living Along the Woonasquatucket River

Baseline Human Health Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island

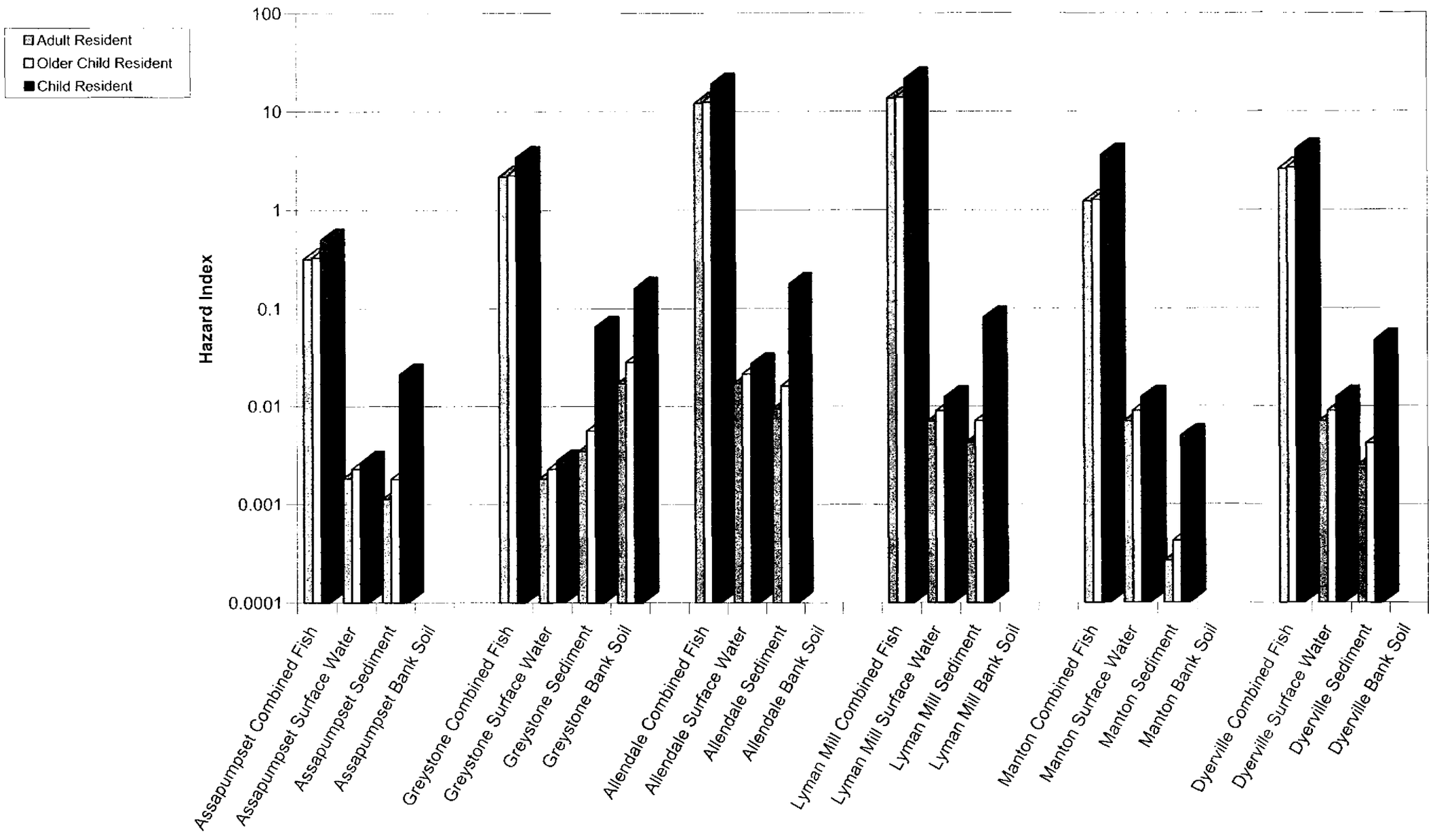

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Figure 55
Summary of CT Human Health Noncancer Risks - Resident Living Along the Woonasquatucket River

