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MEMO

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Date:
July 25, 2008

Subject:
Response to June 13th Request for Additional Information from the Panel

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The purpose of this memorandum is to respond to the Panel's request for additional information, forwarded on June 13, 2008. Specifically, this document addresses the following:

- Michigan State University's (MSU's) shrew data and any report or paper discussing these data
- MSU explanation of the process to choose the toxicity reference values (TRVs) used in the analyses.

A discussion of each of these issues is provided below.

1. Shrew Data Collected by MSU

The MSU evaluation of shrews included the following data collection and analysis:

- A risk evaluation of shrews;
- Measures of Total and Coplanar polychlorinated biphenyls (PCBs) in prey items for the shrew;
- Measures of Total and Coplanar PCBs in whole body shrew tissue; and
- Records of shrew trapping information.

The risk evaluation of shrews was never summarized in a published manuscript; however, a summary of the assessment was submitted to the United States Environmental Protection Agency (USEPA) in November 2001 as part of an ongoing series of progress reports on the status of the MSU studies (Section 4.0 of the Kalamazoo River Area of Concern Draft Interim Ecological Risk Assessment of Former Impoundment Soils, October 5, 2001 Update, included as Attachment 1 of this memo). Also, the results were presented at the 2002 annual meeting of the North American Section of the Society of Environmental Toxicology and Chemistry (SETAC) (Blankenship et al., 2002). A copy of the presentation is included in Attachment 2 of this memo. All of the data used in the assessment are included in the database provided to the Panel at the May 2008 meeting; the complete data packages have been provided to ARCADIS and can be made available to the Panel upon request. The data packages for the shrew tissue and prey tissue contain both Total and Coplanar PCBs. However, at the time that the risk analysis for the shrew was conducted, the coplanar analysis of shrew and prey tissue was not complete. Therefore the analysis conducted includes evaluation of Total PCBs only. A brief overview of the analysis is provided below.

The short-tailed shrew (*Blarina brevicauda*) was selected as a surrogate species for evaluating potential risk to small burrowing terrestrial mammalian species. Given its carnivorous diet consisting largely of soil invertebrates including earthworms, its high ingestion rate, and relatively small home range (varies from 0.39 hectare [ha] to 0.96 ha; Buckner, 1966; Faust et al., 1971), the shrew is likely to be the most exposed terrestrial species in the formerly impounded areas of the Kalamazoo River. This assumption is supported by tissue concentration data reported in Table 1 of Blankenship et al. (2005), which shows that average Total PCBs and toxic equivalents (TEQs) concentrations in shrew tissues from the former Trowbridge Impoundment are approximately an order of magnitude higher than concentrations of PCBs and TEQs in other small mammals from the same area.

Risk to shrews was evaluated based on both dietary and tissue-based exposure estimates derived from Site-specific measures. These estimates were then compared to literature-derived TRVs to calculate hazard quotients (HQs). The dose model used for the dietary assessment used standard risk equations assuming a normalized ingestion rate of 0.629 g/g/d based on the weighted average of data presented in the Wildlife Exposure Factors Handbook (USEPA, 1993). The shrew diet was assumed to be comprised of 90% terrestrial invertebrates and 10% plants. In addition, incidental soil ingestion was assumed to be 2% and included in the dose estimate. At the time the assessment was completed, Total PCB data were not available for terrestrial invertebrates other than earthworms. Therefore, the earthworm data were used to represent all terrestrial invertebrates (i.e., 90% of the diet). This assumption overstates potential exposure as the shrew is an opportunistic feeder and the diet may be composed of a significant portion of litter invertebrates (USEPA, 1993)—litter invertebrates have been demonstrated to contain lower Total PCB concentrations than earthworms at the Site (Table 1 in Blankenship et al., 2005).

Site-specific earthworm data were only available for locations within the former Trowbridge Impoundment and the Ft. Custer Recreation Area (i.e., the reference area). Therefore, Site-specific soil to earthworm bioaccumulation factors were developed using co-located soil and tissue data and used to estimate

earthworm tissue concentrations throughout the former impoundments at Plainwell, Otsego, and Trowbridge. Based on this approach, an average potential daily dose (APDD) was calculated and compared to no observed adverse effect level (NOAEL) and lowest observed adverse effect level (LOAEL) TRVs derived from a two-generational study in rats (Linder et al., 1974) and adjusted allometrically to account for body weight differences between rats and shrews. The resulting HQs for this analysis were all less than 1 for the low TRV (i.e., TRV-based on an NOAEL), indicating no unacceptable risk to shrews from Total PCBs.

