RESPONSE ACTION CONTRACT FOR REMEDIAL, ENFORCEMENT OVERSIGHT, AND NONTIME-CRITICAL REMOVAL ACTIVITIES IN REGION 6

FOCUSED ECOLOGICAL RISK ASSESSMENT BENNETT'S DUMP SITE BLOOMINGTON, MONROE COUNTY, INDIANA

Prepared for

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Region 5
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ABBREVIATIONS AND ACRONYMS

95UCL	95^{th} percent upper confidence limit
Ah	Aryl hydrocarbon
AUF	Area use factor
ATSDR	Agency for Toxic Substances and Disease Registry
Bennett's Dump	Bennett's Stone Quarry
COPC	Contaminants of potential concern
CSM	Conceptual site model
CTE	Central tendency exposure
CV	Coefficient of variation
EPA	U.S. Environmental Protection Agency
EPC	Exposure point concentration
ERA	Ecological risk assessment
FERA	Focused ecological risk assessment
GLI	Great Lakes Initiative
HQ	Hazard quotient
IR	Ingestion rate
kg food/kg_{bw}-day	Kilogram food per kilogram body weight per day
LOAEC	Lowest-observed-adverse-effect concentration
LOAEL	Lowest-observed-adverse-effect level
mg/kg_{BW}-day	Milligram per kilogram body weight per day
NOAEC	No-observed-adverse-effect-concentration
NOAEL	No-observed-adverse-effect-level
PAC	Percent allowable consumption
PADI	Percent allowable daily intake
PCB	Polychlorinated biphenyl
ppm	Part per million
RME	Reasonable maximum exposure
Tetra Tech	Tetra Tech EM Inc.
TRV	Toxicity reference value
µg/kg	Microgram per kilogram
µg/kg_{BW}-day	Microgram per kilogram body weight per day
Westinghouse	Westinghouse Electric Corporation

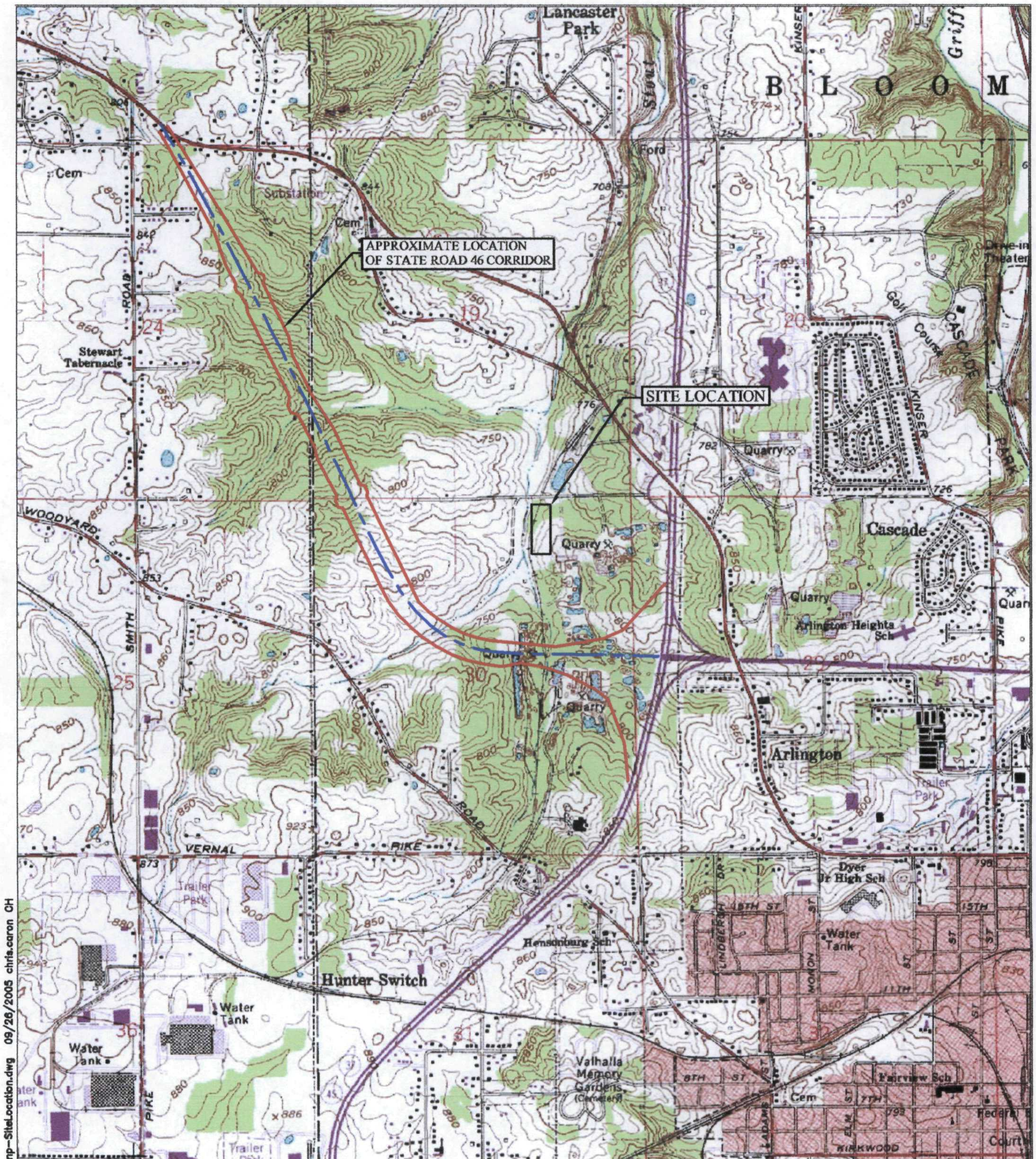
1.0 INTRODUCTION

Tetra Tech EM Inc. (Tetra Tech) has prepared this focused ecological risk assessment (FERA) report for the U.S. Environmental Protection Agency (EPA) in partial fulfillment of the statement of work for Response Action Contract No. 68-W6-0037 for Region 6, Work Assignment No. 945-TATA-05ZZ. The primary objective of this FERA is to investigate the protectiveness of the remedial activities conducted at the Bennett Stone Quarry site, which is also known as the Bennett's Dump site.

The FERA follows the approach developed by EPA for the FERA conducted for the Neal's Landfill site, as detailed in the *Focused Ecological Risk Assessment, PCBs and Mammalian and Avian Piscivores in Conard's Branch and Richland Creek* (EPA 2005). The following sections are included in the report: Section 1.0 describes the site history and ecological risk assessment (ERA) components, Section 2.0 presents the problem formulation, Section 3.0 presents the exposure assessment, Section 4.0 presents the ecological effects assessment, Section 5.0 presents the risk characterization results, Section 6.0 presents the uncertainty analysis, and Section 7.0 presents a summary. The references cited in the report are listed in Section 8.0. The report also contains four appendices and an attachment. Appendix A contains the data used in the FERA, Appendix B contains the risk estimate calculations, Appendix C contains the summary statistics used in the risk estimates, and Appendix D presents an assessment based on polychorinated biphenyl (PCB) congener data. An evaluation of toxicity values conducted by EPA (EPA 2005) is included as an Attachment.

1.1 SITE HISTORY

The Bennett's Dump site consists of two adjacent land parcels totaling about 4 acres in size. The site is located approximately 2.5 miles northwest of Bloomington in Monroe County, Indiana (see Figure 1). The site was formerly a limestone quarry pit, which was filled with waste materials including demolition debris, household wastes, and electrical parts. During the 1960s and 1970s, a large number of electrical capacitors containing PCBs were dumped at the site. Labeling on the capacitors linked the PCB contamination to the Westinghouse Electric Corporation (Westinghouse; later known as CBS Corporation and now known as Viacom), which manufactured capacitors in Bloomington between 1958 and the mid-1970s.



BENNETT'S DUMP
BLOOMINGTON, INDIANA

FIGURE 1
SITE LOCATION MAP

0 1000 2000
SCALE IN FEET

TETRA TECH EM INC.

SOURCE: MODIFIED FROM U.S. GEOLOGICAL SURVEY 7.5-MINUTE SERIES MAP, QUADRANGLE, BLOOMINGTON, INDIANA, 1966, PHOTOREVISED, 1990

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In 1983, Monroe County discovered the site and requested that EPA perform an emergency removal. EPA removed capacitors left at the surface (i.e., not buried) and installed a clay cap, security fencing, and warning signs. The Bennett's Dump site was placed on the National Priorities List in September 1984. In 1985, EPA, the State of Indiana, Monroe County, the City of Bloomington, and Westinghouse (now Viacom) signed a consent decree. Under the terms of the consent decree, Viacom is to remediate six sites in the Bloomington area containing PCBs. The Bennett's Dump site is one of the six sites covered by the consent decree. In 1987, 252 PCB-contaminated capacitors, 14 cubic yards of soil, and PCB-contaminated sediment in Stout's Creek were removed from the site. In 1994, an agreement was reached to explore other remedial alternatives; however, little progress was made, and in 1997, a judicial order was issued that stated that all excavation activities should be completed by December 2000. In October 1998, EPA signed a Record of Decision Amendment, which specified the following remedy:

- Excavation and off-site disposal at a permitted landfill of all materials with a PCB concentration greater than 25 parts per million (ppm) on average (estimated volume of 55,000 cubic yards), followed by placement of a 12-inch clean soil cover
- Incineration of PCB-containing capacitors at a permitted incinerator
- Excavation of sediment in Stout's Creek with PCB concentrations greater than 1 ppm, with subsequent placement under the clean soil cover
- Monitoring of groundwater monitoring wells and on-site springs and deed restrictions

Excavation activities began in August 1999. A total of 36,172 tons of PCB-contaminated material was excavated and disposed of in an off-site landfill permitted to accept PCBs. A total of 1,756 capacitors (118.72 tons) were excavated and incinerated at an off-site incinerator permitted to accept PCBs. The site was then covered with a 12-inch clean soil cover. The final PCB clean up value was 11.3 ppm on average. Excavation activities, including sediment removal, were completed in November 1999. An additional 10 cubic yards of sediment was excavated from Stout's Creek in September 2000; sediment with a maximum PCB concentration greater than 2.7 ppm were excavated and placed under the clean soil cover.

Viacom is currently conducting a hydrogeologic investigation to characterize groundwater flow and determine possible impacts to local surface water. The results of the hydrogeologic investigation and risk assessment will provide information that will be used to determine what types of additional clean up activities are required (EPA 2004a).

1.2 ECOLOGICAL RISK ASSESSMENT COMPONENTS

ERAs generally:

- Characterize the current and potential threats to the environment.
- Establish clean up levels for the selected remedy that will protect natural resources (i.e., plants and animals).
- Evaluate the ecological impacts of remediation strategies (EPA 1997).

The process for performing an ERA is described in the *Ecological Risk Assessment Guidance for Superfund: Process for Designing and Conducting Ecological Risk Assessments* (EPA 1997). One of the first steps in the ERA process is the problem formulation, which includes the following elements:

- **Identification of Contaminants of Potential Concern (COPC; or "stressors"):** Identifies those COPCs attibutable to the source (or site) that are likely to present a risk to the ecosystem.
- **Evaluation of Contaminant Release, Migration, and Fate:** Describes what is known about the extent of contamination, fate, and transport processes (i.e., transport from soil to surface water via runoff or degradation processes).
- **Identification of Receptors:** Receptors are those individual organisms or animals, populations, or communities that are exposed (or potentially exposed) to a COPC through a complete exposure pathway. A COPC moves from a source to a receptor through an exposure pathway.
- **Identification of Effects:** After the COPCs are identified, possible effects resulting from exposure are reviewed.
- **Selection of Endpoints:** *Assessment* endpoints identify critical effects for the receptor; for example, a decrease in reproductive success can be a critical endpoint as it may impact population/community stability. *Measurement* endpoints represent how the critical effect will be estimated or measured (i.e., comparison of COPC concentrations in a receptors diet to dietary concentrations demonstrated to cause reproductive effects in biological studies).

The end product of the problem formulation step is a conceptual site model (CSM) identifying the (1) environmental receptors at risk (what ecological components need protecting), (2) data needed, and (3) analyses to be used. The CSM focuses the ERA on those ecological components demonstrating complete exposure pathways and critical effects.

The problem formulation step is followed by the exposure assessment, which quantifies the magnitude and type of exposure. Key elements include (1) quantifying contaminant release, fate, and transport; (2) characterizing receptors; and (3) estimating exposure point concentrations (EPC).

The ecological effects assessment quantitatively links concentrations of COPCs to adverse effects in receptors. The effects assessment identifies how much of a COPC has the potential to cause how much of an effect. The "quantitative link" between COPC concentrations and a potential adverse effect can be provided by literature reviews, field testing, and/or toxicity testing.

The exposure assessment and the effects assessment are combined in the risk characterization step. During risk characterization, the results of the exposure assessment (i.e., the EPC) is compared to the concentration required to produce an adverse effect. A receptor is considered at risk when the EPC (i.e., concentration in diet or dose) exceeds the concentration demonstrated to produce an adverse effect.

As part of the risk characterization, an uncertainty analysis is conducted. During the ERA process, assumptions are made, all of which contribute to uncertainty in risk evaluations. Lacking site-specific information, assumptions are developed based on best estimates of data quality, exposure parameters, and dose-response relationships. The purpose of the uncertainty analysis is to provide a summary of those factors that may influence the risk results, evaluate their variability, and determine their contribution to an over- or underestimation of the overall risk assessment results.

The ERA concludes with a summary regarding the estimated ecosystem risk. If appropriate, preliminary remedial goals may be calculated.

2.0 PROBLEM FORMULATION

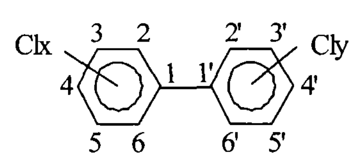
The purpose of this FERA is to investigate the protectiveness of the remedial activities conducted at the Bennett's Dump site. This FERA focuses solely on PCB-related risks to wildlife (specifically, piscivorous [fish-eating] birds and mammals) in Stout's Creek, downstream of the Bennett's Dump site. PCBs are the only COPCs evaluated in this FERA, as remediation at the Bennett's Dump site focused on the removal of PCB-contaminated soils and waste material. Soil with PCB concentrations greater than 25 ppm were excavated and disposed of at an off-site landfill, and a 12-inch clean soil cover was then placed on the site; therefore, future direct contact with PCB-containing soil by terrestrial receptors is unlikely.

Prior to remedial activities at the site, PCB-contaminated sediments at the site were likely washed into the bedrock aquifer. Groundwater discharging from the Bennett's Dump site percolates through conduits in the karst formations to springs located on site and eventually discharges into Stout's Creek.

The following sections (see Sections 2.1 through 2.4) of the problem formulation discuss the COPCs (PCBs), CSM, assessment endpoint, and measurement endpoint.

2.1 POLYCHLORINATED BIPHENYLS

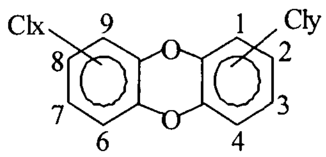
PCBs are produced by the chlorination of a biphenyl molecule. The general structure of PCBs is shown below.



Up to 209 different compounds (called congeners) can be formed based on the degree and position of the chlorine atoms (Cl_x and Cl_y). Congeners with the same number of chlorine atoms (i.e., three) are called isomers and make up a homolog group (i.e., trichlorobiphenyls).

Monsanto Corporation marketed PCBs under the name Aroclor and was the major producer between 1930 to 1977. PCBs were useful in a variety of applications due to their chemical and thermal stability. The different Aroclors are identified by a four-digit code with the first two digits indicating the type of mixture and the last two digits indicating the approximate amount of chlorination (percent weight), for example, Aroclor 1248. Trade names of PCB mixtures produced in other countries include Clophen, Fenclor, Kaneclor, and Phenoclor. The manufacture, processing, distribution, and use of PCBs was banned in the U.S. in 1977 (Agency for Toxic Substances and Disease Registry [ATSDR] 2000), in part, due to their toxicity and persistence in the environment.

A subset of PCB congeners have a structural configuration similar to dioxin. The general structure of dioxin is shown below.



PCB congeners that are coplanar (i.e., the phenyl rings are oriented in the same plane) and are chlorinated at the 4 and 4' positions (i.e., para positions) and at least two of the 3, 3', 5, or 5' positions (i.e., meta positions) are referred to as "dioxin-like PCB congeners;" these congeners exhibit toxicity similar to dioxin once absorbed within cells. Dioxin-like PCB congeners include congeners 77, 81, 105, 114, 118, 123, 126, 156, 157, 167, 169, and 189. For example, PCB congener 77 is chlorinated at the 3, 3', 4, and 4' positions; PCB congener 123 is chlorinated at the 2', 3, 4, 4', and 5 positions.

2.1.1 Fate and Transport

PCBs are nonpolar, lipophilic compounds. In general, PCBs are relatively insoluble in water, with solubility decreasing with increasing degree of chlorination. PCBs are relatively soluble in nonpolar solvents and lipids. In addition to being more water-soluble, the lower chlorinated congeners are also more volatile and susceptible to degradation processes (such as photolysis and microbial degradation) than the more highly chlorinated congeners.

As PCBs are no longer manufactured or imported in large quantities, uncontrolled releases to the environment are rare. PCBs may be released to the environment from uncontrolled landfills/hazardous waste sites, incineration of PCB-containing material, leakage from electrical equipment, or the improper disposal of PCB-containing material (ATSDR 2000). Once in the environment, PCBs partition between media (i.e., soil to water, water to air, or sediment to water). As this FERA focuses on the aquatic habitat, specifically the release of PCB-impacted groundwater to Stout's Creek, the following discussion focuses on the fate and transport of PCBs in the aquatic environment.

At the air-water interface, volatilization of PCBs from water to the atmosphere may occur. For PCBs within the water column, photolysis is the primary degradation process, with the lower chlorinated congeners being more susceptible. Due to their relatively low water solubility and high octanol-water partition coefficients (a measure of hydrophobicity), PCBs tend to sorb strongly to suspended solids and sediments. For those PCBs bound to sediment, biodegradation is the principle degradation process. Although sediment may serve as a sink for PCBs in the aquatic environment, it is possible for PCBs in sediment to serve as a continuing source of PCBs to the water column. As PCB concentrations in the water column decrease, PCBs may desorb from the sediment back into the water column.

Dissolved-phase PCBs can also be taken up directly from the water column (bioconcentration) by aquatic organisms. Aquatic organisms bioaccumulate PCBs through combined exposure to PCB-containing food items, water, and sediment; therefore, upper trophic-level aquatic consumers would be expected to have higher PCB concentrations than their prey. In general, the low-chlorinated congeners are more readily metabolized, while the higher chlorinated congeners are slowly metabolized and preferentially retained in the tissues (especially in the lipids); dioxin-like congeners tend to be poorly eliminated and therefore, bioaccumulate to a greater degree. Within the food web, as each trophic level preferentially accumulates the higher-chlorinated congeners, it is expected that the top-level consumers will have the highest levels of the higher-chlorinated congeners. Protection of piscivorous wildlife from risks associated with PCB exposure should be protective of other aquatic organisms (fish, invertebrates, and plants).

2.1.2 Polychlorinated Biphenyl Ecotoxicity

PCBs exhibit a broad spectrum of effects, including effects on the gastrointestinal system, liver, respiratory system, nervous system, immune system, reproductive system, and endocrine system (Hansen 1994). Certain coplanar PCB congeners have a structure similar to 2,3,7,8-tetrachloro-dibenzo-p-dioxin. The mechanism of toxicity for dioxin involves binding to the aryl hydrocarbon (Ah) receptor. Dioxins are potent Ah receptor agonists. PCBs with similar structures can also bind to the Ah receptor and exhibit dioxin-like toxicity, but are less potent than dioxins. For the noncoplanar PCBs, mechanisms of toxicity are not as well characterized, but include lipid accumulation and vitamin A depletion (which are also associated with Ah receptor activity), enzyme induction, and interference with heme synthesis (Hansen 1994).

PCB-exposure has been related to decreased reproductive success in wildlife populations. Reproductive toxicity in female animals has been established in a number of oral exposure studies; information on the reproductive toxicity in male animals is limited (ATSDR 2000). Effects include decreased conception, complete inhibition of reproduction, and decreased fertility. Mink and monkeys have been demonstrated to be particularly susceptible to the effects of PCBs. Reproductive failure has been shown to occur at concentrations of 2 ppm for mink (Aroclor 1254) (Aulerich and others 1985, as cited in Hansen 1994). Although the adults were not affected, a high death rate of kits resulted. Rhesus monkeys (*Macaca mulatta*) are also sensitive to PCB exposure. Female monkeys have demonstrated increased stillbirths, lowered birth rate, and altered behavioral patterns (Eisler 1986). Exposure to 0.8 milligram per kilogram body weight per day (mg/kg_{BW}-day) Aroclor 1248 for 2 months resulted in a reduced conception rate (Allen and others 1974, as cited in ATSDR 2000).

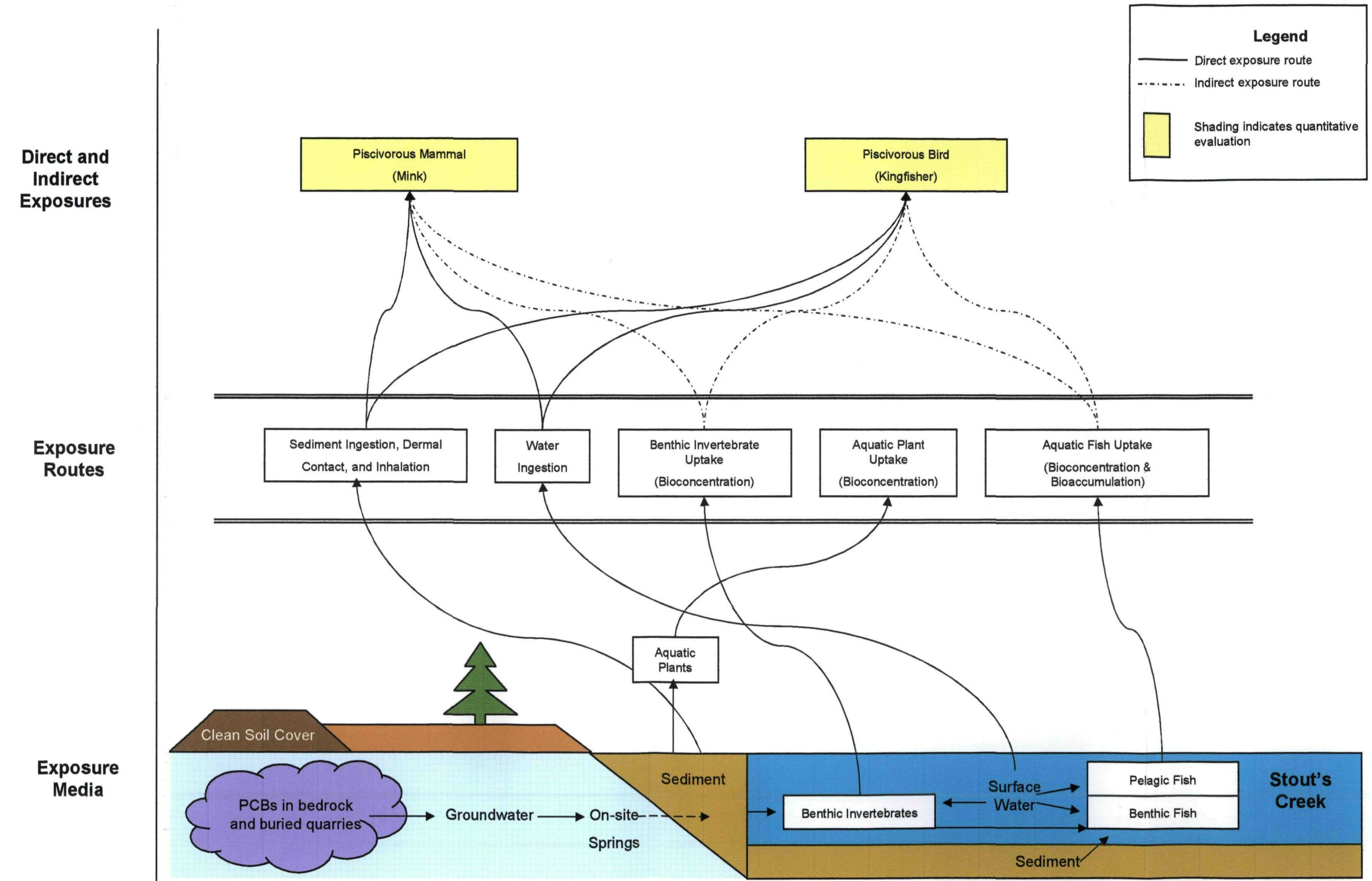
In birds, PCBs affect normal patterns of growth, reproduction, metabolism, and behavior (Eisler 1986). Fish-eating birds accumulate PCBs through their diet. PCB concentrations in the liver were highest in birds that fed on fish, followed by birds feeding on small birds and mammals, worms, and insects, and lowest in plant-eating species (Eisler 1986). PCB exposure in birds has been linked to low reproductive success and deformed chicks. Hormonal and behavioral effects have also been observed in wild bird populations.

Delayed reproductive impairment has been noted in doves fed Aroclor 1254 in the diet. Although the first clutch was not reduced, the hatchability of the second clutch was reduced. Chromosomal aberrations were noted in the embryos. Doves fed PCBs during the courtship phase demonstrated decreased nest-building and incubation (indirect effect on reproduction). Egg hatchability was decreased in hens fed Aroclors in the diet. Although eggshell thinning has been observed in birds with measurable levels of PCB residues, the results are confounded due to the presence of other contaminants (i.e., p,p'-dichlorodiphenyltrichloroethane).

2.2 CONCEPTUAL SITE MODEL

The CSM illustrating potential exposure pathways for PCB exposure for ecological receptors at the Bennett's Dump site is shown on Figure 2. As PCBs in soil were addressed during the remedial action (through excavation and off-site disposal), exposure to PCBs in soil was not considered to represent a significant source of exposure. Therefore, terrestrial exposure pathways are not evaluated in the FERA. The primary release of PCBs from the Bennett's Dump site (post-remediation) is through the infiltration of groundwater, which flows to two springs located on site and eventually discharges to Stout's Creek. Although sediment interactions are depicted in the CSM, a significant reservoir of PCBs in sediment is not expected at this site as contaminated sediments were removed from Stout's Creek during the remedial action. The PCB exposure routes evaluated in the FERA include direct absorption from water (bioconcentration) by aquatic organisms and dietary exposures through consumption of PCB-contaminated food items. Direct absorption of a chemical from water is called "bioconcentration;" exposure through food is known as "trophic transfer" or "biomagnification;" and the integrated exposure through all routes is referred to as "bioaccumulation" (in this example, the combined effects of bioconcentration, trophic transfer, and sediment ingestion). Fish are shown as receiving an integrated exposure through all exposure routes (See Figure 2).