Baseline Human Health Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island

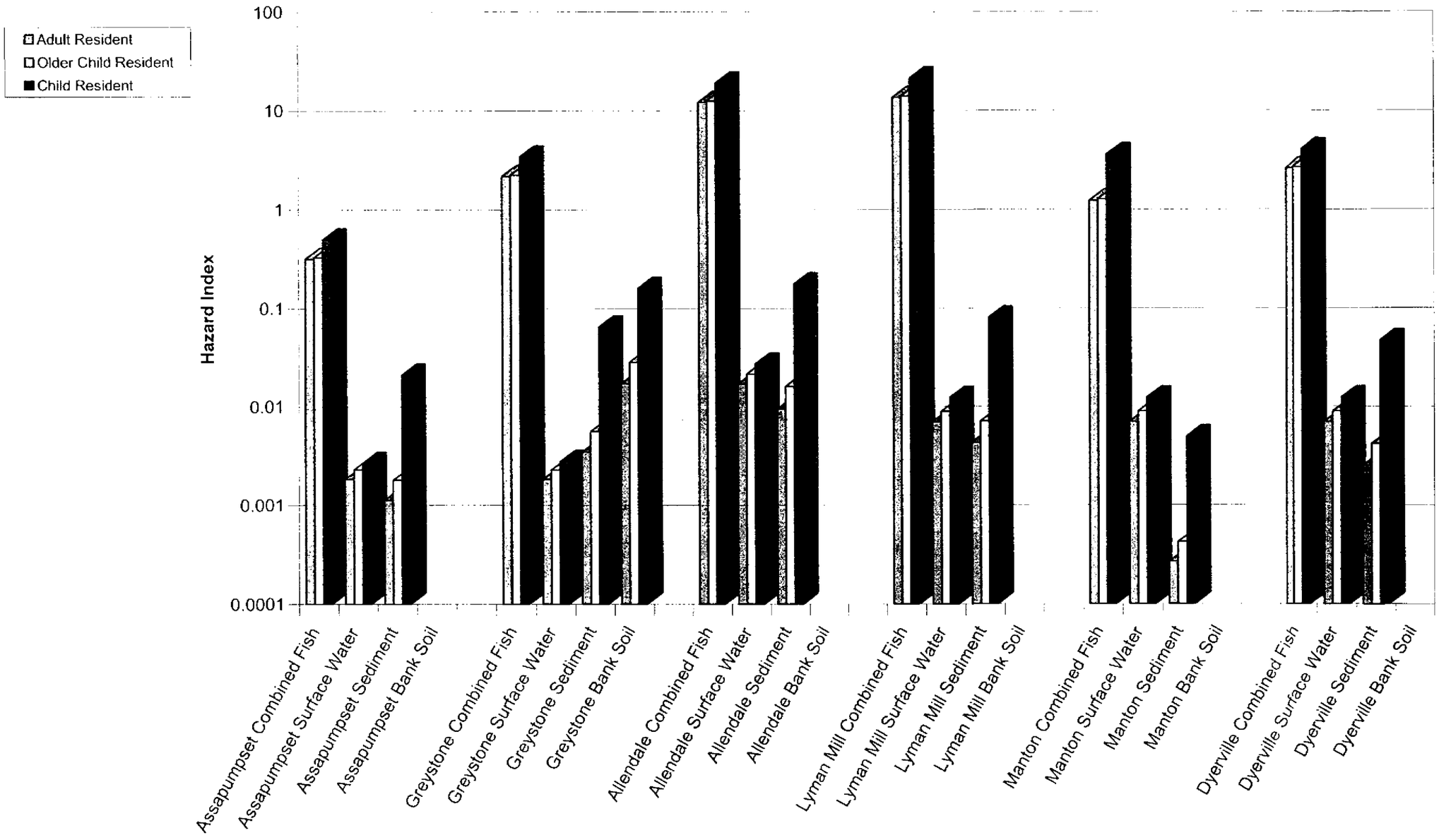

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Figure 56
Summary of CT Human Health Cancer Risks - Resident Living Along the Woonasquatucket River

Baseline Human Health Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island

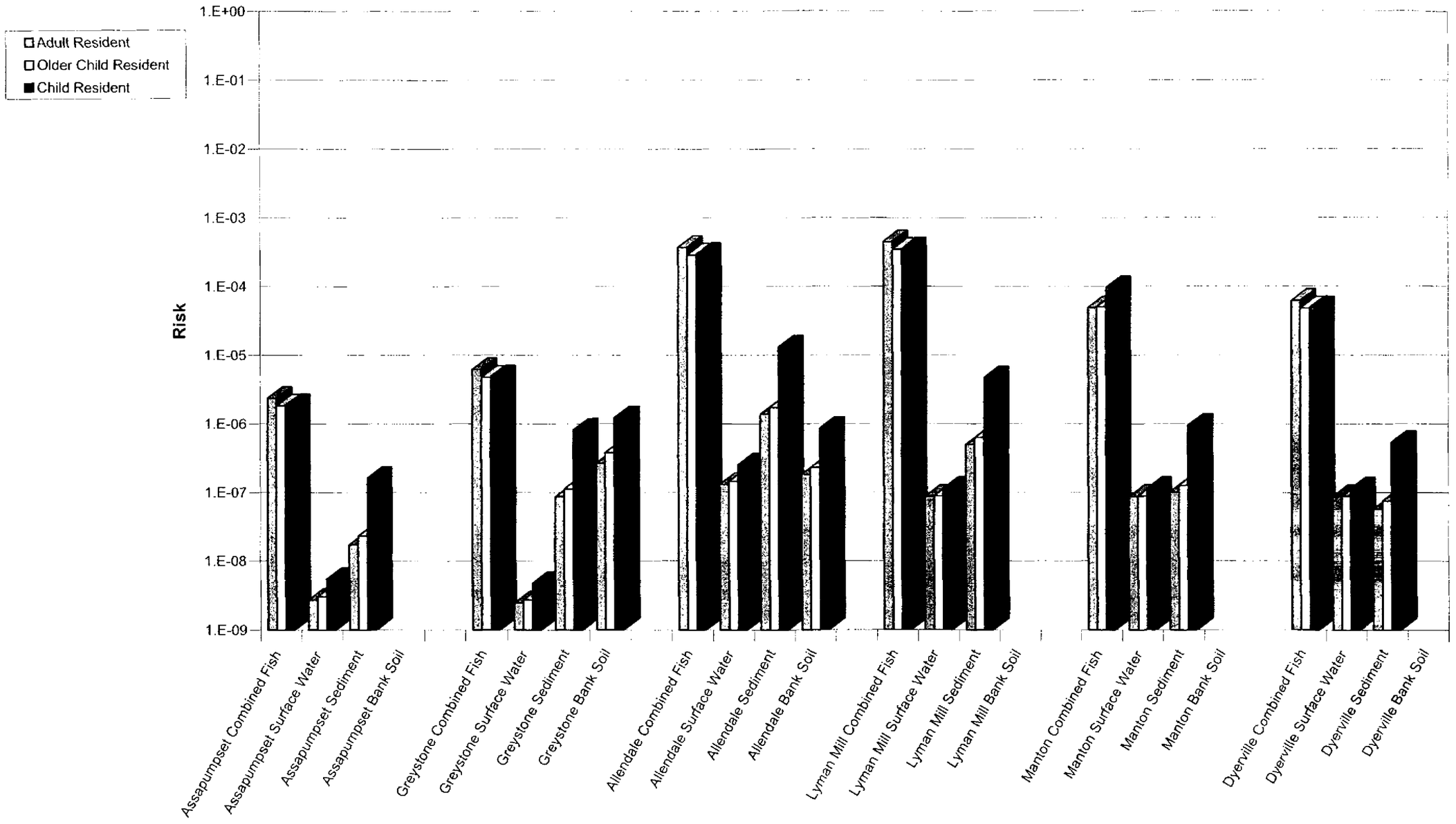

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Figure 57
Summary of CT Human Health Noncancer Risks - Visiting Recreational Angler