The tissue-based evaluation compared measured tissue body burdens in shrews with literature derived tissue-based TRVs (Linder et al., 1974, Grant et al., 1974, Haddad et al, 2000). The resulting HQs were less than 1 for the high TRV (i.e., TRV based on the), and no low TRV was available. Because the derivation of the tissue-based TRVs required a number of extrapolations and assumptions, and no low TRV was available, the tissue-based evaluation is considered to have more associated uncertainty than the dietary evaluation. Details of both the dietary and tissue-based assessments (including the TRV derivations) are provided in Attachment 1. In addition to the HQ analyses, the results of the shrew trapping studies were evaluated as an ancillary line of evidence to provide information on the abundance of shrews at the Site. These data are also summarized in Attachment 1.

2. Explanation of the process used to select TRVs

The selection of TRVs for consideration in the risk evaluation was a deliberate and extensive process. Sources of toxicological data that were reviewed to develop TRVs included an exhaustive search of the primary peer-reviewed scientific literature to find primary sources of information (Searches using Scopus and Science Direct), pertinent reviews of PCBs, the Oak Ridge National Laboratory (ORNL) report "Toxicological Benchmarks for Wildlife" (Sample et al., 1996), miscellaneous USEPA reports, and other relevant sources of information. The original literature searches were conducted early in the course of the investigation; however, additional searches were conducted periodically to ensure that the TRVs were updated to reflect the most current information, as appropriate. Thousands of papers were identified based on our initial literature searches. The majority were quickly eliminated because they lacked suitable dose-response information or were conducted on test species unrelated to the receptor of concern. Ultimately, approximately 29 studies were evaluated for the purpose of developing avian TRVs. These studies were carefully reviewed with respect to the following criteria:

- <u>Close relatedness of the test species to the wildlife receptor of concern.</u> There is a wide range of species sensitivities to PCBs and other Ah receptor (AhR)-active chemicals (Gasiewicz et al. 1991). Thus, the less related the test species is relative to the receptor of concern, the more uncertainty is associated with the TRV. For example, gallinaceous birds have been observed to be much more sensitive to the reproductive and developmental effects of PCBs than most other avian species (Kennedy et al., 1996; USEPA, 2008); therefore, toxicological data for those birds

are not directly relevant for the specific receptor species we evaluated. Therefore, preference was given to studies evaluating risks to wildlife species more similar to the receptor species.

- Measurement of ecologically relevant endpoint. Subtle biochemical effects may have little or no relevance to the long-term reproductive success of a wildlife species; therefore, only studies evaluating developmental and reproductive effects were considered.

- Presence of co-contaminants. The presence of co-contaminants in the diets of a test species can substantially confound the toxicity results relative to a single chemical or class of chemicals. In particular, assignment of causality, which is key to risk assessment, can be problematic when elevated levels of co-contaminants are present. Therefore, preference was given to studies where the potential impact of co-contaminants was determined to be minimal.

- Chronic exposure and focus on sensitive life stages. Acute studies are of little use when trying to establish NOAELs and LOAELs for chronic effects in wildlife; therefore, only chronic studies that included sensitive life stages to evaluate potential developmental and reproductive effects were considered.

- Multi-year studies. Long-term studies, presenting results for multiple years of exposure, were selected when possible.

- Measure of Total PCBs. Emphasis was given to studies providing an estimate of toxicity associated with Total PCBs rather than to individual congeners or Aroclors.

A summary of the rationale for the selection of each of the relevant TRVs is provided below. The specific values are summarized in the MSU summary document and in the relevant manuscripts.

Passerine Species

The passerine dietary TRVs for Total PCBs selected for this evaluation were derived from a study in which ring-necked pheasants (*Phasianus colchicus*) were dosed with Aroclor 1254, and a threshold for effects was assessed for critical reproductive life stages (Dahlgren et al., 1972). This study was reviewed by our researchers and determined to be of high quality based on the reported methods. In addition, it evaluated critical reproductive life stages and focused on a species that appears to be more sensitive to PCB exposure than wild passerine birds (Thiel et al., 1997; Custer et al., 1998; Henning et al., 2003). This study has also been cited by USEPA (1995; 2000) as an appropriate basis for the derivation of PCB dietary TRVs.