The transfer of PCBs from surface water to sediment and from sediment back to surface water is shown on Figure 2, as is the direct transfer of PCBs from sediment to aquatic receptors. Although exposure to sediment is a potentially complete exposure pathway for aquatic receptors, either by incidental ingestion of sediment during consumption of prey/food or by direct contact, this pathway was not evaluated in the FERA. The portion of the stream bed of Stout's Creek at Station 1 is described as sand/silt; however, at



Notes:

PCB Polychlorinated biphenyl

BENNETT'S DUMP SITE BLOOMINGTON, INDIANA	**FIGURE 2** CONCEPTUAL SITE MODEL
TETRA TECH EM INC.	

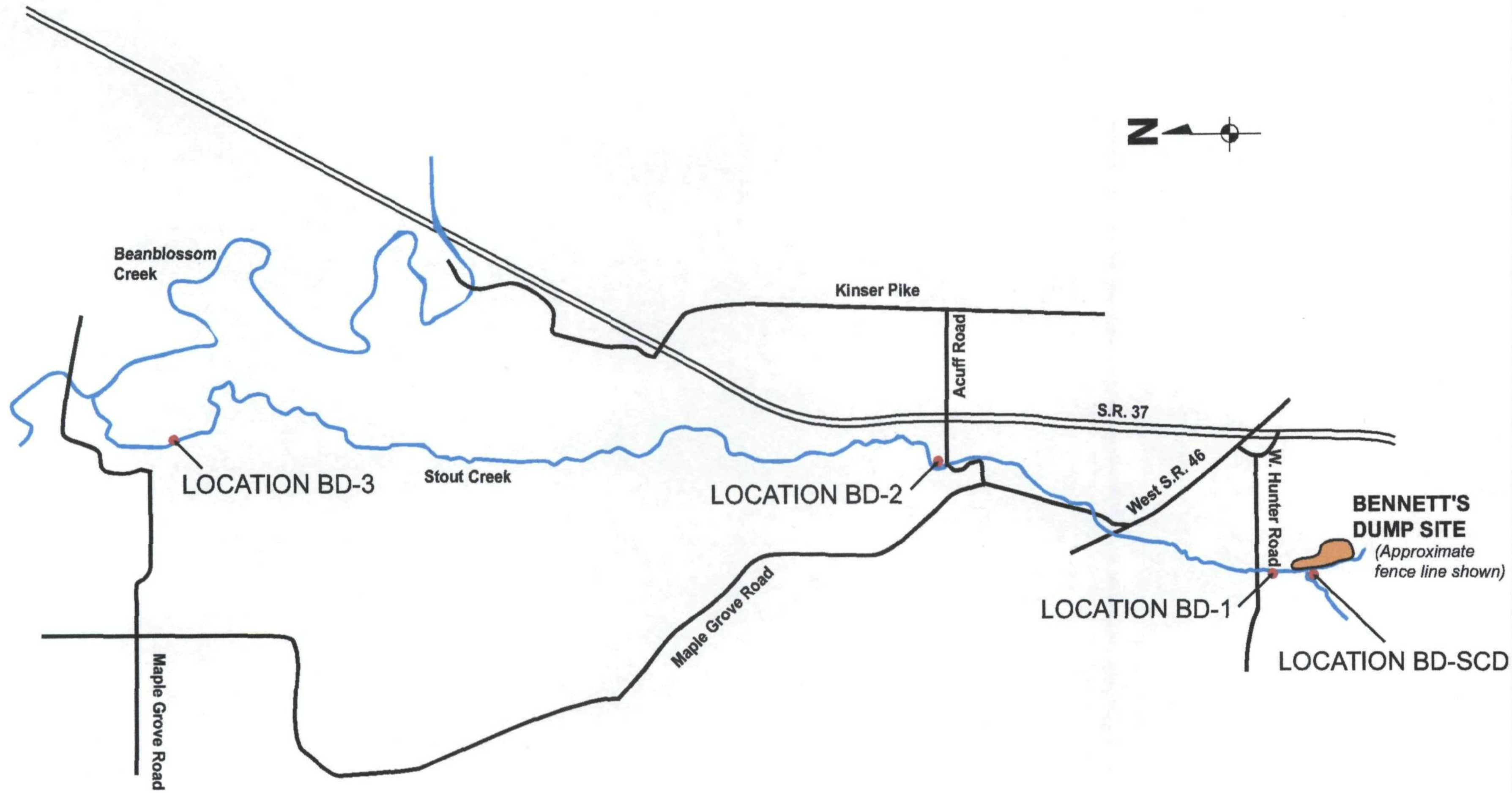
Station 2, the stream bed is described as consisting of mainly cobbles/gravel, but then changes back to sand/silt at Station 3 (Normandeau Associates, Inc. 2004; station locations are presented in Figure 3). For upper trophic levels, exposure to PCBs in sediment during incidental ingestion is expected to be minor (dry weight basis) compared to exposure to PCBs in prey items.

Exposure of piscivorous mammals and birds to PCBs in the diet (i.e., fish and crayfish) was evaluated in the FERA. Although piscivores may be exposed to PCBs through additional exposure routes such as ingestion of PCB-contaminated water or sediments and inhalation of PCBs that volatilize from the water surface, the contribution from these types of exposures was considered minor and were not evaluated in the FERA. For instance, bioconcentration factors for aquatic species can range from 500 to 300,000 (ATSDR 2000), depending on the degree of chlorination. Therefore, the concentration in the prey item (fish or crayfish) may be expected to be 500 to 300,000 times greater than the concentration in the water. An upper level piscivore would need to consume a large amount of surface water to approach the concentrations expected in the piscivorous diet.

Volatilization of PCBs from water to the atmosphere does play a role in influencing global distribution of PCBs; however, the inhalation of PCBs in air above Stout's Creek is expected to be a minor contributor and was not evaluated in the FERA.

2.3 ASSESSMENT ENDPOINT

Reproductive effects in seals, mink, and migratory water birds have been associated with PCB exposure. As PCBs bioaccumulate in the food chain, higher trophic level consumers will have a greater exposure to PCBs. Fish would be expected to have a higher level of PCBs than the aquatic invertebrates and aquatic plants they feed on; consequently, fish-eating animals would be expected to have higher PCB levels than the prey (fish) they consume. Studies also indicate that certain piscivores are sensitive to the effects of PCBs. For this reason, the FERA focused on the protection of piscivorous wildlife from PCB-related risks, which should be protective of lower trophic level aquatic organisms (i.e., fish, invertebrates, and plants).



Legend

● Stream sediment, water, and fish sampling location

NOT TO SCALE

BENNETT'S DUMP
BLOOMINGTON, INDIANA

FIGURE 3
MAY 2004 DOWNSTREAM SEDIMENT, WATER, AND FISH SAMPLING LOCATIONS

Tetra Tech EM Inc.

As discussed in Section 2.1.2, reproductive effects due to PCB exposure have been observed in wildlife. Reproductive success can be adversely impacted by PCB-exposure, including premature births, malformed offspring, and behavioral effects (i.e., inattentiveness or nest abandonment in birds). Adverse impacts on reproduction can affect population stability as the population may not be able to maintain its numbers or the population may become skewed towards the adult-age animals. Due to the potential impact on species populations, any significant reduction in reproductive endpoints is considered to be an ecologically significant adverse effect. The assessment endpoint can be stated as:

- Protection of piscivorous receptors in Stout's Creek that may ingest PCB-contaminated food from a reduction in reproductive success.

The testable hypothesis identified for this assessment endpoint is:

- Levels of PCBs in fish and crayfish are sufficient to cause reproductive effects in piscivorous receptors in Stout's Creek through dietary exposure.

2.4 MEASUREMENT ENDPOINT

To assess ecological risks, measurement endpoints were identified. A measurement endpoint is defined as a "measureable ecological characteristic that is related to the valued characteristic chosen as the assessment endpoint" and measures biological effects (EPA 1997). For each assessment endpoint identified (i.e., reproductive success), one or more measurement endpoints are selected to integrate modeled data or field data with the individual assessment endpoint.

Modeled dietary intake of PCBs were used to evaluate the potential risk to piscivorous mammals and birds that may consume fish and crayfish from Stout's Creek. The selected measurement endpoint receptor species for piscivorous mammals is the mink (*Mustela vison*) and for piscivorous birds is the kingfisher (*Ceryle alcyon*). These two receptors were selected because (1) the majority of Stout's Creek provides suitable habitat for both receptors; (2) natural history information (i.e., dietary composition and home range) is available showing that; both the mink and kingfisher have dietary compositions that maximize exposure; and (3) mink and avian piscivores (represented by the kingfisher) have been shown to be sensitive to the effects of PCB exposure.

As described in Section 2.1.2, mink have been shown to be sensitive to PCB exposure (Eisler 1986; EPA 1993); they are abundant and widespread, being found in various aquatic habitats, including rivers, streams, lakes, and ditches, as well as swamps and marshes (Linscombe and others 1982, as cited in EPA 1993). Avian piscivores have been shown to be sensitive to the effects of PCB exposure, especially the dioxin-like effects of certain PCB congeners. For the FERA, the kingfisher was selected to represent the avian piscivore guild. Kingfishers are found along rivers and streams and lake and pond edges (EPA 1993). Waters that are relatively shallow (less than 2 feet below ground surface), clear, and free of thick vegetation (which obscures their view of the water) are preferred. Stream riffles are preferred as fish tend to accumulate at riffle edges.

Exposure through the diet was estimated based on site-specific measurements of PCBs in Stout's Creek fish and measured and modeled PCB concentrations in crayfish. In addition, for the kingfisher, a dose-related exposure was calculated by adjusting the concentration in the kingfisher diet by its ingestion rate.

Dietary studies evaluating exposure to PCBs in the diet have been performed on mink; therefore, dietary concentrations estimated from field data (measured/modeled concentrations in fish/crayfish) can be compared directly to dietary concentrations in controlled studies. No dietary studies on PCBs in the kingfisher were identified; consequently, a direct diet-to-diet comparison cannot be made. Instead, an extrapolation was made to compare the PCB exposure concentration for the kingfisher to the PCB exposure concentrations for other avian species. The interspecies comparison was done on the basis of dose by converting the dietary concentration to a dose (amount of PCBs ingested per body weight per day).

The measures of effects for mink are studies that identify the reproductive effect levels associated with feeding PCB-contaminated fish to mink (see Section 5.1); for the kingfisher, laboratory studies conducted with other avian species, including chickens, pheasants, doves, and cormorants, and a field study of bald eagles, were used to identify the reproductive effect level (dose) for avian receptors (see Section 4.2). The reproductive effect levels were used to evaluate the level of risk associated with exposure to PCBs in the diet.

Dietary intake of dioxin-like PCB congeners was also modeled for the mink; accumulation of dioxin-like PCB congeners in kingfisher eggs was also modeled. The assessment of risks associated with exposure to PCB congeners in Stout's Creek is included in Appendix D. As only limited PCB congener data is available, the congener-based assessment is not equivalent to the total PCB (as Aroclor) assessment and is not presented in the main text.

3.0 EXPOSURE ASSESSMENT

Exposure parameters, including dietary composition and home range/site utilization for the mink and kingfisher, are discussed in Sections 3.1 and 3.2, respectively.

PCB-contaminated soil was addressed during the removal action. The only complete exposure pathway considered in the FERA is the release of PCB-containing material (i.e., soil/sediment with PCB concentrations below the clean up levels or material washed into the bedrock aquifer) to groundwater, which flows to the two on-site springs, and is subsequently released from the springs to Stout's Creek (see Figure 2); a dietary composition was selected to maximize the contribution of aquatic food items. Both the mink and the kingfisher are assumed to feed equally on the various fish species available. The composition of the mink and kingfisher diets used in this FERA are discussed in Sections 3.1.1 and 3.2.1, respectively.

Home ranges are the areas over which animals travel during routine activities, such as foraging for food. The site area use factor (AUF) is that portion of the affected area that falls within a particular animal's home range. The AUFs for the mink and kingfisher used in this FERA are discussed in Sections 3.1.2 and 3.2.2, respectively.

3.1 MINK EXPOSURE PARAMETERS

Dietary composition and home range assumptions for mink inhabiting Stout's Creek are discussed in the following sections.

3.1.1 Mink Dietary Composition

Based on observations in 31 mink collected along Michigan streams, the mink diet was found to be 61 percent fish, 5 percent amphibians, 11 percent crustaceans, 2 percent insects, 17 percent bird/mammal prey, and 4 percent unidentified (Alexander 1977, as cited in EPA 1993). Dietary composition is also available for mink along Michigan rivers; however, as Stout's Creek more closely resembles a stream, the dietary information for mink collected along streams was determined to be appropriate for use in this FERA.

Using the dietary composition for mink living along Michigan streams, an estimate of dietary exposure for mink along Stout's Creek was determined assuming consumption of fish and crayfish only. For this FERA, the percent fish (61 percent) and amphibians (5 percent) from the Michigan stream study were combined to represent the percent total fish consumption percentage (66 percent). The percentages for crustaceans (11 percent) and insects (2 percent) from the Michigan stream study were combined to represent percentage of crayfish consumed (13 percent). The following general equation was used to model the concentration of PCBs in the mink diet for Stout's Creek:

$$C_{diet-mink} = (0.66 \text{ x } C_{fish}) + (0.13 \text{ x } C_{crayfish})$$

where

$C_{diet-mink}$	=	Concentration of PCBs in mink diet
C_{fish}	=	Measured concentration of PCBs in Stout's Creek fish
$C_{crayfish}$	=	Measured/Modeled concentration of PCBs in Stout's Creek crayfish

Using this approach, 79 percent of the mink diet is modeled. All fish species were assumed to be consumed equally. The remaining 21 percent of the diet, estimated to be terrestrial prey items (mammals and birds), is not accounted for as site-specific PCB data for terrestrial receptors are not available for the Bennett's Dump site. As PCB-contaminated soil has been addressed by the previous remedial actions (either by off-site disposal or consolidation and capping), terrestrial receptors at the site should have little to no direct contact with PCB-contaminated soil. Therefore, PCB concentrations in terrestrial prey at the site are expected to be less than concentrations detected in aquatic receptors. Therefore, mink exposure to PCBs in the diet may be underestimated by up to 21 percent; however, the actual underestimation may be less than 21 percent as terrestrial prey are expected to have lower PCB tissue concentrations compared to aquatic prey. The potential underestimation of PCBs in the mink diet due to the exclusion of terrestrial prey items is discussed as an uncertainty (see Section 6.0).

3.1.2 Mink Home Range and Area Use Factor

A home range of 1 stream mile was selected for the mink based on the home range of adult female mink along Swedish streams (Gerell 1970, as cited in EPA 2005). In this study, adult female mink were found to range from 0.6 stream mile up to 1.7 stream miles with a mean value of 1.1 stream miles. Mean home ranges for adult male mink were greater; however, since the toxicity endpoint for mink is based on reproductive effects, and reproductive effects in female animals have been observed, the smaller home range of the adult female is considered appropriate.
The shape of the home range is dependent on the habitat type; for riverine habitats, the home range is usually linear in shape, while marsh habitats are more circular (Birks and Linn 1982; Eagle and Whitman 1987; as cited in EPA 1993). For this FERA, the home range was assumed to be linear in shape, parallel to, and including Stout's Creek.

As the home range is 1 stream mile and each of the reaches of Stout's Creek evaluated in the FERA are greater than 1 mile apart, the site utilization factor (or AUF) was determined to be 1; therefore, 100 percent of the mink's diet is considered to be food items within a specific reach.

3.2 KINGFISHER EXPOSURE PARAMETERS

Dietary composition and home range assumptions for kingfisher inhabiting Stout's Creek are discussed in the following sections.

3.2.1 Kingfisher Dietary Composition

Although the kingfisher diet consists mainly of fish, they also consume crayfish (EPA 1993). The dietary assumptions used in this FERA are based on observations from three studies. For Michigan streams, Alexander (1977, as cited in EPA 2005) reported diets consisting of 86 percent fish, 9 percent amphibians, and 5 percent insects for 17 kingfishers; Salyer and Lagler (1946, as cited in EPA 1993) reported a diet of 41 percent crayfish with the remainder consisting of various fish species. For Ohio streams (Davis 1982, as cited in EPA 1993), a diet of 13 percent crayfish was reported with the remainder consisting of various fish species. For the FERA, the percentage of crayfish in the kingfisher diet was estimated by determining the mean value (20 percent) using the percent crayfish in diet reported from the two Michigan stream studies and the Ohio study. It should be noted that the three dietary composition studies each have a different basis for calculating the percentage of total diet (i.e., mass, volume, and number of prey), which contributes to uncertainty. The remainder of the diet (80 percent) was assumed to be composed of the fish species available in Stout's Creek. The concentration of PCBs in the kingfisher diet was estimated using the general equation below:

$$\mathbf{C_{diet-kingfisher} = (0.80 \text{ x } C_{fish}) + (0.20 \text{ x } C_{crayfish})}$$

where

$C_{diet-kingfisher}$	=	Concentration of PCBs in kingfisher diet
C_{fish}	=	Measured concentration of PCBs in Stout's Creek fish
$C_{crayfish}$	=	Measured/Modeled concentration of PCBs in Stout's Creek crayfish

3.2.2 Kingfisher Home Range and Area Use Factor

When based on the foraging area during the breeding season, the kingfisher home range may fall between 0.64 mile (Ohio streams) and 1.36 miles (Michigan streams) with a mean value of 1.0 stream mile (Brooks and Davis 1987, as cited in EPA 1993). After the breeding season, the foraging area may be smaller with a mean value of 0.24 stream mile (Davis 1980, as cited in EPA 1993). As the toxicity endpoint for the kingfisher is based on reproductive effects, the home range during breeding season (mean value of 1 stream mile) is appropriate. As the home range is 1 stream mile and each of the reaches of Stout's Creek evaluated in the FERA are greater than 1 mile apart, the site utilization factor (or AUF) was determined to be 1; therefore, 100 percent of the kingfisher's diet is considered to be food items within a specific reach.

3.3 DATA COLLECTION AND ANALYSIS

The U.S. Fish and Wildlife Service conducted fish sampling events near the Bennett's Dump site in 1992 and 2003. Viacom conducted two separate fish sampling events near the Bennett's Dump site (see Figure 3) in 1998 and 2004. By October 2000, the majority of the remediation activities were completed at the Bennett's Dump site. Data from the 1992 and 1998 fish sampling events were considered pre-remediation and were not considered appropriate to assess current conditions. Although a fish sampling event was conducted in 2003, sampling was conducted at only one station (Station 2), and only one species of fish (Creek Chub) was collected; this data was not considered adequate for risk assessment purposes. In May 2004, sampling was conducted at Stout's Creek. All three stations were sampled and at least three species of fish (n > 6 for each species) were collected at each station; in addition, crayfish (n = 3) were collected at Station 1. The data from the May 2004 fish sampling events were used to estimate ecological risks associated with post-remediation conditions in Stout's Creek for this FERA. Station locations sampled during the May 2004 fish sampling event are presented below and are shown on Figure 3.

Station	Approximate Distance Downstream from Bennett's Dump Site	Description	Years Sampled
1	1 mile	Hunter Valley Road	1992, 1998, 2004
2	3 miles	Acuff Road	1992, 1998, 2003*, 2004
3	5 miles	Maple Grove Road	1992, 1998, 2004

Note:

* Only Station 2 was sampled during the 2003 sampling event.

Both benthic and pelagic fish species were targeted for collection. Benthic fish live and feed at the bottom of a water body, while pelagic fish live and feed within the water column or at the surface of a water body; the creek chub is considered an omnivorous species. Fish species collected during the May 2004 fish sampling event included:

<u>Benthic</u>

White sucker (*Catostomus commersoni*)

<u>Omnivorous</u>

Creek chub (*Semotilus atromaculatus*)

<u>Pelagic</u>

Green sunfish (*Lepomis cyanellus*)

Longear sunfish (*Lepomis megalotis*)

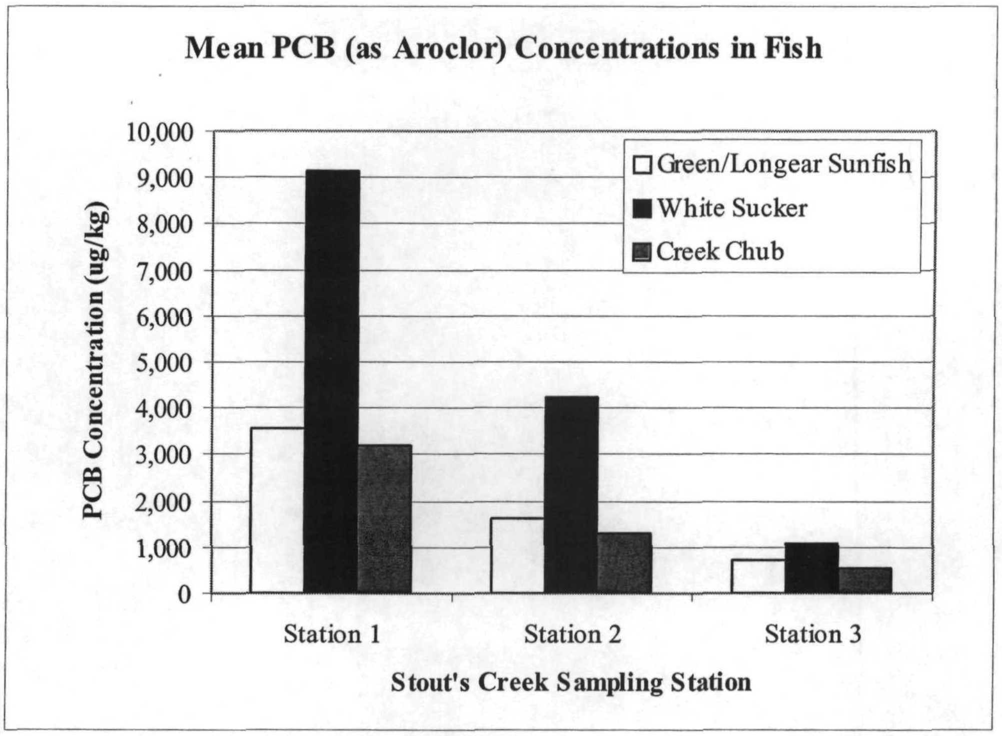
Whole fish and fillet samples were collected during the May 2004 sampling event. Fish samples were analyzed for PCBs (as Aroclors; Enchem Laboratories, Madison, Wisconsin) using gas chromatography methods (EPA SW-846 Method 8082) and percent lipids. In addition, PCB congener analysis was conducted (AXYS Analytical Services, Sidney, British Columbia, Canada) for a subset of the samples using EPA SW-846 Method 1668A. The majority of the samples analyzed were for individual fish; however, some samples were composites of several fish if a single fish did not have sufficient mass for laboratory analysis. Only the data for the whole fish samples were considered in this FERA as piscivorous wildlife consume whole fish; data from fillet samples were not used in the FERA. The number of fish samples collected at each location during the May 2004 sampling event are listed in the table below.

Species	Number of Whole Fish Samples—May 2004		
	Station 1	Station 2	Station 3
Creek chub	7	6	7
Longear sunfish	—	—	7
Green sunfish	7	7	—
White sucker	7	6	7

PCB data for fish collected during the May 2004 sampling event are presented in Table A-1 of Appendix A. Details of the May 2004 sampling activities are presented in the Field Oversight Summary report (Tetra Tech 2004a).

Sunfish were collected at all three stations, with green sunfish collected at Stations 1 and 2 and longear sunfish collected at Station 3. White sucker and creek chub were also collected at all three stations. The mean PCB concentration in green and longear sunfish, white sucker, and creek chub collected at Stations 1, 2, and 3 during the May 2004 fish sampling event is presented on the graph below. A decline in PCB concentrations is noted for all species between the mean sample concentration for fish collected at the station nearest the site (Station 1 at Hunter Valley Road, see Figure 3) and those stations further downstream.

At Station 1, mean PCB concentrations of 3,549, 9,157, and 3,184 micrograms per kilogram (μg/kg) were calculated for green sunfish, white sucker, and creek chub species, respectively. At Station 2, mean PCB concentrations were decreased by approximately 50 percent at concentrations of 1,606, 4,245, and 1,327 μg/kg for green sunfish, white sucker, and creek chub species, respectively. At Station 3, the furthest station downstream, mean PCB concentrations decreased over 50 percent compared to the mean concentrations at Station 2; mean concentrations of 703, 1,091, and 562 μg/kg were calculated for longear sunfish, white sucker, and creek chub, respectively. At all stations sampled, the white sucker (i.e., the benthic species) had the highest reported PCB concentrations.



3.4 CALCULATION OF EXPOSURE POINT CONCENTRATIONS

Once receptors and exposure parameters have been defined and after the data collection phase of the ERA, an estimate of the contaminant concentration in the exposure media (i.e., the EPC) can be calculated. EPA risk assessment guidance recommends that exposure be considered under two scenarios: a reasonable maximum exposure (RME) and a central tendency exposure (CTE) scenario. EPA defines the RME scenario as the maximum exposure that is reasonably expected to occur, while the CTE scenario represent the average exposure expected to occur. The 95th percent upper confidence limit (95UCL) on the arithmetic mean of the data was used as the EPC to evaluate the RME scenario for data sets with four or more samples; the maximum detected concentration was used for data sets with less than four samples. 95UCLs were calculated based on EPA guidance (EPA 2002) using the EPA ProUCL (Version 3.0; EPA 2004b) statistical software program. For data sets with less than four results, the maximum detected concentration was used as the EPC for the RME scenario. The mean fish species PCB concentration was used as the EPC to evaluate the CTE scenario.

3.4.1 Calculation of 95th Percent UCLs for Fish

Two types of RME concentrations were calculated. Species-specific RME PCB concentrations (95UCL or maximum detected concentration) were determined for each fish species at a station. RME concentrations for crayfish were modeled for Stations 2 and 3 (see Section 3.4.2) using the fish species RME concentration (95UCL or maximum detected concentration) for Station 1. Fish and crayfish concentrations used to evaluate the RME scenario are presented in Table 1.