Baseline Human Health Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island

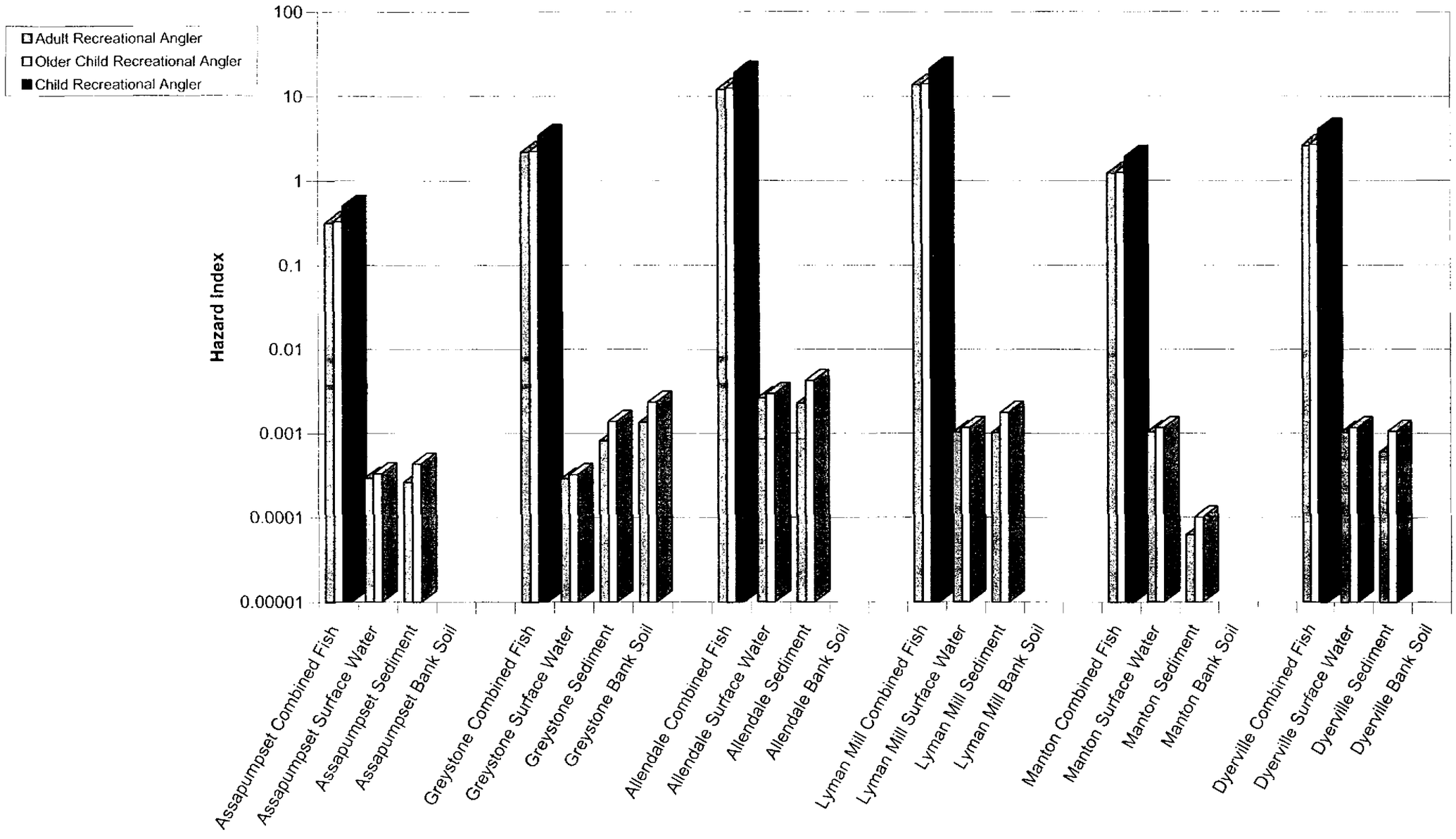

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Figure 58
Summary of CT Human Health Cancer Risks - Visiting Recreational Angler