There were very few available studies evaluating the effects of avian TEQs on wildlife species, particularly in passerines. In the absence of such data, a dietary TEQ-based NOAEL and LOAEL were derived based

on a sub-chronic laboratory study evaluating the effects of intra-peritoneal injections of 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) on ring-necked pheasants (Nosek et al., 1992). This study was reviewed by our researchers and determined to be of high quality based on the reported methods. In addition, it was also used as the basis for the ORNL Toxicological Benchmarks for Wildlife (Sample et al., 1996). In the study, it was determined that concentrations of 1000 ng TCDD/kg/week (140 ng TCDD/kg BW/day) resulted in increased mortality, and decreased egg production and hatchability. This exposure concentration was identified as the LOAEL. Because no effects were observed at the two lower dose levels in the study, 100 and 10 ng/kg/week, the 100 ng/kg/week dose (14 ng/kg/day) was selected as the NOAEL.

In addition to a dietary assessment, risk was assessed by comparing concentrations of Total PCBs measured in eggs to tissue-based TRVs based on the NOAEL and the LOAEL for Total PCBs. Although appropriate toxicity data were not available for an exclusively terrestrial avian species, TRVs were available for the tree swallow, a wild passerine species. The egg-based TRVs selected for this assessment were derived from several studies conducted using tree swallows on the Hudson River and Housatonic River. These studies were reviewed by our researchers and determined to be of high quality based on the reported methods. In addition, they are multi-year field studies evaluating the potential effects of PCBs on one of the specific receptor species being considered.

On the Hudson River, potential effects in tree swallows were monitored over the course of two years. Effects such as abnormal plumage, decreased hatching success, and increased abandonment were observed in one year but not the other. The NOAEL value of 26.7 mg Total PCBs/ kg wet weight (ww) represents the lowest site-wide mean concentration of PCBs in eggs from any of the locations evaluated during the year in which no effects were observed. The LOAEL for Total PCBs was derived from a Housatonic River study in which hatching success was impaired during two years of the study (Custer et al., 2003). The least average concentration of Total PCBs in piping chicks from the two years (63 mg Total PCBs/kg) was used as the LOAEL. This study was deemed appropriate because effects were observed in two consecutive years and sensitive reproductive endpoints were evaluated.

The TRV selected as a threshold for effects based on egg TEQ concentrations was also based on the Hudson River study (USEPA, 2000). The greatest TEQ concentration in the year without effects – 13,000 ng TEQ/kg ww – was selected as the NOAEL value. A LOAEL based on TEQs could not be derived from the literature (Neigh et al., 2006a and b).

<u>Great Horned Owl</u>

The dietary NOAEL of 0.4 ug Total PCBs/g/day applied for Great Horned Owls was based on a controlled, laboratory study on the reproductive effects of Total PCBs on the screech owl (McLane and Hughes, 1980). This study was selected because it was determined to be of high quality, focused on a chronic endpoint, and evaluated potential risks to a non-domesticated species similar to the receptor species. A

LOAEL was not identified in this study; however, because the NOAEL was consistent with the TRVs reported for chickens, which are very sensitive to PCBs relative to Great Horned Owls, it was assumed that applying a safety factor of 3 would result in a sufficiently protective LOAEL value of 1.2 ug Total PCBs/g bw/day.

TRVs were also developed to evaluate the potential for adverse effects due to concentrations of Total PCBs in Great Horned Owl eggs. The TRV values selected were based on the feeding study with screech owls (McLane and Hughes, 1980), described above. As previously noted, this study was determined to be of high quality, focused on a chronic endpoint, and evaluated potential risks to a non-domesticated species similar to the receptor species. The NOAEL in this study was 7.0 ug Total PCBs/g bw/day. A LOAEL was not reported in this study. Therefore, a TRV was estimated by multiplying the NOAEL value by an uncertainty factor of 3 as described for the dietary TRV, resulting in a LOAEL TRV of 21 ug Total PCBs/g bw/day.