In addition, an RME dietary PCB concentration was calculated (see Table 2). The RME dietary concentration represents the contributions of the RME PCB fish concentration and measured/modeled RME PCB crayfish concentration towards the total PCB concentration in the piscivore diet. The RME dietary PCB concentrations for the mink and kingfisher at a given station were calculated using the general equations describing their dietary compositions as shown below:

$$\mathbf{C_{RME\text{-}mink\ diet} = (0.66 \times Mean\ C_{RME\text{-}fish}) + (0.13 \times C_{RME\text{-}crayfish})}$$

$$\mathbf{C_{RME\text{-}kingfisher\ diet} = (0.80 \times Mean\ C_{RME\text{-}fish}) + (0.20 \times C_{RME\text{-}crayfish})}$$

where,

$C_{RME\text{-}mink\ diet}$	=	RME concentration of PCBs in the diet of mink
$C_{RME\text{-}kingfisher\ diet}$	=	RME concentration of PCBs in the diet of kingfisher
$C_{RME\text{-}fish}$	=	Mean of the RME PCB concentrations for the fish species at a station; each fish species at a station contributes equally to the diet.
$C_{RME\text{-}crayfish}$	=	RME PCB concentrations measured/modeled in crayfish at a station using the RME PCB concentration in fish

TABLE 1

TOTAL PCB CONCENTRATIONS IN FISH AND CRAYFISH—MAY 2004
STATIONS 1 THROUGH 3—STOUT'S CREEK
BENNETT'S DUMP SITE, BLOOMINGTON, INDIANA

	May 2004	
Species	**RME (µg/kg-ww)**	**CTE (µg/kg-ww)**
Station 1		
GS	4,595	3,549
WS	10,705	9,157
CRC	4,075	3,184
CF *	1,100 (max)	877
Station 2		
GS	2,311	1,606
WS	5,544	4,245
CRC	1,810	1,327
CF *	533	389
Station 3		
LS	855	703
WS	1,270	1,091
CRC	653	562
CF *	171	144

TABLE 1 (Continued)

PCB CONCENTRATIONS IN FISH AND CRAYFISH—MAY 2004
STATIONS 1 THROUGH 3—STOUT'S CREEK
BENNETT'S DUMP SITE, BLOOMINGTON, INDIANA

Notes:

Station Fish sample collection station (see Figure 3)

1 Hunter Valley Road
2 Acuff Road
3 Maple Grove Road

*	Crayfish (n = 3) were collected at Station 1. The maximum detected concentration was used as the RME concentration at Station 1; the mean concentration was used as the CTE concentration at Station 1. PCB concentrations in crayfish were modeled for Stations 2 and 3 (see Section 3.4.2)
max	If the data set contained less than four samples, a 95UCL was not calculated. The maximum detected concentration was used as the RME concentration.
95UCL	95-percent upper confidence limit on the arithmetic mean
CF	Crayfish
CRC	Creek chub
CTE	Central tendency exposure; for the CTE scenario, the species mean concentration was calculated.
GS	Green sunfish
LS	Longear sunfish
PCB	Polychlorinated biphenyl
RME	Reasonable maximum exposure; for the RME scenario, the species 95th percent upper confidence limit was calculated. For data sets with less than four samples, the maximum detected concentration was used as the RME value.
µg/kg-ww	Microgram per kilogram-wet weight
WS	White sucker

TABLE 2

TOTAL PCB CONCENTRATIONS IN DIET/DOSE – MAY 2004
BENNETT'S DUMP SITE, BLOOMINGTON, INDIANA

Species	Mink		Kingfisher			
	Diet		Diet		Dose	
Station	RME (µg/kg)	CTE (µg/kg)	RME (µg/kg)	CTE (µg/kg)	RME (µg/kg-day)	CTE (µg/kg-day)
Station 1	4,405.50	3,609.77	5,386.67	4,412.67	2,693.33	2,206.33
Station 2	2,195.59	1,629.59	2,683.93	1,991.76	1,341.97	995.88
Station 3	633.39	537.01	775.00	657.03	387.50	328.51

Notes:

CTE dietary/dose concentration calculated using mean fish and measured/modeled crayfish concentrations.
RME dietary/dose concentration calculated using 95th percent upper confidence limit (or maximum detected) fish and measured/modeled crayfish concentrations.

Station Fish sample collection station (see Figure 3)
1 Hunter Valley Road
2 Acuff Road
3 Maple Grove Road

CTE Central tendency exposure
PCB Polychlorinated biphenyl
RME Reasonable maximum exposure
µg/kg Microgram per kilogram diet
µg/kg-day Microgram per kilogram body weight per day

PCB toxicological studies with dietary exposure have been performed with mink; therefore, the calculated dietary EPCs can be directly compared to the dietary concentrations used in controlled feeding studies to estimate risk. Since PCB toxicological studies with dietary exposure have not been performed with kingfishers, studies using surrogate species were used to determine an effect level and estimate risk. To compare kingfisher exposure to that of the surrogate species, a dose concentration was calculated. After calculating the PCB concentrations in the kingfisher diet, this concentration is then converted to a dose by multiplying the dietary concentration by a kingfisher food ingestion rate (IR) of 0.5-kilogram food per kilogram body weight per day (kg food/kg_{BW}-day) (Alexander 1977, as cited in EPA 1993).

$$\mathbf{Dose_{kingfisher} = C_{RME\text{-}kingfisher\ diet} \times IR}$$

where,

$Dose_{kingfisher}$	=	RME dose of PCBs for kingfisher
$C_{RME\text{-}kingfisher\ diet}$	=	RME concentration of PCBs in the diet of kingfisher
IR	=	Kingfisher food ingestion rate

For comparison, the calculations detailed above were repeated using the mean fish concentrations and measured/modeled crayfish concentrations to estimate the PCB concentration in the piscivore diet for the CTE scenario (i.e., the average concentration). CTE fish and crayfish PCB concentrations are presented in Table 1.

3.4.2 Modeled PCB Uptake by Crayfish

Site-specific crayfish data were collected at Station 1 during the May 2004 sampling event. No crayfish samples were collected at Stations 2 and 3. PCB concentrations in crayfish at Stations 2 and 3 were modeled.

Using the fish and crayfish data for Station 1, species-specific mean fish-to-crayfish PCB ratios were calculated as follows:

- Using the data collected at Station 1, a species-specific mean fish PCB value was calculated (minimum of three samples).

- A mean PCB value was calculated for crayfish at Station 1.

- To determine the species-specific mean fish-to-crayfish PCB ratio, the mean fish PCB concentration for a specific fish species at Station 1 was divided by the mean crayfish PCB concentration at Station 1.

These calculations result in mean fish:crayfish ratios of 4.0 for green sunfish:crayfish, 10.4 for white sucker:crayfish, and 3.6 for creek chub:crayfish for Station 1. To model PCB concentrations in crayfish at Stations 2 and 3, the RME fish concentration (or the CTE fish concentration) at Stations 2 and 3 was divided by the species-specific mean fish-to-crayfish PCB ratio calculated for Station 1. An example calculation is shown below:

> Station 2: Three species of fish were collected at Station 2: green sunfish, white sucker, and creek chub. To model a RME PCB crayfish concentration for Station 2, the RME PCB concentrations for green sunfish (2,311 μg/kg) was divided by the species-specific mean green sunfish-crayfish ratio (4.0), which resulted in a modeled crayfish concentration of 578 μg/kg. This calculation was repeated using the white sucker data and creek chub data at Station 2, which resulted in a total of three modeled crayfish concentrations at Station 2; the three modeled crayfish concentrations were averaged to determine the final crayfish concentration for use in the risk calculations.

The mean fish-crayfish ratios and modeled PCB crayfish concentrations are presented in Table A-2 of Appendix A.

4.0 ECOLOGICAL EFFECTS ASSESSMENT

The objective of the ecological effects assessment is to present the measures of effect that were evaluated in the FERA. The effects assessment determines the potential for PCBs to adversely affect the assessment endpoint identified for the aquatic ecosystem within Stout's Creek (in proximity to the Bennett's Dump site). Both field and laboratory studies are available to describe the effects of PCBs on wildlife. Studies are also available that identify toxicity reference values (TRV). TRVs represent a threshold effect-level of a chemical. An exceedance of the TRV (or threshold level) indicates adverse effects may occur, but does not, in itself, indicate that an adverse effect has occurred. Concentrations (or doses) below TRVs are not expected to result in an adverse effect; however, the conclusion is subject to uncertainty including interspecies differences in sensitivity, differences in contaminant bioavailability, and differences between effects observed in a laboratory setting compared to those encountered in the field.

TRVs can be statistically determined from study data and represent whether the severity or occurrence of an effect in a treated group is statistically greater than in an unexposed group. TRVs can be based on no-observed-adverse-effect concentrations (NOAEC), no-observed-adverse-effect levels (NOAEL), lowest-observed-adverse-effect concentrations (LOAEC), and lowest-observed-adverse-effect levels (LOAEL).

NOAECs/LOAECs represent threshold values expressed as a concentration in food for dietary exposures. TRVs based on NOAECs and LOAECs were identified for the mink. NOAELs/LOAELs represent daily dose levels that are normalized for body weight. Dose levels can be used to compare toxicity data across species. As no dietary studies specific to the kingfisher were identified, TRVs based on NOAELs and LOAELs in surrogate species (chicken and dove) were identified for the kingfisher.

NOAELs/NOAECs represent the highest concentration (or dose) that did not result in adverse effects in the test animal. LOAELs/LOAECs represent the lowest concentration (or dose) associated with an adverse effect in the test animal.

A "no-effect" level or "low-effect" level may be selected as a TRV by interpolating an appropriate value from a dose-response curve or exposure-response curve derived from multiple studies. In addition to the NOAEL- and LOAEL-based TRVs identified for the kingfisher, no-effect and low-effect TRVs were identified by EPA. A TRV evaluation was conducted by EPA for a similar site (Neal's Landfill; EPA 2003) and is included in the Attachment.

The tables included in the Attachment present the FERA TRVs considered for the mink and kingfisher. The TRVs selected for the FERA to evaluate PCB exposures to mink and kingfisher through the diet are presented in the following table.

Species	Toxicity Reference Values					
	NOAEC ($\mu g/kg_{diet}$)	**LOAEC ($\mu g/kg_{diet}$)**	**NOAEL ($\mu g/kg_{bw}$-day)**	**LOAEL ($\mu g/kg_{bw}$-day)**	**No Effect ($\mu g/kg_{bw}$-day)**	**Low Effect ($\mu g/kg_{bw}$-day)**
Mink	500	600	--	--	--	--
Kingfisher	--	--	110	1,120	400	500

Notes:

For mink, TRVs were based on reproductive effects in mink.
For kingfishers, no effect- and low effect-based TRVs were extrapolated from reproductive effects (egg hatchability) in chickens. NOAEL- and LOAEL-based TRVs were extrapolated from behavioral effects (i.e., parental inattentiveness) in doves.

LOAEC; LOAEL	Lowest-observed-adverse-effect concentration; lowest-observed-adverse-effect level
NOAEC; NOAEL	No-observed-adverse-effect concentration; no-observed-adverse-effect level
$\mu g/kg_{diet}$	Microgram per kilogram in the diet
$\mu g/kg_{bw}$-day	Microgram per kilogram-bodyweight per day
TRV	Toxicity reference value

4.1 MINK TOXICITY REFERENCE VALUES

The TRVs selected to evaluate PCBs in the mink diet for this FERA are 500 μg/kg based on a NOAEC and 600 μg/kg based on a LOAEC. These values are interpolated from an exposure-response plot for the results of three mink feeding studies in which Aroclor 1254 was added to the diet (Aulerich and Ringer 1977; Kihiström and others 1992; Wren and others 1987a, b; as cited in EPA 2005). The reader is directed to Chapman 2003 (as cited in EPA 2005) for a derivation of the TRV. Critical toxicological endpoints noted in the studies were live kit production and kit body weight. Studies were conducted over a single breeding season. In studies conducted over two breeding seasons with Clophen A50 (a PCB mixture similar to Aroclors)-contaminated prey reported increased adverse effects compared to studies conducted over just one breeding season. No mink feeding studies using Aroclors that continued beyond one breeding season were identified. Therefore, the single-breeding season TRVs were adjusted for continuous exposure over multiple breeding seasons/generations. The TRVs were adjusted by multiplying by the mean ratio of the interpolated TRV for the two breeding season/generation-contaminated prey studies to their respective single-breeding season TRV (yielding a mean ratio of 0.52 for live kit production, kit body weight, and kit survival endpoints) (EPA 2005).

ERAs for the Great Lakes Initiative (GLI), Housatonic River, and Hudson River were reviewed during the TRV selection process. The LOAEC-based TRV identified for the GLI (2,000 μg/kg) was much higher than the interpolated value selected for the FERA (600 μg/kg). Exposure at a concentration equivalent to the GLI LOAEC-based TRV resulted in a complete suppression of the reproductive processes, which is not an acceptable endpoint; therefore, the GLI LOAEC-based TRV was not considered appropriate.

PCBs (as Aroclor 1260) were released to the Housatonic River (EPA 2005). Aroclor 1260-based TRVs of 1,600 μg/kg (NOAEC basis) and 3,700 μg/kg (LOAEC basis) were determined from a mink feeding study. Data from split samples collected by Tetra Tech during the May 2004 sampling event indicate that Aroclor 1248 was the only Aroclor present in detectable levels in fish tissue at Stout's Creek (Tetra Tech 2004b). Aroclor 1260 is less toxic to mammals than other Aroclors (Tillit and others 1992, as cited in EPA 2005); therefore, the Aroclor 1260-based TRVs were not considered sufficiently protective for exposures at the Bennett's Dump site.

The Hudson River ERA used a LOAEC-based TRV of 250 μg/kg based on the Restum and others (1998) study conducted over two breeding seasons and two generations. The Hudson River TRV is lower than the LOAEC-TRV selected for the FERA (600 μg/kg). However, included in the diet were field-contaminated carp from Saginaw Bay, Michigan. It is unknown whether co-contaminants may have interacted with the PCBs to produce additive or synergistic effects (2-breeding season and 2-generation exposure study), but interpretation of the results on an Aroclor basis is complicated by possible additive or multiplicative effects of co-contaminants other than PCBs.

As is shown in the Attachment, most of the NOAEC-based TRVs identified were lower than the value selected for use in the FERA (500 μg/kg). This value is considered sufficiently protective as the values selected for use in the FERA were interpolated from an exposure-response curve, which has a steep slope between the "no effects" and "severe effects" endpoints. In individual experiments, there is often a wide dose spacing; it is not unusual to have doses increasing by an order of magnitude. With wide dose spacing, it may be possible to miss the dose at which effects begin to be observed (i.e., the threshold-effects level).

4.2 KINGFISHER TOXICITY REFERENCE VALUES

An evaluation of avian PCB TRVs (based on dose) performed by EPA (EPA 2005) are presented in the Attachment. Two suitable sets of values were identified to select a dose-related TRV based on TRVs used in the Neal's Landfill FERA (EPA 2005). As there was an appreciable difference between the sets of values, both sets of TRVs were retained for this FERA. The kingfisher dietary TRVs selected for the FERA are 400 $\mu g/kg_{BW}$-day for a no-effect and 500 $\mu g/kg_{BW}$-day for a low-effect (EPA 2005), and a NOAEL-based TRV of 110 $\mu g/kg_{BW}$-day and LOAEL-based TRV of 1,120 $\mu g/kg_{BW}$-day (based on the Fox River and Green Bay ERAs, as cited in EPA 2005).

The no-effect and low-effect TRVs (400 $\mu g/kg_{BW}$-day and 500 $\mu g/kg_{BW}$-day, respectively) are interpolated from a dose-response plot for the results of three chicken feeding studies conducted with Aroclor 1248 in the diet (Cecil and others 1974; Lillie and others 1974 and 1975; and Scott 1977; as cited in EPA 2005). The reader is directed to Chapman 2003 (as cited in EPA 2005) for a derivation of the TRV. Critical toxicological endpoints noted in the studies were hatchability. Dietary PCB concentrations in the chicken studies were converted to a dose (body weight-normalized concentration) by multiplying the dietary concentrations by the study-specific food ingestion rate or by a default leghorn hen food ingestion rate of 0.067 kg feed/kg_{BW}-day (Medway and Kare 1959, as cited in EPA 2005) if no food ingestion rate was available.

The low-effect-based TRV of 500 $\mu g/kg_{BW}$-day is between the avian (pheasant) LOAEL-based TRV (600 $\mu g/kg_{BW}$-day) for PCBs (as Aroclor 1254) used in the GLI ERA (Dahlgren and others1972, as cited in EPA 2005) and the LOAEL-based TRV identified for the Sheboygan River and Harbor ERA (400 $\mu g/kg_{BW}$-day), which is based on exposure to field-contaminated feed (Summer and others 1996a, b, as cited in EPA 2005). The NOAEL-based TRVs identified for both of these ERAs are appreciably lower than the no-effect-based TRV of 400 $\mu g/kg_{BW}$-day selected for the FERA. The FERA value is considered sufficiently protective as the values selected for use in the FERA were interpolated from an exposure-response curve. Interpolation from an exposure-response curve aids in the identification of a threshold-effect level, which can be missed in the dose-spacing in laboratory studies.

The NOAEL-based TRV (110 $\mu g/kg_{BW}$-day) and LOAEL-based TRV (1,120 $\mu g/kg_{BW}$-day) are based on impairment of courtship and nesting behaviors in doves (Peakall and Peakall 1973; Tori and Peterle 1983; as cited in EPA 2003). PCB-exposed doves were inattentive parents, which contributed to a decreased survival of offspring. Birds with impaired courtship behavior are less likely to successfully mate, which affects reproduction. Even though the LOAEL-based TRV is two times higher than the interpolated low-effect-based TRV (500 $\mu g/kg_{BW}$-day) selected for the FERA, the NOAEL-based TRV is appreciably lower than the no-effect-based interpolated TRV (400 $\mu g/kg_{BW}$-day).

The remaining TRVs presented in the Attachment are very high values identified in the Hudson River ERA. These values are up to 10 times greater than the TRVs identified by GLI using the same study due to a difference in toxicological endpoints. Egg production was the critical effect for the Hudson River ERA, while hatchability was identified as the critical effect for the GLI ERA. Also, the Hudson River ERA did not incorporate any modifying factors, while the GLI ERA adjusted the TRVs by a factor of 3. A decline in egg production was not identified as a critical effect for the FERA because studies with chicken, a sensitive species to PCBs, do not exhibit a clear dose response relationship between PCB exposure and a change in egg production (Chapman 2003, as cited in EPA 2005).

4.3 CALCULATION OF HAZARD QUOTIENTS

Potential risks to piscivorous receptors were assessed by a chemical-specific comparison of maximum estimated concentrations (mink) or daily doses (kingfisher). This comparison is expressed as a hazard quotient (HQ). HQs were calculated for each sampling station representing a "reach" of Stout's Creek.

For the mink, dietary concentations were compared to the TRV, and the HQ is expressed as:

$$\mathbf{HQ = C_{diet} / TRV}$$

where,

HQ	=	Hazard quotient for mink
C_{diet}	=	Concentration of PCBs in the mink diet
TRV	=	Toxicity reference value for mink

For the kingfisher, the HQ was calculated in a similar fashion; however, a PCB concentration expressed as a daily dose was substituted for the C_{diet} term in the equation.

A calculated HQ exceeding 1 (i.e., HQ > 1) may indicate that the receptor is at risk of an adverse effect from exposure to PCBs in the diet. HQs were calculated for the RME and CTE scenarios (see Tables B-1 and B-2 of Appendix B).

If a NOAEC/NOAEL/no-effect-based TRV is used to calculate the HQ, then an HQ equal to or less than 1 indicates that adverse effects would not be expected. For LOAEC/LOAEL/low-effect-based TRVs, an HQ of 1 or more indicates that adverse effects are expected (i.e, the concentration in the diet/dose is equal to or greater than a concentration associated with adverse effects). An area of uncertainty exists between the concentration associated with no-adverse-effects and the concentration known to produce adverse effects. Within that area of uncertainty is the threshold effect-level.

4.4 PERCENT ALLOWABLE CONCENTRATION

To address uncertainty in the exposure assumptions for the mink and kingfisher, an additional approach is used as part of the risk characterization. A percent allowable consumption (PAC) for each station was calculated, which represents the percent of the diet an animal can consume within a station area without exceeding the TRV. This approach is modified from the percent allowable daily intake (PADI) approach of Giesy and others (1994). Giesy and others (1994) gives the following equation for PADI:

$$\mathbf{PADI = ((NOAED / C_{fish}) / CR) \times 100} \qquad [1]$$

where,

NOAED	=	No-observed-adverse-effect dose
C_{fish}	=	PCB concentration in diet (fish and crayfish)
CR	=	Food consumption rate

and if, $\mathbf{NOAED = dietary\ NOAEC \times CR}$

then equation 1 simplifies to:

$$\mathbf{PAC = (dietary\ NOAEC / C_{diet}) \times 100} \qquad [2]$$

PADI becomes PAC as the daily food consumption term no longer appears in the equation.

Since, $\mathbf{HQ = C_{contaminant} / TRV}$ or, in this case $\mathbf{HQ = C_{diet} / NOAEC}$

then $$\mathbf{PAC = (1 / HQ) \times 100} \quad [3]$$

PADI/PAC estimate the percent of an animal's diet that can be consumed from a contaminated source without exceeding the threshold for toxic effects. Equation 3 estimates the percent diet an animal can consume from a contaminated source without exceeding toxic levels assuming that the remainder of the diet has zero contamination. As PCBs are ubiquitous contaminants in the environment, it may be unreasonable to assume that PCBs will not be present in other components of the diet.

Equation 3 above can be modified to account for the contribution of ambient or background levels of PCBs by subtracting out the HQ contributed by background or off-site concentrations of PCBs ($HQ_{off\text{-}site}$). Equation 3 then becomes:

$$\mathbf{PAC = ((1 - HQ_{off\text{-}site}) / HQ_{site}) \times 100} \quad [4]$$

No background or reference area sample data is available for use in the FERA from Stout's Creek. Therefore, Equation 3 was used to calculate the PAC for this FERA. PACs based on RME and CTE PCB concentrations in the diet (or dose) based on data collected during the May 2004 sampling event, are presented in Tables B-3 and B-4 of Appendix B.

5.0 RISK CHARACTERIZATION RESULTS

This section summarizes the findings of the risk calculations to form conclusions about potential risks posed to the assessment endpoints (piscivorous mammals and birds) identified for the Bennett's Dump site study areas (i.e., Stout's Creek) in the problem formulation phase.

Risk characterization is the integration of exposure and effects data to determine the likelihood of adverse effects. For the FERA, the HQ (or toxicity quotient) method was used to characterize risk from PCBs. In addition, a PAC was calculated.

HQs were calculated for both the RME and CTE scenarios. For the RME scenario, risks were estimated using RME PCB concentrations in fish, while for the CTE scenario, risks were estimated using the CTE PCB concentration in fish. For all of the stations, the CTE HQs were similar (but slightly lower in some instances) to those calculated for the RME scenario (see Tables 3 and 4).

5.1 MINK HAZARD QUOTIENTS: STATIONS 1 THROUGH 3

HQs for Stations 1 through 3 are discussed in the following sections.

5.1.1 Station 1

Fish samples were collected at Station 1 during the May 2004 fish sampling event. Station 1 is located at Hunter Valley Road, approximately 1 mile downstream from the Bennett's Dump site (see Figure 3). Both the no-effect-based and low-effect-based mink HQs were greater than 1 for both the RME and CTE scenario. For the RME scenario, mink HQs ranged from 7 (low-effect-based) to 9 (no-effect-based) (see Table 3). For the CTE scenario, mink HQs ranged from 6 (low-effect-based) to 7 (no-effect-based) (see Table 3). As the low-effect-based mink HQs exceed 1, the potential for adverse effects exists for mink with a home range within the Station 1 reach.

5.1.2 Station 2

Mink HQs estimated for Station 2 are discussed below. Station 2 is located at Acuff Road, approximately 3 miles downstream from the Bennett's Dump site (see Figure 3).

Mink HQs decreased downstream at Station 2 as compared to the mink HQs estimated for Station 1 (see Table 3). For the RME scenario, both the low-effect-based and no-effect-based mink HQs were estimated at 4, which exceeds the threshold value of 1. For the RME scenario, as the low-effect-based

TABLE 3

MINK HAZARD QUOTIENTS FOR THE RME AND CTE SCENARIOS BENNETT'S DUMP SITE, BLOOMINGTON, INDIANA

Station	May 2004			
	RME Scenario		CTE Scenario	
	No Effect	Low Effect	No Effect	Low Effect
Station 1	9	7	7	6
Station 2	4	4	3	3
Station 3	1	1	1	0.9

Notes:

Hazard quotients are shown to one significant digit. See also Appendix B.

Station	Fish sample collection station (see Figure 3) 1 Hunter Valley Road 2 Acuff Road 3 Maple Grove Road
CTE	Central tendency exposure; HQs were calculated using the mean PCB concentration in fish.
HQ	Hazard quotient; where HQ = RME or CTE concentration of PCBs in mink diet/TRV. For a no-effect-based HQ, an HQ less than or equal to 1 indicates that no adverse effect would be expected. For a low-effect-based HQ, an HQ greater than or equal to 1 indicates the potential for adverse effects. An area of uncertainty exists between the concentration associated with no adverse effects and the concentration known to produce adverse effects.
RME	Reasonable maximum exposure; HQs were calculated using the 95th percent upper confidence limit for PCB concentration in fish.
PCB	Polychlorinated biphenyl
TRV	Toxicity reference value; no-effect-based-TRV = 500 µg/kg and low-effect-based TRV = 600 µg/kg.
µg/kg	Microgram per kilogram

TABLE 4

KINGFISHER HAZARD QUOTIENTS FOR THE RME AND CTE SCENARIOS
BENNETT'S DUMP SITE, BLOOMINGTON, INDIANA

Station	May 2004							
	RME Scenario				CTE Scenario			
	No Effect	NOAEL	Low Effect	LOAEL	No Effect	NOAEL	Low Effect	LOAEL
Station 1	7	24	5	2	6	20	4	2
Station 2	3	12	3	1	2	9	2	0.9
Station 3	1	4	0.8	0.4	0.8	3	0.7	0.3

Notes:

Hazard quotients are shown to one significant digit. See also Appendix B.