Baseline Human Health Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island

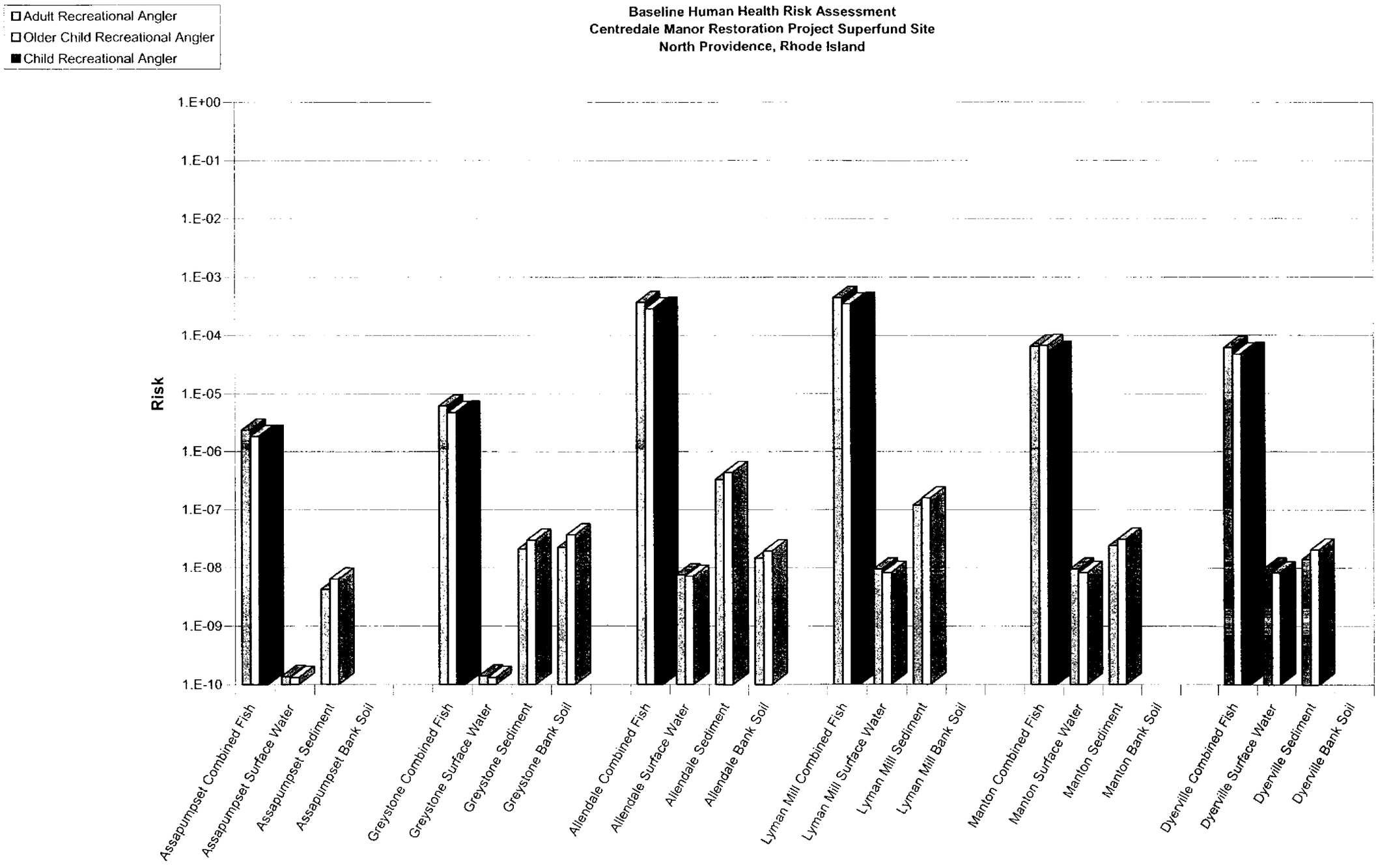
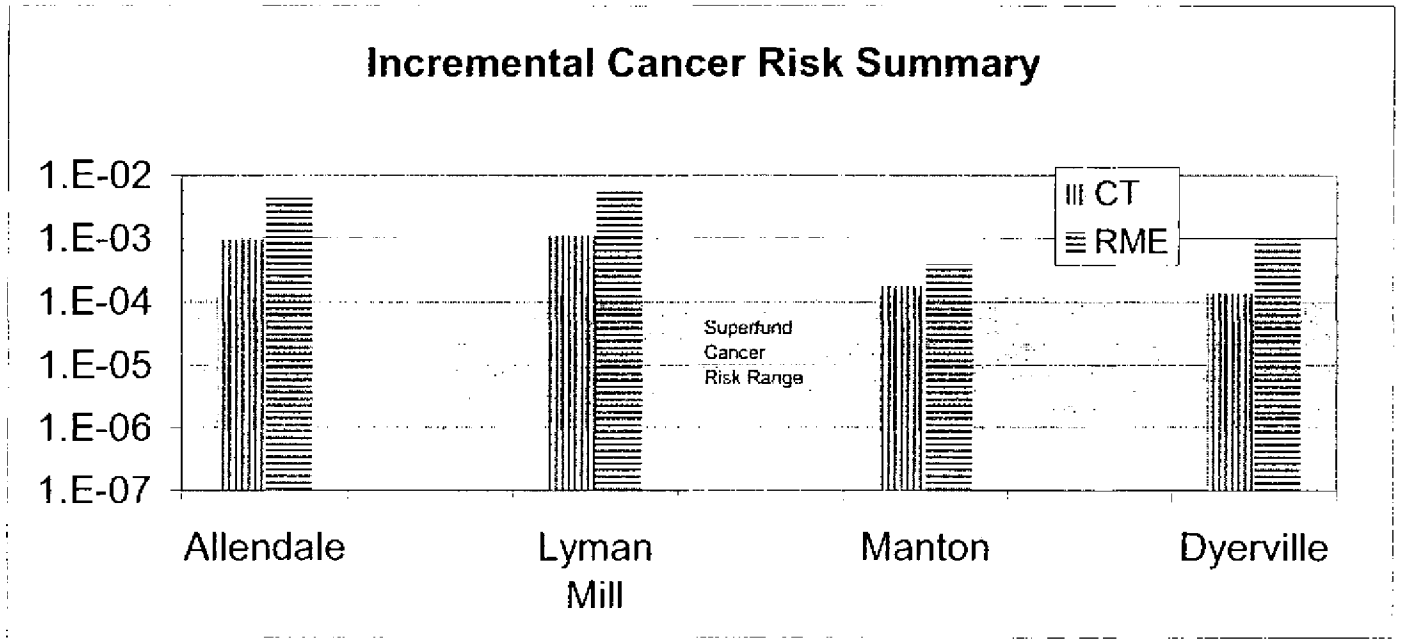
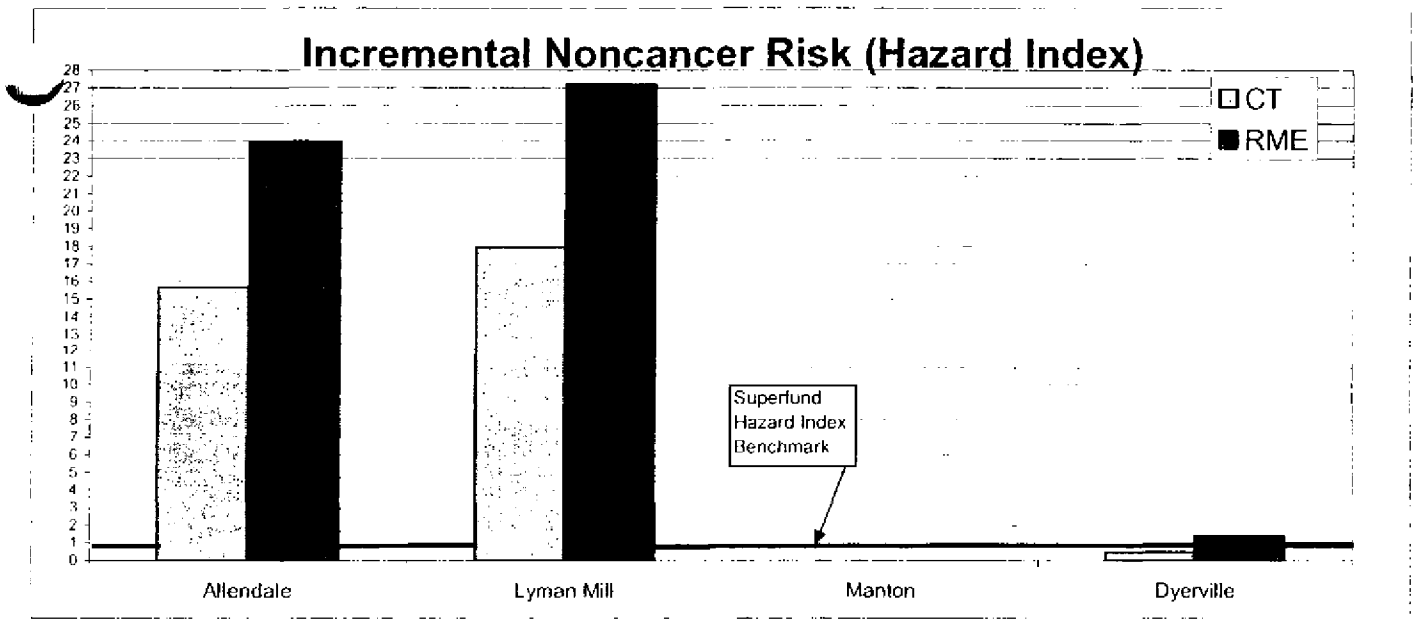


Figure 59. Summary of Incremental Cancer and Non-Cancer Risks -
Resident Living Along the River

Baseline Human Health Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island





CANCER RISK REPRESENTS THE SUM OF RISKS FOR ALL AGE GROUPS COMBINED
HAZARD INDEX REPORTED FOR YOUNG CHILD
INCREMENTAL HAZARD INDEX AT MANTON IS ZERO FOR RME AND CT SCENARIOS

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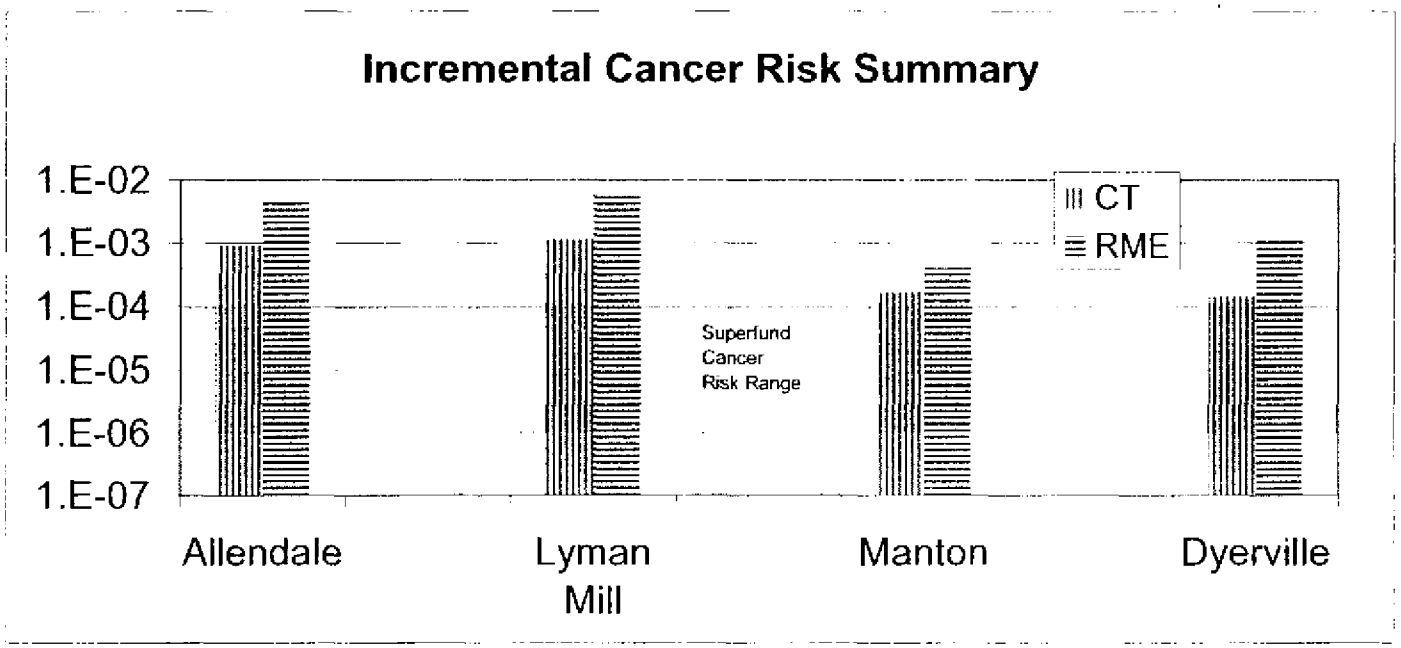
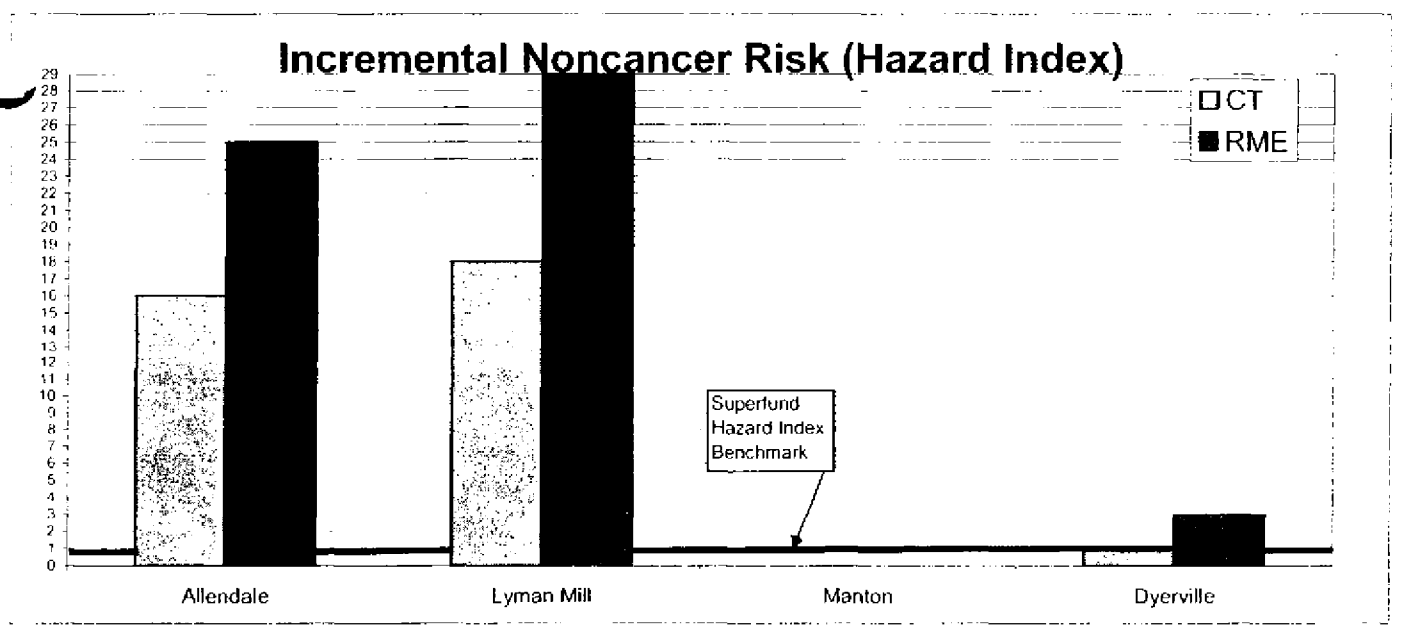
Figure 60. Summary of Incremental Cancer and Non-Cancer Risks -
Visiting Recreational Angler