Station	Fish sample collection station (see Figure 3) 1 Hunter Valley Road 2 Acuff Road 3 Maple Grove Road
CTE	Central tendency exposure; HQs were calculated using the mean PCB concentration in fish.
HQ	Hazard quotient; where HQ = RME or CTE concentration of PCBs in kingfisher diet (on a dose-basis) / TRV. For a no-effect-/NOAEL-based HQ, an HQ less than or equal to 1 indicates that no adverse effect would be expected. For a low-effect/LOAEL-based HQ, an HQ greater than or equal to 1 indicates the potential for adverse effects. An area of uncertainty exists between the concentration associated with no adverse effects and the concentration known to produce adverse effects.
RME	Reasonable maximum exposure; HQs were calculated using the RME PCB concentration (95th percent upper confidence limit or maximum detected concentration) in fish.
LOAEL	Lowest-observed-adverse-effect level
NOAEL	No-observed-adverse-effect level
PCB	Polychlorinated biphenyl
TRV	Toxicity reference value; no-effect-based TRV = 400 µg/kg-day and low-effect-based TRV = 500 µg/kg-day; NOAEL-based TRV = 110 µg/kg-day and LOAEL-based TRV = 1,120 µg/kg-day.
µg/kg-day	Microgram per kilogram per day

mink HQ exceeds 1, the potential for adverse effects exists for mink with a home range within the Station 2 reach.

For the CTE scenario, both the low-effect-based and no-effect-based mink HQs were estimated at 3, which exceeds the threshold value of 1. For the CTE scenario, as the low-effect-based mink HQ exceeds 1, the potential for adverse effects exists for mink with a home range within the Station 2 reach.

5.1.3 Station 3

Mink HQs estimated for Station 3 are discussed below. Station 3 is located at Maple Grove Road, approximately 5 miles downstream from the Bennett's Dump site (see Figure 3).

Mink HQs decreased downstream at Station 3 as compared to the mink HQs estimated for Stations 1 and 2 (see Table 3). For the RME scenario, both the low-effect- and no-effect-based mink HQs were estimated at 1, which is equivalent to the threshold value. For the RME scenario, the low-effect-based mink HQ is equivalent to 1 for a mink within the Station 3 reach, which indicates that exposure is equal to levels shown to cause an adverse effect.

For the CTE scenario, the no-effect-based mink HQs was estimated at 1, which is equivalent to the threshold value of 1; the low-effect-based mink HQ was estimated at 0.9, which is below the threshold value of 1 (see Table 4). For the CTE scenario, as the no-effect-based mink HQ is equivalent to 1 and the low-effect-based mink HQ is less than 1, the potential for adverse effects is unlikely for mink with a home range within the Station 3 reach.

5.2 KINGFISHER HAZARD QUOTIENTS: STATIONS 1 THROUGH 3

Kingfisher HQs for Stations 1 through 3 are discussed in the following sections.

5.2.1 Station 1

Fish samples were collected at Station 1 during the May 2004 fish sampling event. Station 1 is located at Hunter Valley Road, approximately 1 mile downstream from the Bennett's Dump site (see Figure 3).

Both the no-effect-based and low-effect-based kingfisher HQs were greater than 1 for both the RME and CTE scenarios. For the RME scenario, kingfisher HQs calculated using low-effect-based or LOAEL-based TRVs ranged from 2 to 5 (see Table 4). RME HQs calculated using no-effect-based or NOAEL-based TRVs ranged from 7 to 24. CTE HQs were lower than the RME results; however, all kingfisher CTE HQs exceeded the threshold of 1 (ranging from 2 to 20) (see Table 4). As both the low-effect-/LOAEL-based and no-effect-/NOAEL-based kingfisher HQs exceed the threshold of 1, a kingfisher with a home range within the Station 1 reach has an increased risk for adverse effects.

5.2.2 Station 2

Kingfisher HQs estimated for Station 2 are discussed below. Station 2 is located at Acuff Road, approximately 3 miles downstream from the Bennett's Dump site (see Figure 3).

Kingfisher HQs decreased downstream at Station 2 as compared to the kingfisher HQs estimated for Station 1 (see Table 4). For the RME scenario, both the no-effect- and NOAEL-based HQs exceed the threshold of 1 (HQs of 3 and 12, respectively; see Table 4). The low-effect-based HQ was estimated at 3, which exceeds the threshold value of 1 and the LOAEL-based HQ was estimated at 1, which indicates the concentration is equal to levels shown to cause adverse effects. For the RME scenario, as the LOAEL-based kingfisher HQ was equivalent to the threshold value of 1 and as the low-effect-based HQ exceeds 1 (HQ = 3), the potential for adverse effects is exists for the kingfisher with a home range within the Station 2 reach.

For the CTE scenario, both the no-effect- and NOAEL-based HQs exceed the threshold of 1 (HQs of 2 and 9, respectively; see Table 4). The low-effect-HQ was estimated at 2, which exceeds the threshold value of 1; the LOAEL-based HQ is estimated at 0.9, which is less than the threshold value of 1 (see Table 4). For the CTE scenario, as the LOAEL-based HQ is less than 1 and the low-effect-based HQ exceeds the threshold of 1 (HQ = 2), the potential for adverse effects exists for the kingfisher with a home range within the Station 2 reach.

The NOAEL-based HQs were greater than 1 (9 for the CTE scenario and 12 for the RME scenario). The NOAEL-based HQs are based on a TRV for parental inattentiveness.

5.2.3 Station 3

Kingfisher HQs estimated for Station 3 are discussed below. Station 3 is located at Maple Grove Road, approximately 5 miles downstream from the Bennett's Dump site (see Figure 3).

Kingfisher HQs decreased downstream at Station 3 as compared to the kingfisher HQs estimated for Stations 1 and 2 (see Table 4). For the RME scenario, the no-effect-based HQs was equivalent to the threshold of 1; the LOAEL-based HQ was less than 1 (HQ = 0.4). The NOAEL-based HQ was estimated at 4, which is greater than the threshold of 1. The low-effect-based HQ was estimated at 0.8, which is below the threshold value of 1 and indicates the exposure concentration may be below the concentration shown to cause adverse effects. For the RME scenario, as the no-effect-based HQ is equivalent to the threshold value of 1 and as the low-effect- and LOAEL-based HQs were below 1, the potential for adverse effects may be low for the kingfisher with a home range within the Station 3 reach.

For the CTE scenario, the low-effect- and LOAEL-based HQs were estimated at 0.7 and 0.3, respectively, which are below the threshold value of 1. The no-effect- and NOAEL-based HQs ranged from 0.8 to 3 (see Table 4). For the CTE scenario, as the no-effect-based HQ and the low-effect- and LOAEL-based HQs are less than 1, the potential for adverse effects may be low for the kingfisher with a home range within the Station 3 reach.

The NOAEL-based HQs were greater than 1 (3 for the CTE scenario and 4 for the RME scenario). The NOAEL-based HQs are based on a TRV for parental inattentiveness.

5.3 PERCENT ALLOWABLE CONSUMPTION

The PAC results for Stations 1 through 3 discussed in the following sections and are summarized in Table 5 for the RME scenario and Table 6 for the CTE scenario.

5.3.1 Station 1: Percent Allowable Consumption

The Station 1 PAC assumes that the only exposure mink or kingfisher have to PCBs released from the Bennett's Dump site is through consumption of Stout's Creek fish and crayfish. The PAC values therefore, represent the percentage of the mink or kingfisher diet taken from Station 1 that would result in a NOAEC- or LOAEC-based HQ of 1. A NOAEC-based HQ of 1 is not associated with adverse effects; however, a LOAEC-based HQ of 1 indicates that the exposure is equivalent to the lowest concentration associated with potential adverse effects.

The Station 1 NOAEC-based PAC for mink is estimated at 11 percent for the RME scenario and 14 percent for the CTE scenario (see Tables 5 and 6). The mink LOAEC-based PAC ranges from 14 percent for the RME scenario to 17 percent for the CTE scenario (see Tables 5 and 6). The results indicate that, to stay within no-effect dietary concentrations, mink should forage along the Station 1 reach for no more than approximately 10 percent of the total diet, and that potentially adverse effects are possible if greater than approximately 20 percent of the total diet comes from the Station 1 reach.

Station 1 is located approximately 1 mile downstream of Bennett's Dump site. It is described as being 8 to 10 feet wide at bank full with an average depth of 1 to 2 feet. Some runs and pools may be up to 3 feet deep. Evidence of abundant wildlife was noted (Tetra Tech 2004a), including deer, opossum, raccoons, beavers, rabbits, moles, and mice. Numerous bird species were observed during the May 2004 sampling event, including herons, hawks, turkey vultures, crows, and songbirds.

TABLE 5

PERCENT ALLOWABLE CONSUMPTION FOR THE RME SCENARIO
BENNETT'S DUMP SITE, BLOOMINGTON, INDIANA

Station	May 2004					
	Mink		Kingfisher			
	NOAEC	LOAEC	No Effect	NOAEL	Low Effect	LOAEL
Station 1	11	14	15	4	19	42
Station 2	23	27	30	8	37	83
Station 3	79	95	103	28	129	289

Notes:

PAC represents the percent diet an animal can consume within a station area reach without exceeding the TRV, where PAC = (1 / HQ) x 100.

Station	Fish sample collection station (see Figure 3) 1 Hunter Valley Road 2 Acuff Road 3 Maple Grove Road
HQ	Hazard quotient
LOAEC	Lowest-observed-adverse-effect level
LOAEL	Lowest-observed-adverse-effect level
NOAEC	No-observed-adverse-effect level
NOAEL	No-observed-adverse-effect level
PAC	Percent allowable consumption
PCB	Polychlorinated biphenyl
RME	Reasonable maximum exposure; HQs were calculated using the 95th upper confidence limit for PCB concentration in fish.
TRV	Toxicity reference value

TABLE 6

PERCENT ALLOWABLE CONSUMPTION FOR THE CTE SCENARIO
BENNETT'S DUMP SITE, BLOOMINGTON, INDIANA

Station	May 2004					
	Mink		Kingfisher			
	NOAEC	LOAEC	No Effect	NOAEL	Low Effect	LOAEL
Station 1	14	17	18	5	23	51
Station 2	31	37	40	11	50	112
Station 3	93	112	122	33	152	341

Notes:

PAC represents the percent diet an animal can consume within a station area reach without exceeding the TRV, where PAC = (1 / HQ) x 100.

Station	Fish sample collection station (see Figure 3) 1 Hunter Valley Road 2 Acuff Road 3 Maple Grove Road
CTE	Central tendency exposure; HQs were calculated using mean PCB concentration in fish.
HQ	Hazard quotient
LOAEC	Lowest-observed-adverse-effect level
LOAEL	Lowest-observed-adverse-effect level
NOAEC	No-observed-adverse-effect level
NOAEL	No-observed-adverse-effect level
PAC	Percent allowable consumption
PCB	Polychlorinated biphenyl
TRV	Toxicity reference value

Mink PACs account for 79 percent of the mink diet as up to 21 percent of the diet was assumed to have no PCB contamination. To calculate the amount of aquatic prey that can be consumed from Station 1 and result in a mink LOAEC-based HQ of 1, the LOAEC-based PAC was multiplied by 79 percent. This results in an adjusted mink LOAEC-based PAC of 11 (RME scenario LOAEC-based PAC of 14 percent x 0.79) to 14 percent (CTE scenario LOAEC-based PAC of 17 percent x 0.79). If a typical mink is assumed to have a food IR of 160 grams per day and a body weight of 1 kilogram (Bleavins and Aulerich 1981, as cited in EPA 1993), this is equivalent to 18 to 22 grams of aquatic prey (fish+crayfish) from Station 1, with fish comprising 15 to 18 grams and crayfish comprising approximately 3 to 4 grams of the Station 1 diet (diet = 66 percent fish + 13 percent crayfish)[1]. The mean weight for fish caught at Station 1 is approximately 38 grams. Mink PACs of 15 to 18 grams of fish are equivalent to approximately 0.4 fish/day to 0.5 fish/day (or approximately 2 fish/5 days to 1 fish/2 days, respectively), assuming there is no other exposure to PCBs at the Bennett's Dump site with the exception of the relatively small contribution from PCBs in crayfish.

The kingfisher has a diet that is composed of 100 percent aquatic prey (fish+crayfish); therefore, the PAC directly represents the amount of fish and crayfish from the station-reach under consideration. Kingfisher no effect-/NOAEL-based PACs for the aquatic diet range from 4 to 15 percent for the RME scenario and 5 to 18 percent for the CTE scenario (see Tables 5 and 6, respectively). The low-effect-/LOAEL-based PACs range from 19 to 42 percent for the RME scenario and 23 to 51 percent for the CTE scenario (see Tables 5 and 6, respectively). Adjusting the low-effect/LOAEL-based aquatic diet PACs to reflect that portion of the diet comprised of fish results in adjusted PACs of 15 to 34 percent (80 percent of the aquatic diet is fish), respectively. To calculate the amount of fish from Station 1 required to reach concentrations equivalent to the low-effect/LOAEL-based concentrations, the adjusted low-effect-/LOAEL-based PACs were applied to a kingfisher food IR of

[1] For example, 160 grams/day x 11 percent = 17.6 (or 18, if rounded) grams/ aquatic prey-day. The mink aquatic diet is assumed to be composed of 66 percent fish and 13 percent crayfish; therefore, the amount of fish that may be consumed per day is calculated as follows: 18 grams/aquatic prey-day x (66 percent fish in aquatic diet / 79 percent aquatic diet) = 15 grams fish/day.

0.5 gram per body weight per day (EPA 1993) and a kingfisher body weight of 147 grams, which resulted in an estimate of approximately 11 to 25 grams of fish per day[2]. This is equivalent to 0.3 to 0.7 fish per day (or 1 fish/4 days to 2 fish/3 days with rounding).

5.3.2 Stations 2 and 3: Percent Allowable Consumption

As the PCB concentrations in fish were lower at Stations 2 and 3 that at Station 1, the estimated PACs for Stations 2 and 3 were higher than that estimated at Station 1. That is, more fish may be consumed in the Station 2 and 3 reaches than at the Station 1 reach (see Tables 5 and 6).

For the mink, the Station 2 NOAEC- and LOAEC-based PACs (23 and 27 percent, respectively) were approximately double those estimated for Station 1 for the RME and CTE scenarios (see Tables 5 and 6). The Station 3 NOAEC- and LOAEC-based PACs were approximately 7 times higher than those reported for Station 1 for the RME and CTE scenarios (see Tables 5 and 6). Using the same approach described above for Station 1 to estimate the amount of fish that could be consumed in the Station 2 reach to equal a LOAEC-based HQ of 1, a fish consumption rate of 28 grams fish/day was estimated for the RME scenario (or approximately 7 fish over 10 days) at Station 2; a fish consumption rate of 39 grams fish/day (slightly greater than 1 fish per day) was estimated for the CTE scenario at Station 2. At Station 3, the LOAEC-based PACs were 95 percent for the RME scenario and greater than 100 percent for the CTE scenario, resulting in fish consumption rates of 100 grams fish/day (RME; less than 3 fish per day) and 118 grams fish/day (CTE; approximately 3 fish per day).

For the kingfisher, the Station 2 no-effect- and NOAEL-based PACs for the RME scenario (30 and 8 percent, respectively) and the CTE scenario (40 and 11 percent, respectively) were approximately double those estimated for Station 1 as were the low-effect- and LOAEL-based PACs (see Tables 5 and 6). The low-effect-/LOAEC-based PACs ranged from 37 to 83 percent for the RME scenario to 50 to 112 percent for the CTE scenario. For the RME scenario, the resulting fish consumption rate is

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[2] For example, 147 grams-bw x 0.5 grams food/bw-day = 73.5 grams food/day. Assuming a 100 percent aquatic diet with 80 percent fish and 20 percent crayfish, 73.5 grams aquatic prey/day x (19 percent x 0.8 fraction fish in diet) = 11 grams fish/day.

22 grams fish/day (or 3 fish/5 days with rounding) to 49 grams fish/day (greater than 1 average weight fish per day) ; for the CTE scenario, the resulting fish consumption rate is 29 grams fish/day (or 2 fish/3 days with rounding) to 66 grams fish/day (or approximately 2 fish/day). At Station 3, the kingfisher low effect- and LOAEC-based PACs for the aquatic diet were greater than 100 percent for both the RME and CTE scenarios.

6.0 UNCERTAINTY ANALYSIS

The purpose of the uncertainty analysis is to (1) provide risk managers with a summary of those factors that significantly influence the risk results and (2) assess the contribution of these factors to the under- or overestimation of risk.

Virtually every step in the ERA process requires numerous assumptions, all of which contribute to uncertainty in the risk evaluation. In the absence of empirical or site-specific data, assumptions are developed based on best estimates of data quality, exposure parameters, and dose-relationships.

6.1 FISH AND CRAYFISH CONTAMINANT DATA

Site-specific PCB concentrations in fish caught at three sampling stations on Stout's Creek provided the data for the FERA. Sources of uncertainty associated with the site-specific data include the movement of fish between sampling stations, differences in species accumulation, and changes to the dietary composition.

Both RME and CTE risk estimates were calculated using RME and CTE fish concentrations, respectively. A measure of data variability is the coefficient of variation (CV), which is the standard deviation divided by the mean. For the majority of the fish species data sets, the CV was less than or equal to 0.5 indicating a relatively low sample variability within a fish species collected at a given station. The CV for green sunfish at Station 2 was greater than 0.5 at 0.6. CVs are presented in Appendix C.

Data variability is not necessarily considered an "uncertainty;" however, as the majority of the data sets have a relatively low variability, variations in individual fish sample concentrations within a species, should not have an appreciable impact on the risk estimate.

The fish component of the piscivore diet was assumed to be composed of equal amounts of each fish present in a given station-reach. This assumption does not account for dietary preferences or seasonal availability. Due to the differences in contaminant levels between species, this could be a source of uncertainty depending on the actual composition of fish in the diet. Mean PCB concentrations for sunfish (green and longear) and creek chub were similar at all three sampling stations; mean PCB concentrations for white suckers, the benthic (or bottom-feeding) species, were greater than the sunfish or creek chub concentrations. Preferential feeding on one fish species (i.e., white sucker) versus another (i.e., creek chub or sunfish) could have an impact on the risk estimate as the mean PCB concentration in white suckers ranged from 36 percent (at Station 3) to over 60 percent (at Stations 1 and 2) greater than the mean PCB concentrations in sunfish and creek chub. If white sucker were found to comprise the majority of the diet, the risk estimates for certain stations could be underestimated; a diet composed primarily of sunfish or creek chub could result in an underestimation of risk.

The available fish tissue data were measured only in samples collected during the month of May; therefore, seasonal fluctuations in PCB body burdens could not be evaluated in the FERA. For a similar site (Neal's Landfill; EPA 2005), the lowest PCB levels were reported for a November sampling event and were 1.3 to 3.4 times greater depending on species for May and August sampling events; lipid percentages were also lowest in November. If November fish concentrations were assumed to be one-half the summer concentrations, and if the November data was also assumed to be representative for 6 months of the year, the resulting overestimation of risk can be approximated as shown below.

Calculated as: X = PCB Concentration for May; 0.5X = PCB Concentration for November

Yearly Average = (X + 0.5X) / 2 = 0.75X

and as this FERA assumes May data is representative of the entire year,

Percent overestimation = (X – 0.75X) / 0.75X = 0.33 (or 33 percent)

Assuming November data is representative for 4 months of the year results in an overestimation of approximately 6 percent. Depending on the contribution of fall/winter data to average yearly PCB concentrations, the exclusive use of May data may overestimate risk in the range of 6 to 33 percent.

No crayfish data were available for Stations 2 and 3. Therefore, crayfish concentrations were modeled using a ratio calculated from fish and crayfish data for Station 1. The use of a ratio assumes that uptake factors and contaminant loading are the same at all stations. At Neal's Landfill, the fish:crayfish ratios were similar across sampling stations, sampling dates, and analytical methods (EPA 2005).

6.2 TOXICITY REFERENCE VALUES

Source of uncertainty associated with the selection of TRVs include: (1) extrapolation of toxicity values across species, (2) extrapolation of laboratory studies to field conditions, (3) differences in toxicity between the compound administered in a laboratory study and the compound present in the field, and (4) potential interactions between the primary COPC and other contaminants present in the diet.

6.2.1 Mink TRVs

As the TRVs for mink were selected from studies conducted on mink rather than a surrogate species, interspecies extrapolation was not an issue. Field-contaminated prey were not used in these studies, therefore, the results were not confounded by co-contaminants. Although the toxicological studies were based on captive feeding, there are indications that the effects observed in the laboratory studies are similar to those observed in the field. For instance, changes in the otter population in Sweden have been correlated to the concentration of PCBs in muscle tissue, but not to other contaminants. A range of PCB concentrations between 10 ppm to 30 ppm in muscle tissue is associated with a threshold for population effects (Roos and others 2001, as cited in EPA 2005). In mink, PCB concentrations of 40 ppm to 60 ppm have been associated with reduction in litter sizes (Leonards and others 1997, as cited in EPA 2005). The PCB concentrations associated with reproductive effects in mink are of the same order of magnitude as the concentrations associated with adverse effects in wild populations (otters).

In deriving the TRVs used in the FERA, the toxicity values identified for toxicological studies conducted over one breeding season were extrapolated to account for exposure over several breeding seasons.

Studies conducted with Clophen (a European PCB formulation similar to Aroclors) and field-contaminated prey indicated increased effects after two breeding seasons. Although no definite rule is available to account for exposure over time, it is thought that effects are related to both dose and exposure time. For the FERA, the mean difference in toxicity for the long-term Clophen studies (two breeding seasons) was used to adjust the toxicity values identified for the Aroclor studies conducted over one breeding season (Chapman 2003, as cited in EPA 2005). If no adjustment was made in the single-season toxicity values, the TRVs may not be protective of long-term use of Stout's Creek. The adjusted TRVs were still within the range of TRVs identified in other ERAs (see the Attachment).

6.2.2 Kingfisher TRVs

No studies evaluating the effects of PCBs in kingfishers were identified. TRVs selected for the kingfisher were based on surrogate species (i.e., chicken and doves). In order to address the interspecies extrapolation, two sets of TRVs were identified. TRVs were identified from a study in which chickens were exposed to PCBs. Chickens have been shown to be sensitive to the effects of PCB exposure (decreased egg hatchability). A second set of TRVs were identified for behavioral effects in doves. Doves are less sensitive to PCB exposure than the chicken. Therefore, with the use of two sets of TRVs, the kingfisher toxicity is "bracketed." Both the chicken and dove studies were controlled feeding studies where the test animals were exposed to Aroclors. The avian TRVs selected for the FERA are representative of the range of TRVs used in other ERAs (see the Attachment).

HQs were calculated using both the chicken TRVs and the dove TRVs (after dose conversion) so that a risk range for possible adverse effects was estimated for both a NOAEL and LOAEL dose.

6.3 EXPOSURE ASSUMPTIONS

Uncertainties associated with the exposure assumptions used in the FERA are discussed in the following sections.

6.3.1 Mink Dietary Composition and Foraging Assumptions

For the FERA, the mink was assumed to obtain 79 percent of its diet from aquatic prey in the Stout's Creek area. Mink are not limited to a diet composed strictly of aquatic prey. The mink diet will also vary with season and location. Therefore, it is possible that a mink foraging along Stout's Creek may consume less or more than 79 percent of their diet from Stout's Creek. Assumptions were also made concerning dietary composition. Although the mink has a fairly varied diet (including fish, invertebrates, small mammals, and birds), a diet of fish and crayfish only was assumed for the FERA. A mink foraging along Stout's Creek may have a lower or greater composition of fish or crayfish (i.e., a different ratio) in their diet.

Selection of a diet with a much lower component of aquatic items is not appropriate for modeling exposure as only those receptors adhering to these dietary assumptions (mainly a terrestrial diet) would be evaluated. No conclusions could be drawn regarding a receptor that did take a larger portion of its prey from Stout's Creek. The mink dietary assumptions used in the FERA were not the highest available in the literature, but were selected to provide an evaluation based on a reasonable maximum exposure.

PCB uptake was not modeled for the remaining 21 percent of the mink diet, which was assumed to be composed of terrestrial prey. It was assumed that terrestrial prey at the site would be less exposed to PCBs released from Bennett's Dump site as the primary release is to groundwater, which discharges into Stout's Creek. If terrestrial prey was contaminated, then the mink diet assumptions would result in an underestimation of up to 21 percent. Without site-specific terrestrial data, the degree of underestimation is unknown.

Home ranges of 1 mile were assumed for both the mink and kingfisher. For mink, the home range assumption was based on the mean home range for female mink along streams. A range of 0.6 to 1.7 stream miles was actually reported. The use of the 1-mile home range may over- or underestimate exposure based on what the actual home range is within the actual distances observed. As all of the stations are greater than 2 miles apart, which is slightly greater than the upper bound of home ranges reported for female mink, the home range assumption used in the FERA should have little effect on the risk estimate.

6.3.2 Kingfisher Dietary Composition and Foraging Assumptions

The kingfisher diet was assumed to be comprised of 100 percent aquatic prey. This assumption is consistent with dietary information available in the literature (EPA 1993). Although the possibility for non-aquatic items to be consumed does exist, these items constitute a small portion of the total diet (1 to 3 percent [EPA 1993]) and are not estimated to be a significant source of uncertainty.

A dietary composition of 80 percent fish and 20 percent crayfish was assumed for the FERA. The amount of crayfish in the kingfisher diet for the FERA was based on the mean value of crayfish (or crayfish and invertebrates) reported for dietary studies. Two dietary studies reported crayfish/invertebrate intake at 5 percent (based on mass) and 13 percent (based on number of prey). A third study had a higher proportion at 41 percent (percent volume basis). Averaging these three values resulted in a value of approximately 20 percent. The 41 percent value cited by Salyer and Lagler (1946 as cited in EPA 1993) is much higher than what was reported for the other studies—the incorporation of this value "maximized" the amount of crayfish consumed. Therefore, the kingfisher diet may be composed of a smaller amount of crayfish than what was assumed for the FERA. As fish would be expected to have higher PCB concentrations compared to crayfish, overestimating the amount of crayfish in the diet would potentially underestimate overall risk.