Baseline Human Health Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island





CANCER RISK REPRESENTS THE SUM OF RISKS FOR ALL AGE GROUPS COMBINED
HAZARD INDEX REPORTED FOR YOUNG CHILD
INCREMENTAL HAZARD INDEX AT MANTON IS ZERO FOR RME AND CT SCENARIOS

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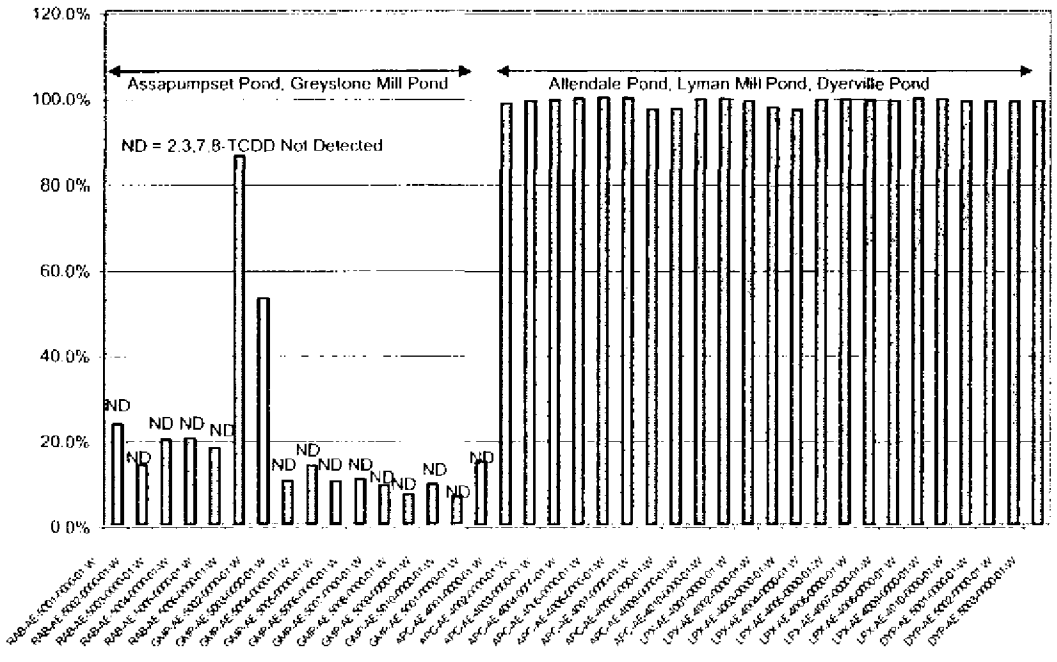
Figure 61. Portion of Dioxin TEQ in Fish Tissue Represented by 2,3,7,8-TCDD

Baseline Human Health Risk Assessment
Centredale manor Restoration Project Superfund Site
North Providence, Rhode Island

Portion of Dioxin TEQ represented by 2,3,7,8-TCDD in American eel



Portion of Dioxin TEQ Represented by 2,3,7,8-TCDD - Largemouth Bass

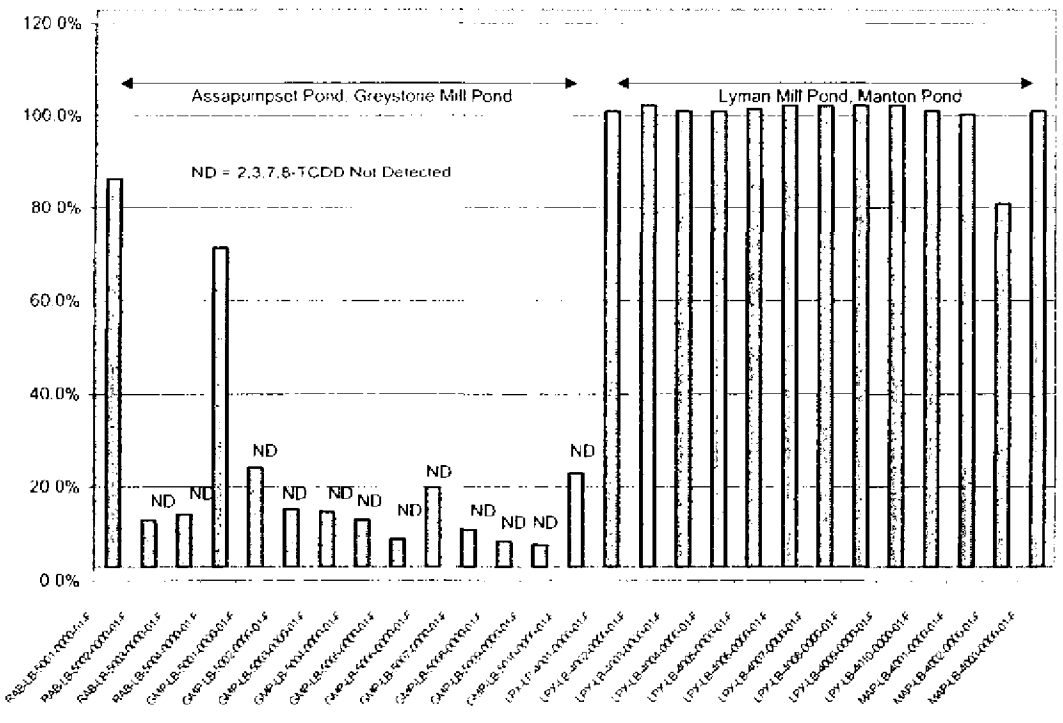


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Figure 62. Distribution of 2,3,7,8-TCDD and Contribution to Dioxin TEQ in Sediment

Baseline Human Health Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island

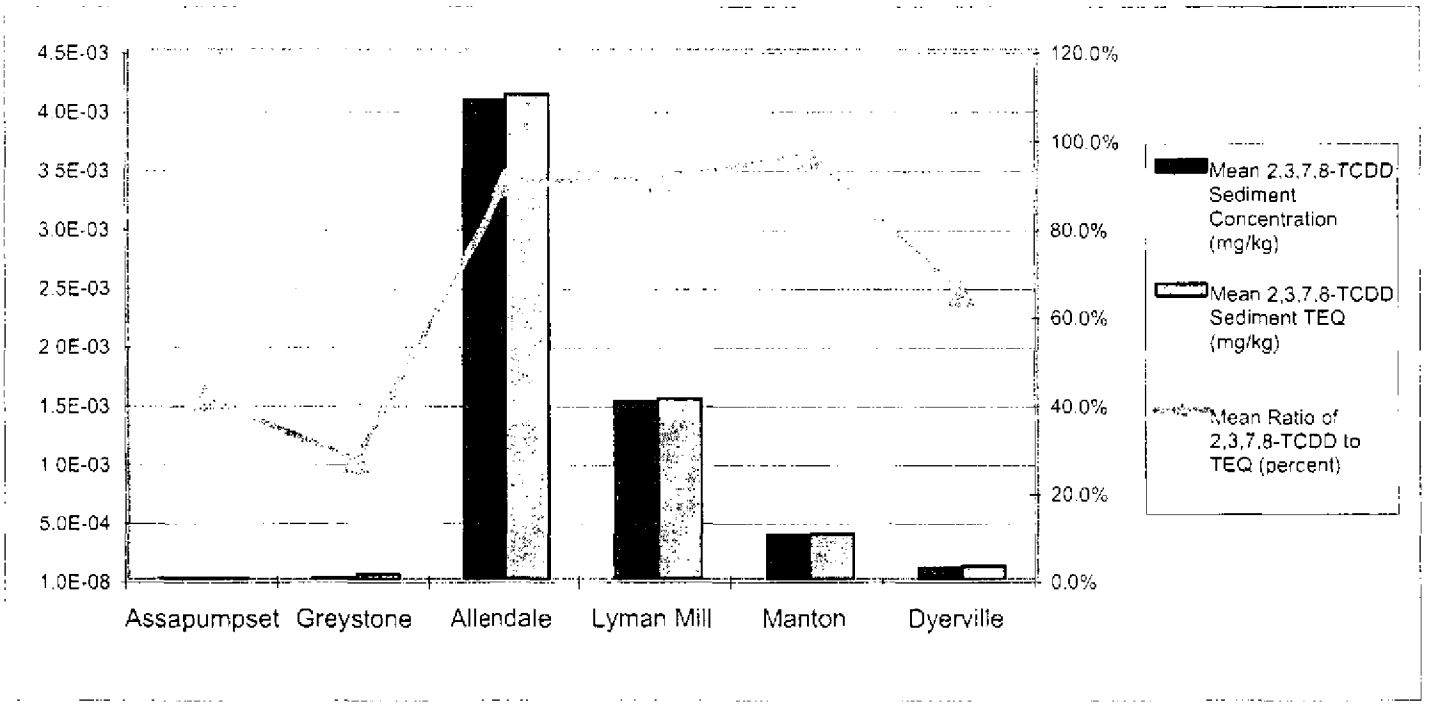

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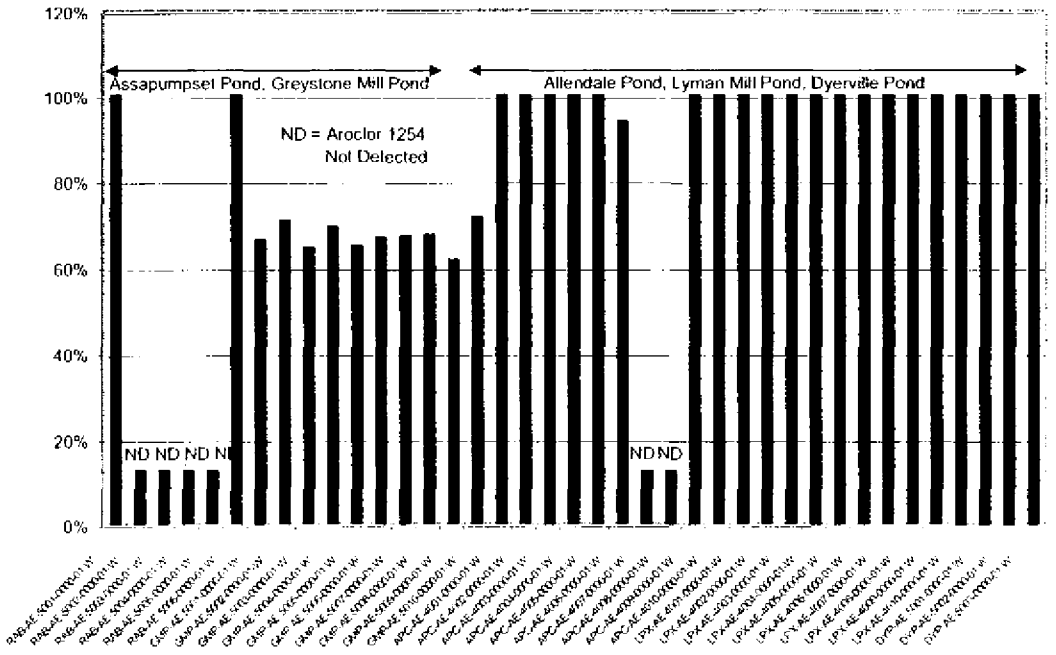
Figure 63. Portion of Total Aroclors in Fish Tissue Represented by Aroclor-1254