Home ranges of 1 mile were assumed for both the mink and kingfisher. A range of 0.6 to 1.4 stream miles was reported in the literature for the kingfisher. The use of the 1-mile home range may over- or underestimate exposure based on what the actual home range is within the actual distances observed. The kingfisher home range may have been over- or underestimated by 30 to 40 percent. As all of the stations are greater than 2 miles apart, which is equivalent to the upper bound of home ranges reported for kingfisher, the home range assumption used in the FERA should have little effect on the risk estimate.

7.0 SUMMARY

Remedial actions have been undertaken at the Bennett's Dump site to reduce the release of PCBs to the environment. However, PCB-impacted groundwater discharging from the Bennett's Dump site has been released to Stout's Creek. The FERA evaluated risk to piscivorous mammals (mink) and birds (kingfisher). Mink have been shown to be sensitive to the effects of PCBs. Both species are potentially highly exposed through the piscivorous diet. As PCBs elicit reproductive effects in both species, and as reproductive success is a critical endpoint for population stability, risks were evaluated based on the likelihood of PCB-exposure to cause adverse reproductive effects in the mink or kingfisher. Fish data from three sampling stations on Stout's Creek were used to estimate risks to the mink and kingfisher.

Despite the reductions in potential PCB release from the Bennett's Dump site due to the remedial action, fish in the upper portion of Stout's Creek (i.e., near Station 1 at Hunter Valley Road, which is approximately 1 mile from the site) are accumulating PCBs at concentrations greater than those shown to cause reproductive effects in the mink or kingfisher.

For the RME scenario, HQs for both the mink and kingfisher at Station 1 were greater than the threshold of 1 (see Tables 3 and 4). The low-effect-based HQ for the mink was 7, with a no-effect-based HQ of 9. For the kingfisher, HQs ranged from 2 (LOAEL-based HQ) to 24 (NOAEL-based HQ) for the RME scenario. Consumption of as little as 11 percent of the diet for the mink or 4 to 15 percent of the diet for the kingfisher from the Station 1-reach could result in an exposure exceeding the threshold for reproductive effects.

PCB concentrations in Stout's Creek fish appear to appreciably decrease downstream of Station 1. Based on whole-fish data, risk to mink or kingfisher appears to be low for the areas greater than 5 miles downstream.

It should be noted that seasonal data was not available for the Stout's Creek evaluation; lipid concentrations appear to be higher in summer months based on data from a similar site (Neal's Landfill; EPA 2005), which could lead to lower whole-body PCB concentrations in fall/winter months. In addition, no crayfish data was available for Stations 2 or 3; crayfish concentrations at these stations were modeled using data from Station 1, which is nearest the site. The lack of seasonal data could result in an overestimation in the range of 6 to 33 percent.

The greatest source of uncertainty in the risk estimate is attributable to the assumptions for the mink diet. Approximately 21 percent of the mink diet is unaccounted for by the dietary composition used to estimate risks. Other sources of uncertainty do not appear to significantly affect the outcome of the risk assessment.

As PCBs are ubiquitous contaminants, it is not surprising that PCBs were detected in fish as far as 5 miles from the site. PCB concentrations in fish at Station 1 are approximately an order of magnitude higher than at stations located farther downstream. The Station 1 location may be impacted by the release of PCB-impacted groundwater from the site, which flows into the on-site springs and then discharges to Stout's Creek upstream of Station 1. Although risks were lower at Station 2 (3 miles downstream) that at Station 1, the low-effect HQs for both the mink and kingfisher were greater than the threshold of 1, indicating the exposure concentration may be greater than the concentration shown to cause an adverse effect (i.e., a potential risk). At Station 3 (5 miles downstream), exposure concentrations are decreased from those estimated at Stations 1 and 2. For the mink, the no-effect-based HQs were equivalent to 1; the no-effect- and LOAEL-based HQs were equal to or less than 1 respectively; although the low-effect-based HQ was equivalent to 1, indicating concentrations equal to those shown to cause an adverse effect.

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APPENDIX A

FISH AND CRAYFISH DATA USED IN THE FERA

(Three Pages)

TABLE A-1

TOTAL PCB CONCENTRATIONS IN WHOLE FISH AND CRAYFISH SAMPLES - MAY 2004
BENNETT'S DUMP SITE, BLOOMINGTON, INDIANA

ID	Station	Species	Total PCB Concentration (ug/kg)	95UCL Concentration (ug/kg)	Mean PCB Concentration (ug/kg)	Percent Lipids
52	1	GS	1,700			2.7
53	1	GS	5,300			2.01
54	1	GS	3,200			1.67
55	1	GS	5,200	4,595	3,549	2.43
56	1	GS	2,500			1.15
57	1	GS	4,400			2
58	1	GS	2,540			2.48
59	1	WS	12,000			4.69
60	1	WS	7,600			3.98
61	1	WS	9,100			5.46
62	1	WS	8,400	10,705	9,157	5.12
63	1	WS	12,000			5.04
64	1	WS	8,500			4.18
64 (duplicate)	1	WS	8,500 (duplicate)			3.8
65	1	WS	6,500			5.78
66	1	CRC	1,800			4.34
67	1	CRC	2,490			6.7
68	1	CRC	1,900			3.67
69	1	CRC	4,300	4,075	3,184	3.62
70	1	CRC	4,100			4.34
71	1	CRC	2,900			3.5
72	1	CRC	4,800			3.76
73	1	CF	1,100			1.47
74	1	CF	560	1,100	877	1.01
75	1	CF	970			2.15
30	2	GS	3,400			2.93
31	2	GS	1,600			1.77
32	2	GS	900			2.43
33	2	GS	1,600	2,311	1,606	1.29
34	2	GS	960			1.27
35	2	GS	2,200			1.6
36	2	GS	581			1.79
43	2	WS	5,900			4.95
44	2	WS	4,300			4.12
45	2	WS	2,700			4.7
45 (duplicate)	2	WS	2,500 (duplicate)	5,544	4,245	4.24
46	2	WS	4,200			4.42
47	2	WS	6,100			4.92
48	2	WS	2,270			4.62
37	2	CRC	1,700			1.25
38	2	CRC	1,100			1.31
39	2	CRC	610	1,810	1,327	2.5
40	2	CRC	1,500			2.18
41	2	CRC	2,200			2.8
42	2	CRC	849			3.17

TABLE A-1

TOTAL PCB CONCENTRATIONS IN WHOLE FISH AND CRAYFISH SAMPLES - MAY 2004
BENNETT'S DUMP SITE, BLOOMINGTON, INDIANA

ID	Station	Species	Total PCB Concentration (ug/kg)	95UCL Concentration (ug/kg)	Mean PCB Concentration (ug/kg)	Percent Lipids
3	3	LS	600	855	703	3.61
4	3	LS	530			5.09
5	3	LS	540			3.94
6	3	LS	1,100			5.57
10	3	LS	860			3.14
11	3	LS	670			2
12	3	LS	621			6.29
18	3	WS	1,100	1,270	1,091	5.64
19	3	WS	1,000			4.38
20	3	WS	1,300			5.34
21	3	WS	1,100			4.34
21 (duplicate)	3	WS	1,000 (duplicate)			4.19
22	3	WS	1,100			5.16
23	3	WS	1,400			4.37
24	3	WS	635			6.83
1	3	CRC	440	653	562	4.31
2	3	CRC	560			5.78
13	3	CRC	770			4.67
14	3	CRC	660			2.44
15	3	CRC	590			4.38
16	3	CRC	480			4.84
17	3	CRC	435			4.71

Notes:

To avoid "double-counting," duplicate results were not included in the calculation of the 95-percent upper confidence limit (95UCL) or the arithmetic mean.
The 95UCL was calculated using ProUCL version 3.0 (EPA 2004b).

Station	Fish sample collection station (see Figure 3 of the main text)	
	1	Hunter Valley Road
	2	Acuff Road
	3	Maple Grove Road
Species	Sampled fish species	
	CF	Crayfish (Station 1 only)
	CRC	Creek chub
	GS	Green sunfish (Station 1 and 2 only)
	LS	Longear sunfish (Station 3 only)
	WS	White sucker
PCB	Polychlorinated biphenyl	
ug/kg	Microgram per kilogram	

TABLE A-2

MODELED TOTAL PCB CONCENTRATIONS IN CRAYFISH
BENNETT'S DUMP SITE, BLOOMINGTON, INDIANA

Station	Species	95UCL Concentration (ug/kg)	Mean PCB Concentration (ug/kg)
1	GS	4,595	3,549
1	WS	10,705	9,157
1	CRC	4,075	3,184
1	CF	1,100	877
		Mean GS:CF Ratio	**4.0**
		Mean WS:CF Ratio	**10.4**
		Mean CRC:CF Ratio	**3.6**
2	GS	2,311	1,606
2	WS	5,544	4,245
2	CRC	1,810	1,327
2	CF	533	389
3	LS	855	703
3	WS	1,270	1,091
3	CRC	653	562
3	CF	171	144

Notes:

Measured crayfish data was available for Station 1; crayfish concentrations were modeled for Stations 2 and 3. To calculate the modeled PCB concentration in crayfish, fish:crayfish ratios were calculated for each species using the mean PCB concentration data for Station 1.

For example, Mean GS:CF Ratio = $\dfrac{\text{Mean PCB concentration in green sunfish at Station 1}}{\text{Mean PCB concentration in crayfish at Station 1}}$

Mean species-specific fish:crayfish ratios were then used to model crayfish concentrations at Stations 2 and 3, where

Modeled crayfish concentration =

[$\sum$ (Species-specific PCB concentration ÷ Mean species-specific fish:crayfish ratio)] ÷ n

and Species-specific PCB concentration = 95UCL or Mean PCB concentration in fish and n = number of modeled crayfish concentrations at each station (n=3).

Station	Fish sample collection station (see Figure 3 of the main text)	
	1	Hunter Valley Road
	2	Acuff Road
	3	Maple Grove Road
Species	Sampled fish species	
	CF	Crayfish (Station 1 only)
	CRC	Creek chub
	GS	Green sunfish (Station 1 and 2 only)
	LS	Longear sunfish (Station 3 only)
	WS	White sucker
95UCL	95-percent upper confidence limit on the arithmetic mean	
PCB	Polychlorinated biphenyl	
ug/kg	Microgram per kilogram	

APPENDIX B

MINK AND KINGFISHER RISK ESTIMATES, MAY 2004

(Six Pages)

TABLE B-1

RME TOTAL PCB CONCENTRATIONS IN DIET AND STATION-SPECIFIC RISK, MAY 2004 FISH DATA BENNETT'S DUMP SITE, BLOOMINGTON, INDIANA

				Mink			Kingfisher					
Station	Species	N	Concentration in Fish and Crayfish (ug/kg)	Concentration in Diet (ug/kg)	HQ No Effect (ratio)	HQ Low Effect (ratio)	Concentration in Diet (ug/kg)	Concentration in Dose (ug/kg-day)	HQ No Effect (ratio)	HQ NOAEL (ratio)	HQ Low Effect (ratio)	HQ LOAEL (ratio)
1	GS	7	4,595	4,405.50	8.81	7.34	5,386.67	2,693.33	6.73	24.48	5.39	2.40
1	WS	7	10,705									
1	CRC	7	4,075									
1	CF	3	1,100									
2	GS	7	2,311	2,195.59	4.39	3.66	2,683.93	1,341.97	3.35	12.20	2.68	1.20
2	WS	6	5,544									
2	CRC	6	1,810									
2	CF	--	533									
3	LS	7	855	633.39	1.27	1.06	775.00	387.50	0.97	3.52	0.78	0.35
3	WS	7	1,270									
3	CRC	7	653									
3	CF	--	171									

Notes:

Station	Fish sample collection station (see Figure 3 of the main text)	
	1	Hunter Valley Road
	2	Acuff Road
	3	Maple Grove Road
Species	Sampled fish species	
	CF	Crayfish
	CRC	Creek chub
	GS	Green sunfish
	LS	Longear sunfish
	WS	White sucker

Definitions:

HQ	Hazard quotient
IR	Ingestion rate (kingfisher)
kg	Kilogram
LOAEL	Lowest-observed-adverse-effect leve
N	Number of samples
NOAEL	No-observed-adverse-effect level
PCB	Polychlorinated biphenyl
RME	Reasonable maximum exposure
TRV	Toxicity reference value
ug/kg	Microgram per kilogram
ug/kg-day	Microgram per kilogram per day
--	Not applicable; crayfish concentration was modeled.

TABLE B-1

RME TOTAL PCB CONCENTRATIONS IN DIET AND STATION-SPECIFIC RISK, MAY 2004 FISH DATA
BENNETT'S DUMP SITE, BLOOMINGTON, INDIANA

Notes (Continued):

Concentration in Fish & Crayfish:	For the RME scenario, the 95-percent upper confidence limit was used as the fish species concentration. At Station 1, as the number of crayfish was less than 4, the maximum detected concentration was used as the crayfish concentration. Crayfish concentrations at Stations 2 and 3 were modeled. Original sample results were used to calculate the 95-percent upper confidence limit; duplicate results were not included in the calculation.
Mink, Concentration in Diet:	Concentration of PCBs in the mink diet = (0.66 x mean of the RME fish species concentration) + (0.13 x RME/modeled crayfish concentration)
Mink, HQ, No Effect:	HQ = Concentration of PCBs in the mink diet / No effect-based TRV, where TRV = 500 ug/kg
Mink, HQ, Low Effect:	HQ = Concentration of PCBs in the mink diet / Low effect-based TRV, where TRV = 600 ug/kg
Kingfisher, Concentration in Diet:	Concentration of PCBs in the kingfisher diet = (0.80 x mean of the RME fish species concentration) + (0.20 x RME/modeled crayfish concentration)
Kingfisher, Concentration in Dose:	Concentration of PCBs in kingfisher diet (expressed as a dose) = (Concentration of PCBs in the kingfisher diet x IR), where IR = 0.5 kg food per kg body weight
Kingfisher, HQ, No Effect:	HQ = Concentration of PCBs in kingfisher dose / No effect-based TRV; where TRV = 400 ug/kg-day
Kingfisher, HQ, NOAEL:	HQ = Concentration of PCBs in kingfisher dose / NOAEL-based TRV; where TRV = 110 ug/kg-day
Kingfisher, HQ, Low Effect:	HQ = Concentration of PCBs in kingfisher dose / Low effect-based TRV; where TRV = 500 ug/kg-day
Kingfisher, HQ, LOAEL:	HQ = Concentration of PCBs in kingfisher dose / LOAEL-based TRV; where TRV = 1,120 ug/kg-day

TABLE B-2

CTE TOTAL PCB CONCENTRATIONS IN DIET AND STATION-SPECIFIC RISK, MAY 2004 FISH DATA
BENNETT'S DUMP SITE, BLOOMINGTON, INDIANA

Station	Species	N	Concentration in Fish and Crayfish (ug/kg)	Mink			Kingfisher					
				Concentration in Diet (ug/kg)	HQ No Effect (ratio)	HQ Low Effect (ratio)	Concentration in Diet (ug/kg)	Concentration in Dose (ug/kg-day)	HQ No Effect (ratio)	HQ NOAEL (ratio)	HQ Low Effect (ratio)	HQ LOAEL (ratio)
1	GS	7	3,549	3,609.77	7.22	6.02	4,412.67	2,206.33	5.52	20.06	4.41	1.97
1	WS	7	9,157									
1	CRC	7	3,184									
1	CF	3	877									
2	GS	7	1,606	1,629.59	3.26	2.72	1,991.76	995.88	2.49	9.05	1.99	0.89
2	WS	6	4,245									
2	CRC	6	1,327									
2	CF	--	389									
3	LS	7	703	537.01	1.07	0.90	657.03	328.51	0.82	2.99	0.66	0.29
3	WS	7	1,091									
3	CRC	7	562									
3	CF	--	144									

Notes:

Station	Fish sample collection station (see Figure 3 of the main text)	
	1	Hunter Valley Road
	2	Acuff Road
	3	Maple Grove Road
Species	Sampled fish species	
	CF	Crayfish
	CRC	Creek chub
	GS	Green sunfish
	LS	Longear sunfish
	WS	White sucker

Definitions:

CTE	Central tendency exposure
HQ	Hazard quotient
IR	Ingestion rate (kingfisher)
kg	Kilogram
LOAEL	Lowest-observed-adverse-effect level
N	Number of samples
NOAEL	No-observed-adverse-effect level
PCB	Polychlorinated biphenyl
TRV	Toxicity reference value
ug/kg	Microgram per kilogram
ug/kg-day	Microgram per kilogram per day
--	Not applicable; crayfish concentration was modeled.

TABLE B-2

CTE TOTAL PCB CONCENTRATIONS IN DIET AND STATION-SPECIFIC RISK, MAY 2004 FISH DATA
BENNETT'S DUMP SITE, BLOOMINGTON, INDIANA

Notes (Continued):

Concentration in Fish & Crayfish:	For the CTE scenario, the arithmetic mean was used as the fish species concentration and the crayfish concentration at Station 1. Crayfish concentrations were modeled at Stations 2 and 3. Original sample results were used to calculate the arithmetic mean; duplicate results were not included in the calculation.
Mink, Concentration in Diet:	Concentration of PCBs in the mink diet = (0.66 x mean of the CTE fish species concentration) + (0.13 x mean/modeled crayfish concentration)
Mink, HQ, No Effect:	HQ = Concentration of PCBs in the mink diet / No effect-based TRV, where TRV = 500 ug/kg
Mink, HQ, Low Effect:	HQ = Concentration of PCBs in the mink diet / Low effect-based TRV, where TRV = 600 ug/kg
Kingfisher, Concentration in Diet:	Concentration of PCBs in the kingfisher diet = (0.80 x mean of the CTE fish species concentration) + (0.20 x mean/modeled crayfish concentration)
Kingfisher, Concentration in Dose:	Concentration of PCBs in kingfisher diet (expressed as a dose) = (Concentration of PCBs in the kingfisher diet x IR), where IR = 0.5 kg food per kg body weight
Kingfisher, HQ, No Effect:	HQ = Concentration of PCBs in kingfisher dose / No effect-based TRV; where TRV = 400 ug/kg-day
Kingfisher, HQ, NOAEL:	HQ = Concentration of PCBs in kingfisher dose / NOAEL-based TRV; where TRV = 110 ug/kg-day
Kingfisher, HQ, Low Effect:	HQ = Concentration of PCBs in kingfisher dose / Low effect-based TRV; where TRV = 500 ug/kg-day
Kingfisher, HQ, LOAEL:	HQ = Concentration of PCBs in kingfisher dose / LOAEL-based TRV; where TRV = 1,120 ug/kg-day

TABLE B-3

RME PERCENT ALLOWABLE CONSUMPTION FOR TOTAL PCBs, MAY 2004 FISH DATA
BENNETT'S DUMP SITE, BLOOMINGTON, INDIANA

Station	Mink		Kingfisher			
	Dietary Basis		Dose Basis			
	NOAEC PAC	LOAEC PAC	No Effect PAC	NOAEL PAC	Low Effect PAC	LOAEL PAC
1	11.35	13.62	14.85	4.08	18.56	41.58
2	22.77	27.33	29.81	8.20	37.26	83.46
3	78.94	94.73	103.23	28.39	129.03	289.03
TRV	500 ug/kg	600 ug/kg	400 ug/kg-day	110 ug/kg-day	500 ug/kg-day	1,120 ug/kg-day

Notes:

Station — Fish sample collection station (see Figure 3 of the main text)
- 1 — Hunter Valley Road
- 2 — Acuff Road
- 3 — Maple Grove Road

PAC (Percent Allowable Consumption, %) = (1 / HQ) x 100; the PAC is the percent of an animals diet that can be consumed within a station reach and not exceed the TRV.

% — Percent

HQ — Hazard quotient; where mink HQ = concentration in mink diet / TRV and kingfisher HQ = concentration in kingfisher dose / TRV

LOAEC — Lowest-observed-adverse-effect concentration

LOAEL — Lowest-observed-adverse-effect level

NOAEC — No-observed-adverse-effect concentration

NOAEL — No-observed-adverse-effect level

PCB — Polychlorinated biphenyl

RME — Reasonable maximum exposure

TRV — Toxicity reference value; see Section 4.2.2 of the main text.

ug/kg — Microgram per kilogram

ug/kg-day — Microgram per kilogram per day

TABLE B-4

CTE PERCENT ALLOWABLE CONSUMPTION FOR TOTAL PCBs, MAY 2004 FISH DATA
BENNETT'S DUMP SITE, BLOOMINGTON, INDIANA

Station	Mink		Kingfisher			
	Dietary Basis		Dose Basis			
	NOAEC PAC	LOAEC PAC	No Effect PAC	NOAEL PAC	Low Effect PAC	LOAEL PAC
1	13.85	16.62	18.13	4.99	22.66	50.76
2	30.68	36.82	40.17	11.05	50.21	112.46
3	93.11	111.73	121.76	33.48	152.20	340.93
TRV	500 ug/kg	600 ug/kg	400 ug/kg-day	110 ug/kg-day	500 ug/kg-day	1,120 ug/kg-day

Notes:

Station — Fish sample collection station (see Figure 3 of the main text)
- 1 — Hunter Valley Road
- 2 — Acuff Road
- 3 — Maple Grove Road

PAC (Percent Allowable Consumption, %) = (1 / HQ) x 100; the PAC is the percent of an animals diet that can be consumed within a station reach and not exceed the TRV.

% — Percent
CTE — Central tendency exposure
HQ — Hazard quotient; where mink HQ = concentration in mink diet / TRV and kingfisher HQ = concentration in kingfisher dose / TRV
LOAEC — Lowest-observed-adverse-effect concentration
LOAEL — Lowest-observed-adverse-effect level
NOAEC — No-observed-adverse-effect concentration
NOAEL — No-observed-adverse-effect level
PCB — Polychlorinated biphenyl
TRV — Toxicity reference value; see Section 4.2.2 of the main text.
ug/kg — Microgram per kilogram
ug/kg-day — Microgram per kilogram per day

APPENDIX C

ProUCL Version 3.0 MODEL OUTPUT FOR DETERMINATION OF 95-PERCENT UPPER CONFIDENCE LIMIT

(Nine Pages)

APPENDIX C

ProUCL Version3.0 MODEL OUTPUT FOR DETERMINATION OF 95-PERCENT UPPER CONFIDENCE LIMIT BENNETT'S DUMP SITE, BLOOMINGTON, INDIANA

Station 1: Hunter Valley Road **Green Sunfish**

Raw Statistics	
Number of Valid Samples	7
Number of Unique Samples	7
Minimum	1700
Maximum	5300
Mean	3548.5714
Median	3200
Standard Deviation	1424.4932
Variance	2029181
Coefficient of Variation	0.4014272
Skewness	0.1578434

Gamma Statistics	
k hat	6.8212593
k star (bias corrected)	3.9931005
Theta hat	520.22234
Theta star	888.6757
nu hat	95.49763
nu star	55.903407
Approx.Chi Square Value (.05)	39.716796
Adjusted Level of Significance	0.01584
Adjusted Chi Square Value	35.639299

Log-transformed Statistics	
Minimum of log data	7.4383835
Maximum of log data	8.5754621
Mean of log data	8.099213
Standard Deviation of log data	0.4282674
Variance of log data	0.183413

RECOMMENDATION
Data are normal (0.05)

Use Student's-t UCL

Normal Distribution Test	
Shapiro-Wilk Test Statisitic	0.9068716
Shapiro-Wilk 5% Critical Value	0.803
Data are normal at 5% significance level	
95% UCL (Assuming Normal Distribution)	
Student's-t UCL	**4594.7945**
Gamma Distribution Test	
A-D Test Statistic	0.3420822
A-D 5% Critical Value	0.7093573
K-S Test Statistic	0.1891799
K-S 5% Critical Value	0.3126166
Data follow gamma distribution at 5% significance level	
95% UCLs (Assuming Gamma Distribution)	
Approximate Gamma UCL	4994.7945
Adjusted Gamma UCL	5566.2496
Lognormal Distribution Test	
Shapiro-Wilk Test Statisitic	0.9227409
Shapiro-Wilk 5% Critical Value	0.803
Data are lognormal at 5% significance level	
95% UCLs (Assuming Lognormal Distribution)	
95% H-UCL	5423.4453
95% Chebyshev (MVUE) UCL	6078.9152
97.5% Chebyshev (MVUE) UCL	7169.5508
99% Chebyshev (MVUE) UCL	9311.8932
95% Non-parametric UCLs	
CLT UCL	4434.1735
Adj-CLT UCL (Adjusted for skewness)	4468.4952
Mod-t UCL (Adjusted for skewness)	4600.148
Jackknife UCL	4594.7945
Standard Bootstrap UCL	4380.2014
Bootstrap-t UCL	4701.0781
Hall's Bootstrap UCL	4284.9546
Percentile Bootstrap UCL	4342.8571
BCA Bootstrap UCL	4614.2857
95% Chebyshev (Mean, Sd) UCL	5895.4368
97.5% Chebyshev (Mean, Sd) UCL	6910.9273
99% Chebyshev (Mean, Sd) UCL	8905.6617

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ProUCL Version3.0 MODEL OUTPUT FOR DETERMINATION OF 95-PERCENT UPPER CONFIDENCE LIMIT BENNETT'S DUMP SITE, BLOOMINGTON, INDIANA

Station 1: Hunter Valley Road **White Sucker**

Raw Statistics	
Number of Valid Samples	7
Number of Unique Samples	6
Minimum	6500
Maximum	12000
Mean	9157.1429
Median	8500
Standard Deviation	2107.8086
Variance	4442857.1
Coefficient of Variation	0.2301819
Skewness	0.5655223

Gamma Statistics	
k hat	22.687435
k star (bias corrected)	13.059487
Theta hat	403.62178
Theta star	701.18704
nu hat	317.62409
nu star	182.83281
Approx.Chi Square Value (.05)	152.5506
Adjusted Level of Significance	0.01584
Adjusted Chi Square Value	144.18054

Log-transformed Statistics	
Minimum of log data	8.7795575
Maximum of log data	9.3926619
Mean of log data	9.100089
Standard Deviation of log data	0.226565
Variance of log data	0.0513317

RECOMMENDATION
Data are normal (0.05)

Use Student's-t UCL

Normal Distribution Test	
Shapiro-Wilk Test Statisitic	0.8860004
Shapiro-Wilk 5% Critical Value	0.803
Data are normal at 5% significance level	

95% UCL (Assuming Normal Distribution)	
Student's-t UCL	**10705.229**

Gamma Distribution Test	
A-D Test Statistic	0.3982678
A-D 5% Critical Value	0.7067687
K-S Test Statistic	0.2074683
K-S 5% Critical Value	0.3114546
Data follow gamma distribution at 5% significance level	

95% UCLs (Assuming Gamma Distribution)	
Approximate Gamma UCL	10974.891
Adjusted Gamma UCL	11612.012

Lognormal Distribution Test	
Shapiro-Wilk Test Statisitic	0.9182663
Shapiro-Wilk 5% Critical Value	0.803
Data are lognormal at 5% significance level	

95% UCLs (Assuming Lognormal Distribution)	
95% H-UCL	11081.461
95% Chebyshev (MVUE) UCL	12576.342
97.5% Chebyshev (MVUE) UCL	14056.868
99% Chebyshev (MVUE) UCL	16965.074

95% Non-parametric UCLs	
CLT UCL	10467.56
Adj-CLT UCL (Adjusted for skewness)	10649.514
Mod-t UCL (Adjusted for skewness)	10733.61
Jackknife UCL	10705.229
Standard Bootstrap UCL	10345.865
Bootstrap-t UCL	11714.538
Hall's Bootstrap UCL	14014.707
Percentile Bootstrap UCL	10442.857
BCA Bootstrap UCL	10800
95% Chebyshev (Mean, Sd) UCL	12629.776
97.5% Chebyshev (Mean, Sd) UCL	14132.388
99% Chebyshev (Mean, Sd) UCL	17083.977

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ProUCL Version3.0 MODEL OUTPUT FOR DETERMINATION OF 95-PERCENT UPPER CONFIDENCE LIMIT BENNETT'S DUMP SITE, BLOOMINGTON, INDIANA

Station 1: Hunter Valley Road **Creek Chub**

Raw Statistics	
Number of Valid Samples	7
Number of Unique Samples	7
Minimum	1800
Maximum	4800
Mean	3184.2857
Median	2900
Standard Deviation	1212.5985
Variance	1470395.2
Coefficient of Variation	0.3808071
Skewness	0.1630623

Gamma Statistics	
k hat	7.7264581
k star (bias corrected)	4.510357
Theta hat	412.12748
Theta star	705.99417
nu hat	108.17041
nu star	63.144998
Approx.Chi Square Value (.05)	45.861851
Adjusted Level of Significance	0.01584
Adjusted Chi Square Value	41.451776

Log-transformed Statistics	
Minimum of log data	7.4955419
Maximum of log data	8.4763712
Mean of log data	7.999877
Standard Deviation of log data	0.3985866
Variance of log data	0.1588713

RECOMMENDATION
Data are normal (0.05)

Use Student's-t UCL

Normal Distribution Test	
Shapiro-Wilk Test Statisitic	0.9032458
Shapiro-Wilk 5% Critical Value	0.803
Data are normal at 5% significance level	

95% UCL (Assuming Normal Distribution)	
Student's-t UCL	**4074.8822**

Gamma Distribution Test	
A-D Test Statistic	0.377123
A-D 5% Critical Value	0.7091075
K-S Test Statistic	0.2312223
K-S 5% Critical Value	0.3124211
Data follow gamma distribution at 5% significance level	

95% UCLs (Assuming Gamma Distribution)	
Approximate Gamma UCL	4384.2913
Adjusted Gamma UCL	4850.7382

Lognormal Distribution Test	
Shapiro-Wilk Test Statisitic	0.9049588
Shapiro-Wilk 5% Critical Value	0.803
Data are lognormal at 5% significance level	

95% UCLs (Assuming Lognormal Distribution)	
95% H-UCL	4676.2723
95% Chebyshev (MVUE) UCL	5290.1084
97.5% Chebyshev (MVUE) UCL	6199.3524
99% Chebyshev (MVUE) UCL	7985.3861

95% Non-parametric UCLs	
CLT UCL	3938.1537
Adj-CLT UCL (Adjusted for skewness)	3968.336
Mod-t UCL (Adjusted for skewness)	4079.59
Jackknife UCL	4074.8822
Standard Bootstrap UCL	3878.8286
Bootstrap-t UCL	4171.1643
Hall's Bootstrap UCL	3774.4073
Percentile Bootstrap UCL	3884.2857
BCA Bootstrap UCL	3755.7143
95% Chebyshev (Mean, Sd) UCL	5182.0527
97.5% Chebyshev (Mean, Sd) UCL	6046.488
99% Chebyshev (Mean, Sd) UCL	7744.5039

APPENDIX C

ProUCL Version3.0 MODEL OUTPUT FOR DETERMINATION OF 95-PERCENT UPPER CONFIDENCE LIMIT BENNETT'S DUMP SITE, BLOOMINGTON, INDIANA

Station 2: Acuff Road **Green Sunfish**

Raw Statistics	
Number of Valid Samples	7
Number of Unique Samples	6
Minimum	581
Maximum	3400
Mean	1605.8571
Median	1600
Standard Deviation	960.70468
Variance	922953.48
Coefficient of Variation	0.5982504
Skewness	1.1282545

Gamma Statistics	
k hat	3.4831054
k star (bias corrected)	2.0855841
Theta hat	461.04178
Theta star	769.97958
nu hat	48.763476
nu star	29.198177
Approx.Chi Square Value (.05)	17.861686
Adjusted Level of Significance	0.01584
Adjusted Chi Square Value	15.247809

Log-transformed Statistics	
Minimum of log data	6.3647508
Maximum of log data	8.1315307
Mean of log data	7.2310486
Standard Deviation of log data	0.5968095
Variance of log data	0.3561816

RECOMMENDATION
Data are normal (0.05)

Use Student's-t UCL

Normal Distribution Test	
Shapiro-Wilk Test Statisitic	0.9067052
Shapiro-Wilk 5% Critical Value	0.803
Data are normal at 5% significance level	

95% UCL (Assuming Normal Distribution)	
Student's-t UCL	**2311.4494**

Gamma Distribution Test	
A-D Test Statistic	0.2319306
A-D 5% Critical Value	0.7109777
K-S Test Statistic	0.1860378
K-S 5% Critical Value	0.3133142
Data follow gamma distribution	
at 5% significance level	

95% UCLs (Assuming Gamma Distribution)	
Approximate Gamma UCL	2625.0658
Adjusted Gamma UCL	3075.0714

Lognormal Distribution Test	
Shapiro-Wilk Test Statisitic	0.975307
Shapiro-Wilk 5% Critical Value	0.803
Data are lognormal at 5% significance level	

95% UCLs (Assuming Lognormal Distribution)	
95% H-UCL	3161.1633
95% Chebyshev (MVUE) UCL	3192.6254
97.5% Chebyshev (MVUE) UCL	3879.579
99% Chebyshev (MVUE) UCL	5228.9663

95% Non-parametric UCLs	
CLT UCL	2203.1236
Adj-CLT UCL (Adjusted for skewness)	2368.5784
Mod-t UCL (Adjusted for skewness)	2337.257
Jackknife UCL	2311.4494
Standard Bootstrap UCL	2148.6169
Bootstrap-t UCL	2605.7453
Hall's Bootstrap UCL	5339.1423
Percentile Bootstrap UCL	2171.4286
BCA Bootstrap UCL	2477.2857
95% Chebyshev (Mean, Sd) UCL	3188.6267
97.5% Chebyshev (Mean, Sd) UCL	3873.4923
99% Chebyshev (Mean, Sd) UCL	5218.7783

APPENDIX C

ProUCL Version3.0 MODEL OUTPUT FOR DETERMINATION OF 95-PERCENT UPPER CONFIDENCE LIMIT BENNETT'S DUMP SITE, BLOOMINGTON, INDIANA

Station 2: Acuff Road **White Sucker**

Raw Statistics	
Number of Valid Samples	6
Number of Unique Samples	6
Minimum	2270
Maximum	6100
Mean	4245
Median	4250
Standard Deviation	1579.4144
Variance	2494550
Coefficient of Variation	0.3720647
Skewness	-0.036201

Gamma Statistics	
k hat	7.9657749
k star (bias corrected)	4.0939986
Theta hat	532.90484
Theta star	1036.8836
nu hat	95.589299
nu star	49.127983
Approx.Chi Square Value (.05)	34.034222
Adjusted Level of Significance	0.01222
Adjusted Chi Square Value	29.567434

Log-transformed Statistics	
Minimum of log data	7.7275351
Maximum of log data	8.7160441
Mean of log data	8.2894173
Standard Deviation of log data	0.4029981
Variance of log data	0.1624075

RECOMMENDATION
Data are normal (0.05)

Use Student's-t UCL

Normal Distribution Test	
Shapiro-Wilk Test Statisitic	0.9103981
Shapiro-Wilk 5% Critical Value	0.788
Data are normal at 5% significance level	

95% UCL (Assuming Normal Distribution)	
Student's-t UCL	**5544.2895**

Gamma Distribution Test	
A-D Test Statistic	0.3554138
A-D 5% Critical Value	0.6982161
K-S Test Statistic	0.2019337
K-S 5% Critical Value	0.3326061
Data follow gamma distribution at 5% significance level	

95% UCLs (Assuming Gamma Distribution)	
Approximate Gamma UCL	6127.6056
Adjusted Gamma UCL	7053.3103

Lognormal Distribution Test	
Shapiro-Wilk Test Statisitic	0.9046931
Shapiro-Wilk 5% Critical Value	0.788
Data are lognormal at 5% significance level	

95% UCLs (Assuming Lognormal Distribution)	
95% H-UCL	6743.5892
95% Chebyshev (MVUE) UCL	7308.9832
97.5% Chebyshev (MVUE) UCL	8629.347
99% Chebyshev (MVUE) UCL	11222.946

95% Non-parametric UCLs	
CLT UCL	5305.5905
Adj-CLT UCL (Adjusted for skewness)	5295.4083
Mod-t UCL (Adjusted for skewness)	5542.7013
Jackknife UCL	5544.2895
Standard Bootstrap UCL	5207.0461
Bootstrap-t UCL	5635.6642
Hall's Bootstrap UCL	5555.6431
Percentile Bootstrap UCL	5166.6667
BCA Bootstrap UCL	5450
95% Chebyshev (Mean, Sd) UCL	7055.5886
97.5% Chebyshev (Mean, Sd) UCL	8271.7325
99% Chebyshev (Mean, Sd) UCL	10660.612

APPENDIX C

ProUCL Version3.0 MODEL OUTPUT FOR DETERMINATION OF 95-PERCENT UPPER CONFIDENCE LIMIT BENNETT'S DUMP SITE, BLOOMINGTON, INDIANA

Station 2: Acuff Road **Creek Chub**

Raw Statistics	
Number of Valid Samples	6
Number of Unique Samples	6
Minimum	610
Maximum	2200
Mean	1326.5
Median	1300
Standard Deviation	587.41595
Variance	345057.5
Coefficient of Variation	0.4428315
Skewness	0.3485359

Gamma Statistics	
k hat	5.7858871
k star (bias corrected)	3.0040547
Theta hat	229.26476
Theta star	441.56986
nu hat	69.430646
nu star	36.048656
Approx.Chi Square Value (.05)	23.305611
Adjusted Level of Significance	0.01222
Adjusted Chi Square Value	19.691128

Log-transformed Statistics	
Minimum of log data	6.413459
Maximum of log data	7.6962126
Mean of log data	7.1014
Standard Deviation of log data	0.4738888
Variance of log data	0.2245706

RECOMMENDATION
Data are normal (0.05)

Use Student's-t UCL

Normal Distribution Test	
Shapiro-Wilk Test Statisitic	0.9740084
Shapiro-Wilk 5% Critical Value	0.788
Data are normal at 5% significance level	

95% UCL (Assuming Normal Distribution)	
Student's-t UCL	**1809.7319**

Gamma Distribution Test	
A-D Test Statistic	0.1782065
A-D 5% Critical Value	0.698094
K-S Test Statistic	0.1698642
K-S 5% Critical Value	0.3327413
Data follow gamma distribution at 5% significance level	

95% UCLs (Assuming Gamma Distribution)	
Approximate Gamma UCL	2051.8038
Adjusted Gamma UCL	2428.4309

Lognormal Distribution Test	
Shapiro-Wilk Test Statisitic	0.9760939
Shapiro-Wilk 5% Critical Value	0.788
Data are lognormal at 5% significance level	

95% UCLs (Assuming Lognormal Distribution)	
95% H-UCL	2375.9842
95% Chebyshev (MVUE) UCL	2452.3501
97.5% Chebyshev (MVUE) UCL	2937.52
99% Chebyshev (MVUE) UCL	3890.5423

95% Non-parametric UCLs	
CLT UCL	1720.9549
Adj-CLT UCL (Adjusted for skewness)	1757.4154
Mod-t UCL (Adjusted for skewness)	1815.419
Jackknife UCL	1809.7319
Standard Bootstrap UCL	1691.4657
Bootstrap-t UCL	1831.1934
Hall's Bootstrap UCL	1747.5911
Percentile Bootstrap UCL	1700
BCA Bootstrap UCL	1816.6667
95% Chebyshev (Mean, Sd) UCL	2371.8143
97.5% Chebyshev (Mean, Sd) UCL	2824.1227
99% Chebyshev (Mean, Sd) UCL	3712.5949

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ProUCL Version3.0 MODEL OUTPUT FOR DETERMINATION OF 95-PERCENT UPPER CONFIDENCE LIMIT BENNETT'S DUMP SITE, BLOOMINGTON, INDIANA

Station 3: Maple Grove Road **Longear Sunfish**

Raw Statistics	
Number of Valid Samples	7
Number of Unique Samples	7
Minimum	530
Maximum	1100
Mean	703
Median	621
Standard Deviation	207.03381
Variance	42863
Coefficient of Variation	0.2945004
Skewness	1.4527881

Gamma Statistics	
k hat	15.548379
k star (bias corrected)	8.980026
Theta hat	45.213717
Theta star	78.284852
nu hat	217.6773
nu star	125.72036
Approx.Chi Square Value (.05)	100.81871
Adjusted Level of Significance	0.01584
Adjusted Chi Square Value	94.090276

Log-transformed Statistics	
Minimum of log data	6.272877
Maximum of log data	7.0030655
Mean of log data	6.5228546
Standard Deviation of log data	0.2664123
Variance of log data	0.0709755

RECOMMENDATION
Data are normal (0.05)

Use Student's-t UCL

Normal Distribution Test	
Shapiro-Wilk Test Statisitic	0.830234
Shapiro-Wilk 5% Critical Value	0.803
Data are normal at 5% significance level	

95% UCL (Assuming Normal Distribution)	
Student's-t UCL	**855.05657**

Gamma Distribution Test	
A-D Test Statistic	0.5066163
A-D 5% Critical Value	0.7074813
K-S Test Statistic	0.2553364
K-S 5% Critical Value	0.3116692
Data follow gamma distribution	
at 5% significance level	

95% UCLs (Assuming Gamma Distribution)	
Approximate Gamma UCL	876.63702
Adjusted Gamma UCL	939.32571

Lognormal Distribution Test	
Shapiro-Wilk Test Statisitic	0.8784742
Shapiro-Wilk 5% Critical Value	0.803
Data are lognormal at 5% significance level	

95% UCLs (Assuming Lognormal Distribution)	
95% H-UCL	883.84017
95% Chebyshev (MVUE) UCL	1009.8738
97.5% Chebyshev (MVUE) UCL	1143.3265
99% Chebyshev (MVUE) UCL	1405.4685

95% Non-parametric UCLs	
CLT UCL	831.71214
Adj-CLT UCL (Adjusted for skewness)	877.62411
Mod-t UCL (Adjusted for skewness)	862.21791
Jackknife UCL	855.05657
Standard Bootstrap UCL	822.39299
Bootstrap-t UCL	1152.0822
Hall's Bootstrap UCL	1636.298
Percentile Bootstrap UCL	827.14286
BCA Bootstrap UCL	895.71429
95% Chebyshev (Mean, Sd) UCL	1044.0901
97.5% Chebyshev (Mean, Sd) UCL	1191.68
99% Chebyshev (Mean, Sd) UCL	1481.5919

APPENDIX C

ProUCL Version3.0 MODEL OUTPUT FOR DETERMINATION OF 95-PERCENT UPPER CONFIDENCE LIMIT BENNETT'S DUMP SITE, BLOOMINGTON, INDIANA

Station 3: Maple Grove Road **White Sucker**

Raw Statistics	
Number of Valid Samples	7
Number of Unique Samples	5
Minimum	635
Maximum	1400
Mean	1090.7143
Median	1100
Standard Deviation	243.4548
Variance	59270.238
Coefficient of Variation	0.2232068
Skewness	-0.915771

Gamma Statistics	
k hat	19.917742
k star (bias corrected)	11.476805
Theta hat	54.760942
Theta star	95.036407
nu hat	278.84838
nu star	160.67527
Approx.Chi Square Value (.05)	132.36358
Adjusted Level of Significance	0.01584
Adjusted Chi Square Value	124.59498

Log-transformed Statistics	
Minimum of log data	6.453625
Maximum of log data	7.2442275
Mean of log data	6.9692748
Standard Deviation of log data	0.2546378
Variance of log data	0.0648404

RECOMMENDATION
Data are normal (0.05)

Use Student's-t UCL

Normal Distribution Test	
Shapiro-Wilk Test Statisitic	0.9064279
Shapiro-Wilk 5% Critical Value	0.803
Data are normal at 5% significance level	

95% UCL (Assuming Normal Distribution)	
Student's-t UCL	**1269.5204**

Gamma Distribution Test	
A-D Test Statistic	0.5405102
A-D 5% Critical Value	0.7066948
K-S Test Statistic	0.2591168
K-S 5% Critical Value	0.3114639
Data follow gamma distribution	
at 5% significance level	

95% UCLs (Assuming Gamma Distribution)	
Approximate Gamma UCL	1324.0108
Adjusted Gamma UCL	1406.5639

Lognormal Distribution Test	
Shapiro-Wilk Test Statisitic	0.8418959
Shapiro-Wilk 5% Critical Value	0.803
Data are lognormal at 5% significance level	

95% UCLs (Assuming Lognormal Distribution)	
95% H-UCL	1361.0077
95% Chebyshev (MVUE) UCL	1552.7142
97.5% Chebyshev (MVUE) UCL	1751.4976
99% Chebyshev (MVUE) UCL	2141.9689

95% Non-parametric UCLs	
CLT UCL	1242.0692
Adj-CLT UCL (Adjusted for skewness)	1208.0372
Mod-t UCL (Adjusted for skewness)	1264.212
Jackknife UCL	1269.5204
Standard Bootstrap UCL	1231.1837
Bootstrap-t UCL	1238.5928
Hall's Bootstrap UCL	1242.8323
Percentile Bootstrap UCL	1214.2857
BCA Bootstrap UCL	1242.8571
95% Chebyshev (Mean, Sd) UCL	1491.8082
97.5% Chebyshev (Mean, Sd) UCL	1665.3619
99% Chebyshev (Mean, Sd) UCL	2006.2745

APPENDIX C

ProUCL Version3.0 MODEL OUTPUT FOR DETERMINATION OF 95-PERCENT UPPER CONFIDENCE LIMIT BENNETT'S DUMP SITE, BLOOMINGTON, INDIANA

Station 3: Maple Grove Road **Creek Chub**

Raw Statistics	
Number of Valid Samples	7
Number of Unique Samples	7
Minimum	435
Maximum	770
Mean	562.14286
Median	560
Standard Deviation	123.41857
Variance	15232.143
Coefficient of Variation	0.2195502
Skewness	0.6835535

Gamma Statistics	
k hat	25.220031
k star (bias corrected)	14.506684
Theta hat	22.289539
Theta star	38.750609
nu hat	353.08044
nu star	203.09358
Approx.Chi Square Value (.05)	171.11197
Adjusted Level of Significance	0.01584
Adjusted Chi Square Value	162.22303

Log-transformed Statistics	
Minimum of log data	6.075346
Maximum of log data	6.6463905
Mean of log data	6.3117995
Standard Deviation of log data	0.2140998
Variance of log data	0.0458387

RECOMMENDATION
Data are normal (0.05)

Use Student's-t UCL

Normal Distribution Test	
Shapiro-Wilk Test Statisitic	0.9259141
Shapiro-Wilk 5% Critical Value	0.803
Data are normal at 5% significance level	

95% UCL (Assuming Normal Distribution)	
Student's-t UCL	**652.78798**

Gamma Distribution Test	
A-D Test Statistic	0.2737309
A-D 5% Critical Value	0.7068523
K-S Test Statistic	0.1883805
K-S 5% Critical Value	0.3114496
Data follow gamma distribution at 5% significance level	

95% UCLs (Assuming Gamma Distribution)	
Approximate Gamma UCL	667.20993
Adjusted Gamma UCL	703.76941

Lognormal Distribution Test	
Shapiro-Wilk Test Statisitic	0.9373299
Shapiro-Wilk 5% Critical Value	0.803
Data are lognormal at 5% significance level	

95% UCLs (Assuming Lognormal Distribution)	
95% H-UCL	672.04653
95% Chebyshev (MVUE) UCL	760.38546
97.5% Chebyshev (MVUE) UCL	846.25258
99% Chebyshev (MVUE) UCL	1014.9219

95% Non-parametric UCLs	
CLT UCL	638.87172
Adj-CLT UCL (Adjusted for skewness)	651.74933
Mod-t UCL (Adjusted for skewness)	654.79662
Jackknife UCL	652.78798
Standard Bootstrap UCL	631.20262
Bootstrap-t UCL	674.91864
Hall's Bootstrap UCL	674.86513
Percentile Bootstrap UCL	635.71429
BCA Bootstrap UCL	667.14286
95% Chebyshev (Mean, Sd) UCL	765.47605
97.5% Chebyshev (Mean, Sd) UCL	853.45849
99% Chebyshev (Mean, Sd) UCL	1026.2829

APPENDIX D

EVALUATION OF RISK ASSOCIATED WITH EXPOSURE TO DIOXIN-LIKE PCB CONGENERS IN FISH TISSUE

(17 Pages)

APPENDIX D

The main text of the focused ecological risk assessment (FERA) for the Bennett Stone Quarry (Bennett's Dump) site evaluates risk to piscivorous wildlife (i.e., mink and kingfisher) associated with exposure to total polychorinated biphenyls (PCB) as Aroclors. This appendix evaluates risk to piscivorous wildlife associated with exposure to dioxin-like PCB congeners and follows the approach developed by U.S. Environmental Protection Agency (EPA) for the FERA conducted for the Neal's Landfill site, as detailed in the *Focused Ecological Risk Assessment, PCBs and Mammalian and Avian Piscivores in Conard's Branch and Richland Creek* (EPA 2005).

The following sections describe the process for evaluating risk associated with dioxin-like congeners at the Bennett's Dump site.

D.1 PROBLEM FORMULATION

The purpose of this FERA is to investigate the protectiveness of the remedial activities conducted at the Bennett's Dump site. The assessment presented in this appendix focuses solely on dioxin-like PCB-related risks to wildlife (specifically, piscivorous [fish-eating] birds and mammals) in Stout's Creek, downstream of the Bennett's Dump site.

Components of the problem formulation, including the conceptual site model, assessment endpoint, and measurement endpoint are the same as detailed in Section 2.0 of the main text, except as noted below.

Assessment Endpoints. Assessment endpoints are as described in Section 2.3 of the main text, that is, the reproductive conditions of piscivorous mammals and birds that inhabit or potentially inhabit Stout's Creek are the assessment endpoints.

Measurement Endpoints. As discussed in Section 2.4 of the main text, two piscivorous measurement endpoints are assessed: mink to represent mammalian piscivores and kingfisher to represent avian piscivores. The measures of effects for mink are studies that identify the reproductive effect levels associated with feeding PCB-contaminated fish to mink (see Section 5.1 of the main text); for the kingfisher, accumulation of dioxin-like PCB congeners in kingfisher eggs was modeled to identify the reproductive effect level (dose) for avian receptors. In addition, the effects of dioxin-like PCB congeners (expressed as a dose) on avian fertility and embryo mortality were also evaluated for the kingfisher. The reproductive effect levels were used to evaluate the level of risk associated with exposure to dioxin-like PCB congeners in the diet.

D.2 EXPOSURE ASSESSMENT

Exposure Assumptions. The assumptions for assessing exposure were as described in Section 3.0 of the main text, with the following exception:

- As no PCB congener analysis was conducted for crayfish samples, no data were available to extrapolate the contribution of crayfish to the dietary composition; therefore, the concentration of PCB congeners in crayfish was assumed to equal the PCB congener concentration in fish. The mink diet was assumed to be composed of 79 percent fish (based on a dietary assumption of 66 percent fish and 13 percent crayfish); the kingfisher diet was assumed to be composed of 100 percent fish (based on a dietary assumption of 80 percent fish and 20 percent crayfish).

Data Collection and Analysis. As discussed in Section 3.3 of the main text, fish tissue samples were collected at three sampling stations in Stout's Creek downgradient of the Bennett's Dump site (see Figure 3 of the main text). Fish tissue samples were analyzed for total PCBs as Aroclors. Split samples were collected from a subset of the fish tissue samples and analyzed for dioxin-like PCB congeners.

At each sampling station, three species of fish were collected; one split sample representing each species was submitted for PCB congener analysis at each station; the sample identifications and species submitted for congener analysis are presented below.

Sample ID	Station	Species	Number of Samples
58	1	Green sunfish	1
65	1	White sucker	1
67	1	Creek chub	1
36	2	Green sunfish	1
48	2	White sucker	1
42	2	Creek chub	1
12	3	Longear sunfish	1
24	3	White sucker	1
17	3	Creek chub	1

Total PCB results for these split samples are presented in Table A-1 of Appendix A; results of the congener analyses are presented in Table D-1. This appendix presents an evaluation of risk associated with exposure to dioxin-like PCB congeners in fish tissue collected from Stout's Creek.

Calculation of Exposure Point Concentrations. As only one sample was submitted for congener analysis per species at each station, no summary statistics could be calculated; that is, a mean or 95-percent upper confidence limit (95UCL) on the arithmetic mean could not be calculated.

In order the assess exposure to the dioxin-like PCB congeners, the PCB congener data was transformed to dioxin equivalent concentrations. Prior to performing the data transformations, non-detect values in the PCB congener data set were addressed. Non-detect values were set equal to one-half the sample quantitation limit; congener 169 had the only reported non-detect results (see Table D-1). The congener data for each fish sample were converted to World Health Organization (WHO) toxic equivalent concentrations (TEC) for mammals and birds according to Van den Berg and others 1998.