Baseline Human Health Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island

Portion of Total Aroclors Represented by Aroclor-1254 in American eel



Portion of Total Aroclors Represented by Aroclor-1254 in Largemouth Bass

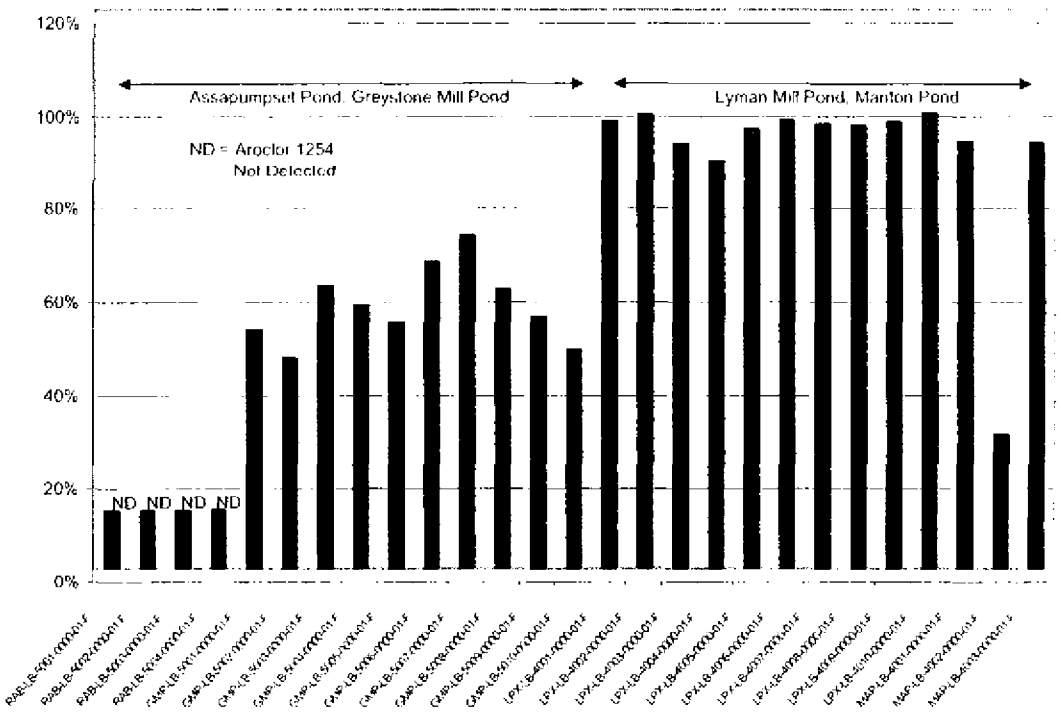


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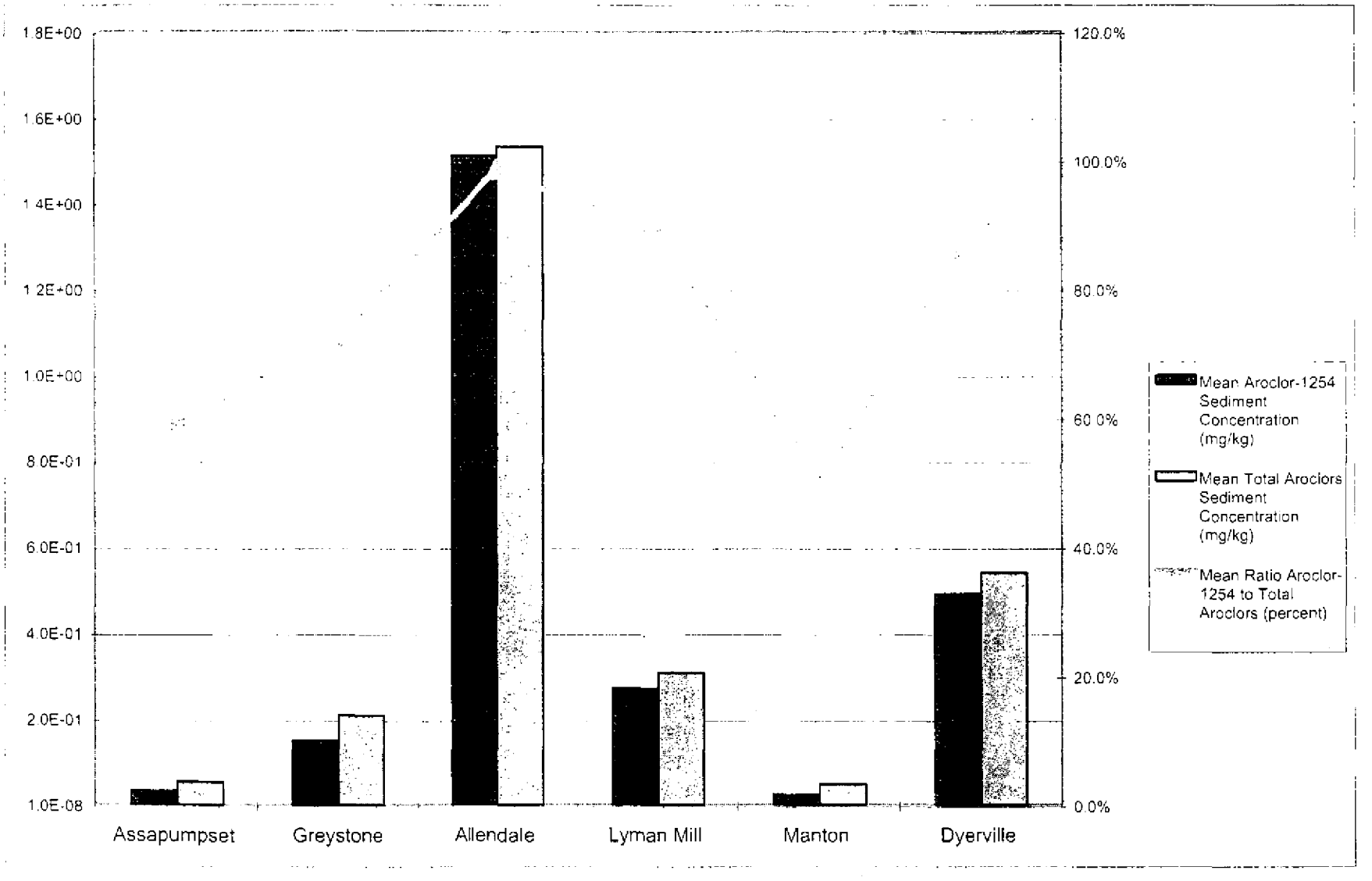
Figure 64. Distribution of Aroclor-1254 and Contribution to Total Aroclors in Sediment

Baseline Human Health Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



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P:\W9-GVT\COE-NAE\Battelle\Centredale\T25 - BCRA\INTERIM FINAL BHHRA\NEW FIGURES\
TEQ PCBC RatioHOME Sheet3 Chart 1

8/25/2005

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