TABLE D-1

PCB CONGENER CONCENTRATIONS IN WHOLE FISH SAMPLES
BENNETT'S DUMP SITE, BLOOMINGTON, INDIANA

Station	Species	May 2004 - Concentrations in pg/g											Mammalian WHO TEQ (pg/g)
		77 TeCB	81 TeCB	105 PeCB	114 PeCB	118 PeCB	123 PeCB	126 PeCB	167HxCB	156/157 HxCB	169 HxCB[a]	189 HpCB	
1	GS	8,540	491	51,200	3,270	95,500	2,930	199	717	2,190	3.79	57.8	38.55
1	WS	20,500	1,380	95,600	6,420	185,000	5,590	358	1,320	3,740	6.1	90.7	71.77
1	CRC	9,870	512	28,100	1,880	63,100	2,220	138	435	1,140	2.23	30.4	25.72
2	GS	2,110	139	15,800	1,160	30,200	1,270	76.4	331	962	2.13	26.8	13.68
2	WS	7,910	619	53,800	3,340	106,000	3,420	207	815	2,220	3.57	62.1	40.70
2	CRC	5,410	290	20,200	1,390	46,300	1,640	98.4	394	1,070	2.04	29.3	18.48
3	LS	3,980	17,800	1160	42,400	1,440	62.7	413	907	1.74	37.1	37.1	65.33
3	WS	3,500	232	16,500	1,100	38,600	1,240	70	371	865	1.9	31.5	14.02
3	CRC	2,830	152	10,500	610	20,200	799	46.2	218	477	1.96	19.4	8.64
Mammalian TEF		**0.0001**	**0.0001**	**0.0001**	**0.0005**	**0.0001**	**0.0001**	**0.1**	**0.00001**	**0.0005**	**0.01**	**0.0001**	--

Notes:

[a] Results shown represent one-half the sample quantitiation limit; all results for congener 169 were non-detect.

Station — Fish sample collection station (see Figure 3 of the main text)
- 1 — Hunter Valley Road
- 2 — Acuff Road
- 3 — Maple Grove Road

Species — Sampled fish species
- CRC — Creek chub
- GS — Green sunfish (Station 1 and 2 only)
- LS — Longear sunfish (Station 3 only)
- WS — White sucker

TEQ = $\sum n\,[PCB_i \times TEF_i]$

where,

PCB_i = Concentration of the ith PCB congener

TEF_i = Mammalian toxic equivalency factor for the ith PCB congener

Mammalian TEFs as listed in Van den Berg and others 1998.

Definitions:

HpCB	Heptachlorobiphenyl
HxCB	Hexachlorobiphenyl
PCB	Polychlorinated biphenyl
PeCB	Pentachlorobiphenyl
pg/g	Picogram per gram
TeCB	Tetrachlorobiphenyl
TEF	Toxic equivalency factor
TEQ	Toxic equivalency quotient
WHO	World Health Organization

To calculate the TEC for mammals (i.e., mink), the dioxin-like PCB congener concentration was multiplied by its corresponding mammalian toxic equivalency factor (TEF), which is the relative potency of that congener relative to the dioxin congener 2,3,7,8-tetrachloro-dibenzo-p-dioxin (or TCDD). After each congener was multiplied by its respective TEF, the products were summed to derive the TEC for the sample. Mammalian TEFs and TECs are presented in Table D-1.

A similar process was followed to convert modeled dioxin-like congener concentrations in kingfisher eggs to TECs for birds, where the TEFs are based on relative toxicity in bird eggs. Prior to calculating the TECs for bird eggs, the PCB congener data was lipid-normalized. PCB congener concentrations were divided by the percent lipid in the sample, which results in the concentration of individual congeners in the fat tissues of the animal that was sampled. Lipid-normalized PCB congener concentrations are presented in Table D-2. After the data was lipid normalized, the lipid-normalized congener concentrations were multiplied by a diet-to-egg biomagnification factor (BMF) to derive the lipid-normalized concentration in eggs; the concentration in eggs were then multiplied by their respective avian TEFs to calculate the TEC for each sample. Lipid-normalized diet-to-egg BMFs are available for dioxin-like congeners 77, 105, 118, 126, and 169 (Blankenship and Geisy 2002, as cited in EPA 2005); therefore, congeners 81, 114, 123, 156/157, 167, and 189 were not modeled. However, these congeners have very small TEFs and would contribute only a small portion of the total TEC. Diet-to-egg BMFs, avian TEFs, and TECs are presented in Table D-3.

To calculate the TEC for avian receptors (i.e., kingfisher), the dioxin-like PCB congener concentration was multiplied by its corresponding mammalian toxic equivalency factor (TEF). The available avian TEFs (i.e., those listed in Van den Berg and others 1998) are based on toxicological studies in which eggs were dosed via injection; therefore, the WHO mammalian TEFs were used to calculate a toxic equivalency quotient (TEQ) ingestion dose for kingfisher. Mammalian TEFs have been used historically to estimate TEQ ingestion doses for avian receptors. Relative potencies for ethoxyresorufin-O-deethylase (EROD) induction for non-ortho PCBs (and also dioxins/furans) in birds have been found to be in the same order of magnitude as those reported from mammalian systems (Van den Berg and others 1998). As described above for the mink, each congener was multiplied by its respective TEF and the products were summed to derive the TEC for the sample. The mammalian TEFs and TECs, which were used to calculate an exposure point concentration for the kingfisher, are presented in Table D-1.

TABLE D-2

LIPID-NORMALIZED PCB CONGENER CONCENTRATIONS IN WHOLE FISH SAMPLES
BENNETT'S DUMP SITE, BLOOMINGTON, INDIANA

Station	Species	Percent Lipid	May 2004 - Concentrations in ug/kg Lipid										
			77 TeCB	81 TeCB	105 PeCB	114 PeCB	118 PeCB	123 PeCB	126 PeCB	167HxCB	156/157 HxCB	169 HxCB	189 HpCB
1	GS	2.48	344.35	19.80	2,064.52	131.85	3,850.81	118.15	8.02	28.91	88.31	0.15	2.33
1	WS	5.78	354.67	23.88	1,653.98	111.07	3,200.69	96.71	6.19	22.84	64.71	0.11	1.57
1	CRC	6.7	147.31	7.64	419.40	28.06	941.79	33.13	2.06	6.49	17.01	0.03	0.45
2	GS	1.79	117.88	7.77	882.68	64.80	1,687.15	70.95	4.27	18.49	53.74	0.12	1.50
2	WS	4.62	171.21	13.40	1,164.50	72.29	2,294.37	74.03	4.48	17.64	48.05	0.08	1.34
2	CRC	3.17	170.66	9.15	637.22	43.85	1,460.57	51.74	3.10	12.43	33.75	0.06	0.92
3	LS	6.29	63.28	282.99	18.44	674.09	22.89	1.00	6.57	14.42	0.03	0.59	0.59
3	WS	6.83	51.24	3.40	241.58	16.11	565.15	18.16	1.02	5.43	12.66	0.03	0.46
3	CRC	4.71	60.08	3.23	222.93	12.95	428.87	16.96	0.98	4.63	10.13	0.04	0.41

Notes:

Station — Fish sample collection station (see Figure 3 of the main text)
- 1 — Hunter Valley Road
- 2 — Acuff Road
- 3 — Maple Grove Road

Species — Sampled fish species
- CRC — Creek chub
- GS — Green sunfish (Station 1 and 2 only)
- LS — Longear sunfish (Station 3 only)
- WS — White sucker

Lipid-normalized PCB congener concentration (ug kg) = PCB concentration (pg g) (Percent lipids x 10)

Definitions:

- HpCB — Heptachlorobiphenyl
- HxCB — Hexachlorobiphenyl
- PCB — Polychlorinated biphenyl
- PeCB — Pentachlorobiphenyl
- pg g — Picogram per gram
- TeCB — Tetrachlorobiphenyl
- ug kg — Microgram per kilogram

TABLE D-3

MODELED (LIPID-NORMALIZED) PCB CONGENER CONCENTRATIONS IN KINGFISHER EGGS
BENNETT'S DUMP SITE, BLOOMINGTON, INDIANA

Station	Species	May 2004 - Concentrations in ug/kg-lipid											Avian WHO TEQ
		77 TeCB	81 TeCB	105 PeCB	114 PeCB	118 PeCB	123 PeCB	126 PeCB	167HxCB	156/157 HxCB	169 HxCB	189 HpCB	
1	GS	306.48	--	16,412.90	--	100,698.59	--	238.64	--	--	4.78	--	41.84
1	WS	315.66	--	13,149.13	--	83,698.10	--	184.20	--	--	3.30	--	36.36
1	CRC	131.11	--	3,334.25	--	24,627.84	--	61.26	--	--	1.04	--	13.26
2	GS	104.91	--	7,017.32	--	44,118.99	--	126.93	--	--	3.72	--	19.09
2	WS	152.38	--	9,257.79	--	59,997.84	--	133.25	--	--	2.41	--	22.47
2	CRC	151.89	--	5,065.93	--	38,193.85	--	92.32	--	--	2.01	--	17.72
3	LS	56.31	--	146.61	--	598.66	--	195.27	--	--	18.43	--	22.38
3	WS	45.61	--	1,920.57	--	14,778.77	--	30.48	--	--	0.87	--	5.67
3	CRC	53.48	--	1,772.29	--	11,215.07	--	29.17	--	--	1.30	--	5.88
Diet-Egg BMF		**0.89**	**--**	**7.95**	**--**	**26.15**	**--**	**29.74**	**--**	**--**	**31.25**	**--**	--
Avian TEF		**0.05**		**0.0001**		**0.00001**		**0.1**			**0.001**		

Notes:

Station — Fish sample collection station (see Figure 3 of the main text)
- 1 — Hunter Valley Road
- 2 — Acuff Road
- 3 — Maple Grove Road

Species — Sampled fish species
- CRC — Creek chub
- GS — Green sunfish (Station 1 and 2 only)
- LS — Longear sunfish (Station 3 only)
- WS — White sucker

TEQ = $\sum n\,[PCB_i \times TEF_i]$

where,

PCB_i = Concentration of the ith PCB congener and TEF_i = Avian toxic equivalency factor for the *i*th PCB congener

Avian TEFs as listed in Van den Berg and others 1998.

Definitions:

- BMF — Biomagnification factor
- HpCB — Heptachlorobiphenyl
- HxCB — Hexachlorobiphenyl
- PCB — Polychlorinated biphenyl
- PeCB — Pentachlorobiphenyl
- TeCB — Tetrachlorobiphenyl
- TEF — Toxic equivalency factor
- TEQ — Toxic equivalency quotient
- WHO — World Health Organization
- ug/kg — Microgram per kilogram

D.3 ECOLOGICAL EFFECTS ASSESSMENT

Mink TEC-based toxicity reference values (TRV) and their derivations are presented in the Attachment, both on a dose and dietary basis. The dietary TRVs selected for this FERA are a no-observed-adverse-effect-concentration (NOAEC)-based TRV of 4.6 picograms per gram (pg/g) and lowest-observed-adverse-effect-concentration (LOAEC)-based TRV of 18 pg/g; these TRVs are multiple season or multiple generation TRVs. Both TRVs were calculated as the geometric means of the TRVs presented in in Brunström and others (2001, as cited in EPA 2005) and Restum and others (1998, as cited in EPA 2005) mink feeding studies, as detailed in Section 4.2.2.2 of EPA 2003.

Two sets of kingfisher egg TEC-based TRVs were selected for use in the FERA as they cluster around two different values (EPA 2005). Derivation of the kingfisher egg TRVs is detailed in Section 4.2.2.5 of EPA 2005. One set of kingfisher egg TRVs (lipid-normalized) were a NOAEC-based TRV of 1.8 microgram per kilogram (μg/kg) lipid and LOAEC-based TRV of 5.3 μg/kg lipid (Blankenship and Geisy 2002, as cited in EPA 2005); the basis of the Blankenship and Geisy (2002) TRVs was a field study of bald eagle eggs (Elliot and others 1996, as cited in EPA 2005). A second set of kingfisher egg TRVs (lipid-normalized) were a NOAEC-based TRV of 17 μg/kg lipid and a LOAEC-based TRV of 68 μg/kg lipid selected from the Hudson River ERA. The Hudson River TRVs are an order of magnitude greater (i.e., 10 times greater) than other available TRVs (see the Attachment) and are based on embryo mortalilty in double-crested cormorant eggs injected with TCDD (Powell and others 1977, as cited in EPA 2005), hatchability of bald eagle eggs (Elliot and others 1996, as cited in EPA 2005), and embryo mortality of kestrel eggs injected with PCB congener 77 (Hoffman and others 1998, as cited in EPA 2005).

Avian dose-based TEC TRVs and their derivations are presented in the Attachment (see also Section 4.2.2.4 of EPA 2005). The dose-based TRVs selected for this FERA are a no-observed-adverse-effect-level (NOAEL)-based TRV of 1.4 nanogram per kilogram body weight per day (ng/kg-day) and a lowest-observed-adverse-effect-level (LOAEL)-based TRV of 14 ng/kg-day. These TRVs are based on a study of dioxin in pheasant (Nosek and others 1992 as cited in EPA 2005); the toxicological endpoints are fertility and embryo mortality. The original TRVs from Nosek and others (1992) was adjusted downward by a factor of 10 to account for extrapolation from subchronic exposure (i.e., 10 weeks in the Nosek and others [1992] study) to chronic exposure.

D.4 RISK CHARACTERIZATION RESULTS

This section summarizes the findings of the risk calculations to form conclusions about potential risks posed to the assessment endpoints (piscivorous mammals and birds) identified for the Bennett's Dump site study areas (i.e., Stout's Creek) in the problem formulation phase.

For the FERA, the hazard quotient (HQ; or toxicity quotient) method was used to characterize risk from dioxin-like PCB congeners. In addition, a percent allowable consumption (PAC) estimate was calculated. Calculation of HQ and PAC risk estimates followed the same procedure as described in Sections 4.3 and 4.4 of the main text.

Mink Hazard Quotients: Stations 1 Through 3

PCB congener HQs for Stations 1 through 3 are discussed in the following sections.

Station 1. Station 1 is located at Hunter Valley Road, approximately 1 mile downstream from the Bennett's Dump site (see Figure 3 of the main text). Both the NOAEC-based and LOAEC-based mink HQs were greater than 1. Mink HQs ranged from 2 (LOAEC-based) to 8 (NOAEC-based) (see Table D-4). As the low-effect-based mink HQ exceeds 1, the potential for adverse effects exists for mink with a home range within the Station 1 reach.

Station 2. Station 2 is located at Acuff Road, approximately 3 miles downstream from the Bennett's Dump site (see Figure 3 of the main text).

Mink HQs decreased downstream at Station 2 as compared to the mink HQs estimated for Station 1 (see Table D-4). The NOAEC-based mink HQ was estimated at 4 and the LOAEC-based mink HQ was equivalent to the threshold value of 1. As the low-effect-based mink HQ is equivalent to the threshold value of 1, which indicates the exposure concentration is equal to the concentration show to cause adverse effects in mink.

TABLE D-4

TEQ CONGENER CONCENTRATIONS IN DIET AND STATION-SPECIFIC RISK, MAY 2004 FISH DATA
BENNETT'S DUMP SITE, BLOOMINGTON, INDIANA

				Mink			Lipid-normalized	Kingfisher: Egg-based				Kingfisher: Dose-based			
Station	Species	N	TEQ Concentration in Fish (pg/g)	TEQ Concentration in Diet (pg/g)	HQ NOAEC (ratio)	HQ LOAEC (ratio)	TEQ Concentration in Eggs (ug/kg-lipid)	HQ NOAEC-low (ratio)	HQ NOAEC-high (ratio)	HQ LOAEC-low (ratio)	HQ LOAEC-high (ratio)	TEQ Concentration in Diet (pg/g)	TEQ Concentration in Dose (pg/g-day)	HQ NOAEL (ratio)	HQ (LOAEL) (ratio)
1	GS	1	38.55	35.82	7.79	1.99	41.84	16.94	1.79	5.75	0.45	45.35	22.67	16.19	1.62
1	WS	1	71.77				36.36								
1	CRC	1	25.72				13.26								
2	GS	1	13.68	19.19	4.17	1.07	19.09	10.98	1.16	3.73	0.29	24.29	12.14	8.67	0.87
2	WS	1	40.70				22.47								
2	CRC	1	18.48				17.72								
3	LS	1	65.33	23.17	5.04	1.29	22.38	6.28	0.67	2.13	0.17	29.33	14.66	10.47	1.05
3	WS	1	14.02				5.67								
3	CRC	1	8.64				5.88								

Notes:

Station — Fish sample collection station (see Figure 3 of the main text)
- 1 — Hunter Valley Road
- 2 — Acuff Road
- 3 — Maple Grove Road

Species — Sampled fish species
- CRC — Creek chub
- GS — Green sunfish
- LS — Longear sunfish
- WS — White sucker

TEQ Concentration in Fish: As n = 1 for each species, the fish TEQ concentration, which was calculated using mamalian TEFs, was used.

Lipid-normalized TEQ Concentration in Eggs: As n = 1 for each species, the lipid-normalized TEQ concentration, which was calculated using avian TEFs, was used.

No PCB congener data was available for crayfish samples collected in May 2004; therefore, the TEQ concentration in crayfish was assumed to equal the TEQ concentration in fish.

Mink, Concentration in Diet:	Concentration of TEQs in the mink diet = 0.79 x mean of the fish species concentration (assumes crayfish TEQ = fish tissue TEQ)
Mink, HQ, NOAEC:	HQ = Concentration of TEQs in the mink diet / NOAEC-based TRV, where TRV = 4.6 pg/g
Mink, HQ, LOAEC:	HQ = Concentration of TEQs in the mink diet / LOAEC-based TRV, where TRV = 18 pg/g
Kingfisher, Egg-based HQ, NOAEC-low:	HQ = Mean Concentration of TEQs in kingfisher egg / NOAEC-low-based TRV; where TRV = 1.8 ug/kg lipid-day
Kingfisher, Egg-based HQ, NOAEC-high:	HQ = Mean Concentration of TEQs in kingfisher egg / NOAEC-high-based TRV; where TRV = 17 ug/kg lipid-day
Kingfisher, Egg-based HQ, LOAEC-low:	HQ = Mean Concentration of TEQs in kingfisher egg / LOAEC-low-based TRV; where TRV = 5.3 ug/kg lipid-day
Kingfisher, Egg-based HQ, LOAEC-high:	HQ = Mean Concentration of TEQs in kingfisher egg / LOAEL-high-based TRV; where TRV = 68 ug/kg lipid-day
Kingfisher, Concentration in Diet:	Concentration of TEQs in the kingfisher diet =Mean fish species concentration (assumes crayfish TEQ = fish TEQ)
Kingfisher, Concentration in Dose:	Concentration of TEQs in kingfisher diet (expressed as a dose) = (Concentration of TEQs in the kingfisher diet x IR), where IR = 0.5 kg food per kg body weight
Kingfisher, Dose-based HQ, NOAEL:	HQ = Concentration of TEQs in kingfisher dose / NOAEL-based TRV; where TRV = 1.4 ng/kg-day (or 1.4 pg/g-day)
Kingfisher, Dose-based HQ, LOAEL:	HQ = Concentration of TEQs in kingfisher dose / LOAEL-based TRV; where TRV = 14 ng/kg lipid-day (or 14 pg/g-day)

Definitions:

HQ	Hazard quotient
IR	Ingestion rate (kingfisher)
kg	Kilogram
LOAEC	Lowest-observed-adverse-effect concentration
LOAEL	Lowest-observed-adverse-effect level
N	Number of samples
NOAEC	No-observed-adverse-effect concentration
NOAEL	No-observed-adverse-effect level
PCB	Polychlorinated biphenyl
pg/g	Picogram per gram
pg/g-day	Picogram per gram per day
TEQ	Toxic equivalency quotient
TRV	Toxicity reference value
ug/kg Lipid	Microgram per kilogram lipid
ug/kg Lipid-day	Microgram per kilogram lipid per day

Station 3. Station 3 is located at Maple Grove Road, approximately 5 miles downstream from the Bennett's Dump site (see Figure 3 of the main text).

Mink PCB congener HQs increased slightly at Station 3 compared to those at Station 2; this is largely in part due to the increased percentage of lipids reported for longear sunfish compared to green sunfish for the May 2004 sampling event (see Tables D-2 and D-4). Longear sunfish were only collected at Station 3; no green sunfish were collected at Station 3. The NOAEC-based mink HQ was estimated at 5, which exceeds the threshold of 1; the LOAEC-based mink HQ was estimated at 1, which is equivalent to the threshold value and indicates the exposure concentration is equal to the concentration shown to cause adverse effects in mink.

Kingfisher Egg-based Hazard Quotients: Stations 1 through 3

Kingfisher egg-based HQs for Stations 1 through 3 are discussed in the following sections.

Station 1. Station 1 is located at Hunter Valley Road, approximately 1 mile downstream from the Bennett's Dump site (see Figure 3 of the main text).

Both of the NOAEC-low-based and LOAEC-low-based kingfisher egg HQs were greater than 1 (HQs of 17 and 6, respectively). The kingfisher egg HQs calculated using the higher TRVs from the Hudson Valley ERA (NOAEC-high and LOAEC-high) were either below 1 (HQ = 0.45, LOAEC-high) or greater than 1 (HQ = 2; NOAEC-high) (see Table D-4). Although the LOAEC-high-based HQ is less than 1 (HQ = 0.45), the LOAEC-low-based HQ is estimated at 6; therefore, exposure concentrations within the Station 1 reach may be within the range shown to cause adverse effects in kingfisher.

Station 2. Station 2 is located at Acuff Road, approximately 3 miles downstream from the Bennett's Dump site (see Figure 3).

Kingfisher egg HQs decreased downstream at Station 2 as compared to the kingfisher egg HQs estimated for Station 1 (see Table D-4). As the LOAEL-based kingfisher HQ is below 1, the potential for adverse effects is low for kingfisher with a home range within the Station 2 reach is low.

Station 3. Station 3 is located at Maple Grove Road, approximately 5 miles downstream from the Bennett's Dump site (see Figure 3 of the main text).

Kingfisher egg HQs decreased downstream at Station 3 as compared to the kingfisher HQs estimated for Stations 1 and 2 (see Table D-4). The NOAEC-high- and LOAEC-high-based HQs were below the threshold of 1. The NOAEC-low-based HQ exceeded the threshold of 1 with an estimated value of 6. Although both the LOAEC-high- and NOAEC-high-based HQs are less than 1 (HQs = 0.17 and 0.67, respectively); the LOAEC-low-based is only greater than 1 (HQ = 2), indicating exposure concentrations may be within the range shown to cause adverse effects in avian receptors.

Kingfisher Dose-based Hazard Quotients: Stations 1 through 3

Kingfisher dose-based HQs for Stations 1 through 3 are discussed in the following sections.

Station 1. Station 1 is located at Hunter Valley Road, approximately 1 mile downstream from the Bennett's Dump site (see Figure 3 of the main text).

Both of the NOAEL-based and LOAEL-based kingfisher dose-based HQs were greater than 1 (HQs of 16 and 2, respectively). As the LOAEL-based HQ is greater than 1, there is a potential risk for a kingfisher within the Station 1 reach.

Station 2. Station 2 is located at Acuff Road, approximately 3 miles downstream from the Bennett's Dump site (see Figure 3).

Kingfisher dose-based HQs decreased downstream at Station 2 as compared to the kingfisher dose-based HQs estimated for Station 1 (see Table D-4). Although the NOAEL-based HQ was greater than 1 (HQ

= 9), the LOAEL-based HQ was below the threshold of 1 (HQ = 0.87). Although the LOAEC-high-based HQ is less than 1 (HQ = 0.29); the LOAEC-low-based HQ is estimated at 4, indicating the exposure concentration may be within the range shown to cause adverse effects in avian receptors.

Station 3. Station 3 is located at Maple Grove Road, approximately 5 miles downstream from the Bennett's Dump site (see Figure 3 of the main text).

Kingfisher dose-based HQs increased slightly at Station 3 compared to those at Station 2; as mentioned above for mink, this is largely in part due to the increased percentage of lipids reported for longear sunfish compared to green sunfish for the May 2004 sampling event (see Tables D-2 and D-4). Longear sunfish were only collected at Station 3; no green sunfish were collected at Station 3. The NOAEL-based HQ was estimated at 10, which exceeds the threshold of 1; the LOAEL-based HQ was estimated at 1, which is equivalent to the threshold value and indicates the exposure concentration is equal to the concentration shown to cause adverse effects in avian receptors. As the LOAEL-based HQ is equivalent to 1, there is a potential for adverse effects for kingfisher with a home range within the Station 3 reach.

D.5 PERCENT ALLOWABLE CONSUMPTION

The PAC results for the PCB congener assessment are summarized in Table D-5.

The Station 1 PAC assumes that the only exposure mink or kingfisher have to PCBs released from the Bennett's Dump site is through consumption of Stout's Creek fish. The PAC values therefore represent the percentage of the mink or kingfisher diet taken from Station 1 that would result in a NOAEC- or LOAEC-based HQ of 1. A NOAEC-based HQ of 1 is not associated with adverse effects; however, a LOAEC-based HQ of 1 indicates that the exposure is equivalent to the lowest concentration associated with potential adverse effects.

The Station 1 NOAEC-based PAC for mink is estimated at 13 percent (see Table D-5). The mink LOAEC-based PAC is 50 percent. The results indicate that, to stay within no-effect dietary concentrations, mink should forage along the Station 1 reach for no more than 13 percent of the total

TABLE D-5

PERCENT ALLOWABLE CONSUMPTION FOR PCB CONGENERS (EXPRESSED AS TEQs), MAY 2004 FISH DATA
BENNETT'S DUMP SITE, BLOOMINGTON, INDIANA

Station	Mink		Kingfisher					
	Dietary Basis		Egg-based				Dose-based	
	NOAEC PAC	LOAEC PAC	NOAEC-low PAC	NOAEC-high PAC	LOAEC-low PAC	LOAEC-high PAC	NOAEL PAC	LOAEL PAC
1	12.84	50.25	5.90	55.76	17.38	223.05	6.17	61.75
2	23.97	93.81	9.11	86.04	26.82	344.16	11.53	115.28
3	19.85	77.69	15.91	150.30	46.86	601.19	9.55	95.48
TRV	4.6 pg/g	18 pg/g	1.8 ug/kg lipid	17 ug/kg lipid	5.3 ug/kg lipid	68 ug/kg lipid	1.4 pg/g-day	14 pg/g-day

Notes:

Station — Fish sample collection station (see Figure 3 of the main text)
- 1 — Hunter Valley Road
- 2 — Acuff Road
- 3 — Maple Grove Road

PAC (Percent Allowable Consumption, %) = (1 / HQ) x 100; the PAC is the percent of an animal's diet that can be consumed within a station reach and not exceed the TRV.

% — Percent

HQ — Hazard quotient; where mink HQ = concentration in mink diet / TRV and
mink HQ = TEC concentration in mink diet / TRV
kingfisher egg-based HQ = lipid-normalized TEC concentration in kingfisher egg / TRV
kingfisher dose-based HQ = TEC concentration in kingfisher dose / TRV

LOAEC — Lowest-observed-adverse-effect concentration

LOAEL — Lowest-observed-adverse-effect level

NOAEC — No-observed-adverse-effect concentration

NOAEL — No-observed-adverse-effect level

PCB — Polychlorinated biphenyl

pg/g — Picogram per gram

pg/g-day — Picogram per gram per day

TEQ — Toxic equivalent concentration

TRV — Toxicity reference value; see Section 4.2.2 of the main text and Section D.3.

ug/kg lipid-day — Microgram per kilogram per day diet, and that potentially adverse effects are possible if greater than approximately 50 percent of the total diet comes from the Station 1 reach.

Station 1 is located downstream of Bennett's Dump site. It is described as being 8 to 10 feet wide at bank full with an average depth of 1 to 2 feet. Some runs and pools may be up to 3 feet deep. Evidence of abundant wildlife was noted (Tetra Tech 2004), including deer, opossum, raccoons, beavers, rabbits, moles, and mice. Numerous bird species were observed during the May 2004 sampling event, including herons, hawks, turkey vultures, crows, and songbirds.

Mink PACs only account for 79 percent of the mink diet; the remaining 21 percent of the diet was assumed to have no PCB contamination. To calculate the amount of aquatic prey that can be consumed from Station 1 to stay within the PAC amount, the LOAEC-based PAC was multiplied by 79 percent. This results in an adjusted mink PAC of 40. If a typical mink is assumed to have a food ingestion rate of 160 grams per day and a body weight of 1 kilogram (Bleavins and Aulerich 1981, as cited in EPA 1993), this is equivalent to approximately 64 grams of fish from Station 1. The mean weight for fish caught at Station 1 is approximately 38 grams; therefore, at least 1 fish caught at Station 1 could be consumed per day (or 1.7 fish per day).

The kingfisher has a diet that is composed of 100 percent aquatic (80 percent fish and 20 percent crayfish). As mentioned previously, as no PCB congener data was available for the crayfish samples, the concentrations in crayfish were assumed to equal the concentration in fish. Kingfisher NOAEC-low-/NOAEC-high-based PACs protective of eggs range from 6 to over 56 percent; LOAEC-low-/LOAEC-high-based PACs range from 17 to over 100 percent at Station 1. To calculate the amount of fish from Station 1 required to reach the LOAEC-low, a kingfisher food ingestion rate of 0.5 grams per body weight per day (EPA 1993) with a kingfisher body weight of 147 grams were assumed, resulting in an egg LOAEC-low-based PAC of 10 grams of aquatic prey. As the average fish weight from Station 1 is 38 grams, only a portion of 1 fish (i.e., no more than one-third of a fish or 1 fish per 4 days) caught at Station 1 could be consumed.

PACs calculated for the adult kingfisher based on the dose administered through fish consumption ranged from 6 to 62 percent for the NOAEL- and LOAEL-based PACs for Station 1. Using the kingfisher food ingestion rate of 0.5 grams per body weight per day and a kingfisher body weight of 147 grams, approximately 36 grams of fish must be consumed to reach the LOAEL-based PAC. As the mean weight for fish caught at Station 1 is approximately 38 grams, approximate one fish per day may be consumed within the Station 1 reach.

As shown in Table D-5, the PACs at Stations 2 and 3 increase, which is expected with decreasing PCB congener concentrations in fish collected in the downstream reaches. The Station 1 evaluation represents the conservative evaluation; the allowable amount of fish that may be consumed in the downstream reaches will increase to reflect the increased PACs. For the mink, the LOAEC-based PAC at Station 2 nears 100 percent at Station 2 and decreased approximately 75 percent at Station 3. For the kingfisher, the egg-based LOAEC-low and LOACE-high PACs range from 27 to greater than 100 at the Station 2 and 3 reaches, which reflects the wide range in egg-based TRVs. For the kingfisher, the LOAEL-based PACs approach 100 percent for the Station 2 and Station 3 reaches.

D.6 UNCERTAINTY ANALYSIS

The purpose of the uncertainty analysis is to (1) provide risk managers with a summary of those factors that significantly influence the risk results and (2) assess the contribution of these factors to the under- or overestimation of risk.

Virtually every step in the ERA process requires numerous assumptions, all of which contribute to uncertainty in the risk evaluation. In the absence of empirical or site-specific data, assumptions are developed based on best estimates of data quality, exposure parameters, and dose-relationships.

In addition to the uncertainties detailed in Section 6 of the main text, the primary uncertainty associated with the assessment of risk for exposure to dioxin-like PCB congeners at Stout's Creek is the limited PCB congener data. Only one sample per species was submitted for congener analysis; therefore, no summary statistics could be performed to approximate a representative dioxin-like PCB concentration for each fish species collected at a reach. With only one data point per species, it was assumed that the available data was representative of the dioxin-like PCB congener concentrations in all fish species within a given reach. This may have resulted in an under- or overestimation of risk. In addition, no congener analysis was conducted for crayfish samples; therefore, crayfish were not included as a component in the diet for the evaluation of dioxin-like PCB risks. For the FERA, PCB congener concentrations in crayfish were assumed to be equal to the concentration in fish. This assumption may result in an over- or underestimation of risk depending on actual concentrations.

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ATTACHMENT

APPENDIX E OF *FOCUSED ECOLOGICAL RISK ASSESSMENT, PCBs AND MAMMALIAN AND AVIAN PISCIVORES IN CONARD'S BRANCH AND RICHLAND CREEK* (EPA 2005)

(13 Pages)

Appendix E

Comparison of PCB Toxicity Reference Values in Recent Ecological Risk Assessments

Appendix E3. Avian PCB Dose Toxicity Reference Values (TRVs) (mg/kg_{BW}-d) in Recent Ecological Risk Assessments

Site or Application - Receptor	NOAEL	LOAEL	Endpoint	Contaminant Source	Test Species	UF	Reference
Great Lakes Initiative (GLI) Water Quality Criteria - belted kingfisher, bald eagle, herring gull	0.2	0.6 [a]	hatchability	product (A1254)	pheasant	0.33 (inter-specific); 0.33 (LOAEL-to-NOAEL)	Dahlgren et al 1972
Fox River, WI - piscivorous and carnivorous birds	0.11	1.12	courtship and nesting behaviors	product (A1254)	ring dove; mourning dove	0.1 (LOAEL-to-NOAEL)	Peakall and Peakall 1973; Tori and Peterle 1983
Hudson River, NY - belted kingfisher, great blue heron, bald eagle	1.8	7.1	egg production	product (A1254)	pheasant	none	Dahlgren et al 1972
Kalamazoo River, MI	0.4 [b]	0.5 [c]	hatchability	product (A1248)	chicken	none	Lillie et al 1974, 1975; Cecil et al 1974; Scott 1977
Sheboygan River and Harbor, WI, aquatic ERA - great blue heron	0.046	0.4	hatchability; deformity	field	chicken	none	Summer et al 1996
Sheboygan River and Harbor, WI, terrestrial ERA - robin	0.042	0.36	hatchability; deformity	field	chicken	none	Summer et al 1996
Upper Green Bay, WI - caspian tern, double-crested cormorant	0.11	1.12	courtship and nesting behaviors	product (A1254)	ring dove; mourning dove	0.1 (LOAEL-to-NOAEL)	Peakall and Peakall 1973; Tori and Peterle 1983

a) The GLI calculated water quality criteria solely on a NOAEL basis, so did not evaluate the appropriateness of using the LOAEL by itself for decision making. In this case, the LOAEL served as a starting point for calculating the NOAEL.

b) Interpolated 10 % effective dose (ED_{10}), the dose associated with a 10 % decrement in reproductive endpoints compared to control values based on combined dose-response data (Appendix D of the Kalamazoo ERA).

c) Interpolated 25 % effective dose (ED_{25}), the dose associated with a 25 % decrement in reproductive endpoints compared to control values based on combined dose-response data (Appendix D of the Kalamazoo ERA).

Other definitions as in Appendix E1

Appendix E4. Avian Dioxin Toxic Equivalent Concentration (TEC) Dose Toxicity Reference Values (TRVs) (µg/kg$_{BW}$-d) in Recent Ecological Risk Assessments

Site or Application - Receptor	NOAEL	LOAEL	Endpoint	Contam Source	Test Species	UF	TEF/TEC	Reference
GLI Water Quality Criteria - belted kingfisher, bald eagle, herring gull	0.0014	0.014	fertility; embryo mortality	TCDD	pheasant	0.1 (subchronic-to-chronic)	1 (by definition)	Nosek et al 1992a
Hudson River, NY - belted kingfisher, great blue heron, bald eagle	0.0014	0.014	fertility; embryo mortality	TCDD	pheasant	0.1 (subchronic-to-chronic)	1 (by definition)	Nosek et al 1992a
Sheboygan River and Harbor, WI, aquatic ERA - great blue heron	0.0029	0.028	hatchability; deformity	field	chicken	none	recalculated with TEFs from Kennedy et al 1996	Summer et al 1996
Sheboygan River and Harbor, WI, terrestrial ERA - robin	0.00144	0.00323	hatchability; deformity	field	chicken	none	HII4E bioassay [a]	Summer et al 1996
	0.014	0.14	hen and embryo mortality	TCDD	pheasant	none	1 (by definition)	Nosek et al 1992a
Revised Summer, et al. (1996) TRVs	0.0014	0.012	hatchability; deformity	field	chicken	none	recalculated with mammalian TEFs from Van den Berg et al. 1998 [b]	Summer et al 1996

a) TEC as measured by the H4IIE rat hepatoma cell line bioassay (Tillitt, et al. 1996).

b) TEC is calculated from the following regression of total PCB (mg/kg) and TEC (pg/g) (data from Tillitt, et al. 1996 recalculated with mammalian WHO-TEFs): TEC = (32.594 * PCB) - 1.577 $r^2 = 0.99$, $p = 0.005$, for a PCB range 0.015–2.56 mg/kg. The Heaton, et al. (1995) and Summer, et al. (1996) studies were performed with the same collection of field-contaminated prey (homogenized and frozen in large batches for multiple feeding studies), so the PCB-TEC regression for the feed used by Heaton, et al. (1995), as reported by Tillitt, et al. (1996), applies to the Summer, et al. (1996) treatments as well. The avian TEFs reported by Van den Berg, et al. (1998) are not used for recalculating the Summer, et al. (1996) TECs because the avian TEFs are based on egg studies, which means that congeners should be modeled or measured in eggs to apply the avian TEFs. Summer, et al. (1996) did not report egg TECs, only the dietary TECs as determined by the HII4E bioassay performed with a rat hepatoma cell line. Since Summer, et al. (1996) originally reported the dietary TECs on a mammalian basis, and because ingestion-based avian TEFs are unavailable, the dietary TEC is recalculated with the updated mammalian TEFs. Dietary TEC is converted to dose by converting to µg/kg and multiplying by the mean daily food consumption over the exposure period (weeks 3–10): 0.0548 kg/kg-d (high-dose treatment) and 0.0553 kg/kg-d (low-dose treatment). An additional uncertainty for the high-dose treatment is that the dietary PCB concentration (6.6 mg/kg) is greater than the highest PCB concentration (2.56 mg/kg) reported by Tillitt, et al. (1996) in the data used to develop the PCB-TEC regression.

Other definitions as in Appendices E1 and E2

Appendix E5. Avian PCB Egg Toxicity Reference Values (TRVs) in Recent Ecological Risk Assessments

Site or Application - Receptor	NOAEC		LOAEC		Endpoint	Contam Source	Test Species	UF	Reference
	mg/kg	mg/kg lipid	mg/kg	mg/kg lipid					
Fox River, WI - all birds	4.7	49 [a]	7.6	80 [a]	hatchability	field	common tern	none	Hoffman et al 1993
Fox River, WI - all birds	0.8	14 [b]	8	136 [b]	deformity	field	double-crested cormorant	10 (NOAEC-to-LOAEC)	Ludwig et al 1996
Hudson River, NY - belted kingfisher	4.7	49 [a]	7.6	80 [a]	hatchability	field	common tern	none	Hoffman et al 1993
Hudson River, NY - great blue heron	2	32 [c]	7.6	80 [a]	hatchability	field	GBH (NOAEC); common tern (LOAEC)	none	Halbrook et al 1999 (NOAEC); Hoffman et al 1993 (LOAEC)
Hudson River, NY - bald eagle	5.5	73 [d]	8.7	116 [d]	nest success	field	bald eagle	none	Wiemeyer et al 1993
Kalamazoo River, MI - general	1	8.9 [e]	1.5	13.4 [e]	hatchability	product (A1242)	chicken	none	Britton and Huston 1973
Kalamazoo River, MI - great blue heron	5.8 [f]	61 [a]	20.6 [g]	217 [a]	hatchability, population size or reproductive success	field	Forster's tern	none	Barron et al 1995
Kalamazoo River, MI - bald eagle	1.5	20 [d]	7.7 [h]	103 [d]	hatchability, nesting success, population size or reproductive success	field	bald eagle	0.2 (LOAEC-to-NOAEC) [i]	Wiemeyer et al 1984; various secondary sources [h]

Appendix E5. Avian PCB Egg Toxicity Reference Values (TRVs) in Recent Ecological Risk Assessments

Site or Application - Receptor	NOAEC		LOAEC		Endpoint	Contam Source	Test Species	UF	Reference
	mg/kg	mg/kg lipid	mg/kg	mg/kg lipid					
Kalamazoo River, MI - robin	2.8 [j]	25 [e]	6.2 [k]	55 [e]	hatchability, egg production, fertility, deformity	product (A1242, A1248, A1254) (Barron); field (Summer)	chicken	none	Barron et al 1995; Summer et al 1996
Sheboygan River and Harbor, WI, terrestrial ERA - robin	5	45 [e]	24	214 [e]	hatchability; deformity	field	chicken	none	Summer et al 1996
Upper Green Bay, WI - caspian tern, double-crested cormorant	4.7	49 [a]	7.6	80 [a]	hatchability, deformity	field	common tern	none	Hoffman et al 1993
USEPA Region 5 proposed	0.7	6 [e]	1.3	12 [e]	hatchability	product (A1248)	chicken	none	Lillie et al 1974; Cecil et al 1974; Scott 1977

a) Tern egg lipid content of 9.5 % (semi-precocial) (Carey, et al. 1980)
b) Cormorant egg lipid content of 5.9 % (altricial) (Carey, et al. 1980)
c) Heron egg lipid content of 6.3 % (semi-altricial) (Carey, et al. 1980)
d) Bald eagle egg lipid content of 7.5 % (Blankenship and Giesy 2002)
e) Chicken egg fat content of 11.2 % (5.6 g fat in 50 g egg - Pennington and Church 1985)
f) Arithmetic mean NOAEC: 4.5 mg/kg (Kubiak, et al. 1989 as cited in Barron, et al. 1995) and 7.0 mg/kg (Bosveld and Van den Berg, in press as cited in Barron, et al. 1995). Note: the article as published by Bosveld and Van den Berg (1994) show an egg PCB NOAEC of 2.3 mg/kg for Forster's tern hatching success based on King, et al. (1991).
g) Arithmetic mean LOAEC: 22.2 mg/kg (Kubiak, et al. 1989 as cited in Barron, et al. 1995) and 19 mg/kg (Bosveld and Van den Berg, in press as cited in Barron, et al. 1995). Note: the published article by Bosveld and Van den Berg (1994) show an egg PCB LOAEC of 19.2 mg/kg for Forster's tern hatching success based on Kubiak, et al. (1989).
h) Arithmetic mean LOAEC: 4.0 mg/kg (Wiemeyer, et al. 1984 as derived by Ludwig, et al. 1993 as cited by Stratus 1999), 4.0 mg/kg (Wiemeyer, et al. 1984 as derived by Ludwig, et al. 1993 as cited by Barron, et al. 1995), 4.5 mg/kg (40 % decrement in productivity, Wiemeyer, et al. 1984), 13 mg/kg (unsuccessful nests, Wiemeyer, et al. 1984), 13 mg/kg (reproductive success, Wiemeyer, et al. 1984 as cited by Bosveld and Van den Berg 1994). Note: all of the values are estimates of the LOAEC of the same study.
i) Table 4-9 of the Kalamazoo ERA mistakenly states that the NOAEC is "est. from mean LOAEC/10", but the actual divisor was 5.
j) Arithmetic mean NOAEC: 0.36 mg/kg (Scott 1977 as cited by Barron, et al. 1995), 0.95 mg/kg (Britton and Huston 1973 as cited by Barron, et al. 1995), <5 (entered as 5) mg/kg (Platanow and Reinhart 1973 as cited by Barron, et al. 1995), and 5 mg/kg (Summer, et al. 1996).

k) Arithmetic mean LOAEC: 1.5 mg/kg (Britton and Huston 1973 as cited by Barron, et al. 1995) [note: Table 4-9 of the Kalamazoo ERA mistakenly cites this as "Britton 1973"], 2.5 mg/kg (Scott 1977 as cited by Barron, et al. 1995), 3 mg/kg (RCB/Hagler, Bailly, Inc 1994), 4 mg/kg (Tumasonis, et al. 1973 as cited by Barron, et al. 1995), 4.8 mg/kg (RCB/Hagler, Bailly, Inc 1994), 5 mg/kg (Platanow and Reinhart 1973 as cited by Barron, et al. 1995), and 24 mg/kg (Summer, et al. 1996).

Appendix E6. Avian Dioxin Toxic Equivalent Concentration (TEC) Egg Toxicity Reference Values (TRVs) in Recent Ecological Risk Assessments										
Site or Application - Receptor	NOAEC		LOAEC		Endpoint	Contam Source	Test Species	UF	TEFs	Reference
	µg/kg	µg/kg lipid	µg/kg	µg/kg lipid						
Blankenship & Giesy 2002 - bald eagle	0.134	1.79 [a]	0.4	5.33 [a]	P4501A induction	field	bald eagle	none	Recalculated with TEFs from Van den Berg et al 1998	Elliott et al. 1996
Fox River, WI - all birds	0.007	0.04 [b]	0.19 - 0.31 [c]	2.5 - 4.1 [d]	egg lethality	field	wood duck (NOAEC), double-crested cormorant, caspian tern (LOAEC) [e]	unknown (treatment-to-LC_{50}) [f], 0.1 (LC_{50} -to-NOAEC); none [g] (LOAEC)	USEPA 1989 (NOAEC); H4IIE bioassay (LOAEC) [h]	Giesy et al 1995 (NOAEC), Giesy et al 1994a (LOAEC)
Fox River, WI - all birds	0.038	0.6 [i]	0.38	6.4 [i]	deformity	field	double-crested cormorant	10 (NOAEC-to-LOAEC)	H4IIE bioassay	Ludwig et al 1996
Hudson River, NY - belted kingfisher	1	17 [i]	4	68 [i]	embryo mortality	TCDD injection	double-crested cormorant	none	1 (by definition)	Powell et al 1997
Hudson River, NY - great blue heron	0.3	4.8 [j]	0.5	7.9 [j]	chick BW	field	great blue heron	none	Safe et al 1990	Sanderson et al 1994
Hudson River, NY - bald eagle	0.214 [k]	12.8 [l]	5	79 [m]	hatch rate (NOAEC); embryo mortality (LOAEC)	field (NOAEC); PCB 77 injection (LOAEC)	bald eagle (NOAEC); kestrel (LOAEC)	none	Ahlborg et al 1994 (NOAEC); Van den Berg et al 1998 (LOAEC)	Elliot et al 1996 (NOAEC); Hoffman et al 1998 (LOAEC)
Sheboygan River and Harbor, WI, terrestrial ERA - robin	0.08	0.7 [n]	0.16	1.4 [n]	embryo mortality	TCDD injection	chicken	none	1 (by definition)	Powell et al 1996

a) Bald eagle egg lipid content of 7.5 % (Blankenship and Giesy 2002)
b) Wood duck egg lipid content of 18 % (White and Seginak 1994)

c) Lethal concentration to 20 % (LC_{20}) and 30 % (LC_{30}) of eggs based on a linear regression of data on field-exposed double-crested cormorant and caspian tern colonies (Table 9 in Giesy, et al. 1994a) and the NOAEC from Giesy, et al. (1995). Note–the data are incorrectly attributed in the Fox River ERA Figure 6-4 (Giesy and Tillitt are transposed), Table 6-7 (should read "Giesy, et al. 1994b"), and Table 6-5 (should read "derived from Giesy, et al. 1994b and 1995", the regression based on Tillitt, et al. 1992 data was not used in the derivation).

d) Based on the following regression: egg LC_n = 0.006 lipid-normalized TEC (pg/g) + 5.282, $r^2 = 0.99$, $p < 0.01$, where LC_n is the lethal concentration in eggs associated with mortality in n % of eggs. Each egg TEC datum in Table 6-7 of the Fox River ERA was lipid-normalized by species: wood duck (see b), cormorant (see g), and caspian tern (semi-precocial–egg lipid content of 9.5 %, Carey, et al. 1980).

e) The regression includes the wood duck NOAEC value in addition to the cormorant and tern data.

f) Giesy, et al. (1995) wrote: "The LC-50 for wood ducks (*Aix sponsa*) has been reported to be approximately 70 ng TCDD-EQ/kg in the eggs of wood ducks (White and Setinak [*sic*] 1994). If this value is divided by an application factor of 10, the estimated NOAEC for eggs is estimated to be approximately 7 ng TCDD-EQ/kg.". However, LC_{50} values are not presented in White and Seginak (1994) or the companion paper by White and Hoffman (1995). The nearest datum to a LC_{50} is 55 % eggs hatched at >50 ppt TEC (Table 4 in White and Seginak 1994). Giesy, et al. (1995) do not discuss the procedure for deriving a LC_{50} of 70 ppt from White and Seginak (1994).

g) The cormorant and tern data were used without UFs (6 data points). One data point in the regression was derived with UFs (the wood duck NOAEC).

h) Six of the 7 TEC data points used in the regression are based on the HII4E bioassay (Giesy, et al. 1994a), but one datum (the Giesy, et al. 1995 NOAEC derived from White and Seginak 1994) is based on USEPA 1989 TEFs.

i) Cormorant egg lipid content of 5.9 % (altricial) (Carey, et al. 1980)

j) Heron egg lipid content of 6.3 % (semi-altricial) (Carey, et al. 1980)

k) Average of whole egg TECs at Powell River (210 ng/kg) and East Vancouver (217 ng/kg – calculated from 13,000 ng TEC/kg lipid ÷ 60 [yolk lipid concentration-to-whole egg concentration reported by Elliott, et al. 1996]).

l) Average of lipid-normalized egg yolk TECs at Powell River (12,600 ng/kg lipid) and East Vancouver (13,000 ng/kg lipid – estimated from Figure 4 of Elliott, et al. 1996).

m) Falcon egg lipid content of 6.3 % (semi-altricial) (Carey, et al. 1980)

n) Chicken egg fat content of 11.2 % based on 5.6 g fat in a 50-g egg (Pennington and Church 1985)

Appendix E7. Ecological Risk Assessment Sources

Fox - Final Baseline Human Health and Ecological Risk Assessment, Lower Fox River and Green Bay, Wisconsin, Remedial Investigation and Feasibility Study, 12/2002, prepared by The RETEC Group for Wisconsin Dept. of Natural Resources. http://www.dnr.state.wi.us/org/water/wm/lowerfox/rifs/riskassessment.html

Great Lakes Initiative - USEPA. 1995. Great Lakes Water Quality Initiative Criteria Documents for the Protection of Wildlife: DDT, Mercury, 2,3,7,8-TCDD, PCBs. Office of Water. EPA-820-B-95-008.

Hudson - Hudson River PCB Reassessment, Phase 2 Report, Further Characterization and Analysis, 11/2000, prepared by TAMS Consultants and Menzie-Cura Assoc. for USEPA Region 2 and USACE, Kansas City District. http://www.epa.gov/hudson/reports.htm

Housatonic - Bursian, S., R. Aulerich, B. Yamini, and D. Tillitt. 2003. Dietary Exposure of Mink to Fish from the Housatonic River: Effects on Reproduction and Survival. 6/10/03 Revised Final Report. submitted to Weston Solutions, Inc. http://www.epa.gov/ne/ge/thesite/restofriver-reports.html

Kalamazoo - Final (Revised) Baseline Ecological Risk Assessment. Allied Paper, Inc./Portage Creek/Kalamazoo River Superfund Site. 2003. prepared by CDM for Michigan Department of Environmental Quality., and Appendix D. Toxicity Reference Values (TRVs) for Mammals and Birds Based on Selected Aroclors. 3/6/03. memo from James Chapman, ecologist, USEPA Region 5 to Shari Kolak, RPM. USEPA Region 5, Chicago.

Sheboygan aquatic ERA - Sheboygan River and Harbor Aquatic Ecological Risk Assessment, 11/98, prepared by EVS and NOAA for USEPA Region 5. http://response.restoration.noaa.gov/cpr/library/publications.html

Sheboygan terrestrial ERA - Sheyboygan River and Harbor Floodplain Terrestrial Ecological Risk Assessment, Sheboygan, Wisconsin, 11/99, prepared by James Chapman, USEPA ecologist, for USEPA Region 5.

Upper Green Bay - Focused Ecological Risk Assessment. Upper Green Bay Portion of the Fox River Site, Green Bay, WI. 2000. prepared by Mark Sprenger, Nancy Beckham and Karen Kracko, USEPA Environmental Response Team Center, NJ. Appendix C in Final Baseline Human Health and Ecological Risk Assessment, Lower Fox River and Green Bay, Wisconsin Remedial Investigation and Feasibility Study, 2002, Volume 2.

See Section 8 for all other references.