As noted for the passerine species, there are very few available studies evaluating the effects of avian TEQs on wildlife species such as Great Horned Owls. In the absence of data for a closely related test species the TRVs derived for the passerine species were applied (Nosek et al., 1992). Similarly, no relevant studies on effects of TEQ in the eggs of owl species were available from which to derive a TRV. Therefore, a tissue-based NOAEL for TEQ in Great Horned Owl eggs of greater than 140 picograms TEQ per gram of egg (pg TEQ/g egg) (wet weight) was estimated (Strause et al., 2007a) based on the results of studies evaluating bald eagle chicks (presented on an egg basis [Elliot et al., 1996] and osprey egg exposures [Elliott et al., 2000; Woodford et al., 1998]). These studies were determined to be of high quality based on the methods reported and evaluated potential risks to non-domesticated raptor species. A LOAEL of 400 pg TEQ/g egg (wet weight) was applied, based on CYP1A induction in bald eagles (Strause et al., 2007a).

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ATTACHMENT 1:

KALAMAZOO RIVER AREA OF CONCERN

DRAFT INTERIM ECOLOGICAL RISK ASSESSMENT OF FORMER IMPOUNDMENT SOILS

UPDATE

OCTOBER 05, 2001

KALAMAZOO RIVER AREA OF CONCERN
DRAFT INTERIM ECOLOGICAL RISK ASSESSMENT
OF FORMER IMPOUNDMENT SOILS

UPDATE
OCTOBER 05, 2001

Terrestrial Plant Data, Shrew Data, Preliminary Great Horned Owl Productivity and Dietary Composition, Tree Swallow Nest Box Data, Mink Liver Data, and Preliminary Bald Eagle Dietary Composition

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October, 2001

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Definitions and Acronyms

BAF	Bioaccumulation Factor
BERA	Baseline Ecological Risk Assessment
BSAF	Biota-Sediment (or Soil) Accumulation Factor; relates the concentration of PCBs in sediment or soil to the concentration of PCBs in biota that are in contact with either the sediment or soil.
CDM	Camp Dresser and McKee
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act
DI	Dietary Intake: the rate of intake of the chemical via the diet, generally given in units of mg chemical/kg body weight per day (mg/kg-day) or mg chemical/kg food (mg/kg).
ERA	Ecological Risk Assessment
HQ	Hazard Quotient: the ratio of DI to the toxicity reference value (i.e., the dose of chemical assumed to be without deleterious effect for the receptor of concern, generally given in units of mg/kg-day or mg/kg).
KRAOC	Kalamazoo River Area of Concern
MDEQ	Michigan Department of Environmental Quality
MSU-ATL	Michigan State University Aquatic Toxicology Laboratory
NFSTC	National Food Safety and Toxicology Center
NPL	National Priorities List
PCBs	Polychlorinated Biphenyls
SLERA	Screening Level Ecological Risk Assessment
TRVs	Toxicity Reference Values
UCL	Upper Confidence Interval
USEPA	United States Environmental Protection Agency

1.0 EXECUTIVE SUMMARY

This report has been issued as an updated supplement for the report entitled, "Phase I Kalamazoo River Area Of Concern Draft Interim Ecological Risk Assessment Of Former Impoundment Soils", which was issued in October of 2000 and which will hereafter be referred to as the Draft Interim Giesy Ecotoxicology ERA (2000). It should be noted that this update to the Draft Interim Giesy Ecotoxicology ERA (2000) includes only the first few data sets from a three-year site-specific study and should be considered preliminary until all of the data have been collected, analyzed, and interpreted. Furthermore, as additional data become available, additional updates to the Draft Interim Giesy Ecotoxicology ERA (2000) will be prepared and submitted. By submitting data updates in this way, decision makers will be provided with the most up-to-date and relevant information in a timely manner in order to describe the potential risk of harm to terrestrial receptors in the Kalamazoo River Area of Concern (KRAOC) from formerly impounded, now exposed sediments. Similarly, information necessary to describe the potential risk of harm to ecological receptors exposed through aquatic pathways in the KRAOC will be included in these updates.

The enclosed updated information for the ERA includes analytical chemistry data for terrestrial plants, shrews, and mink liver, and the associated risk calculations and conclusions for receptors affected by this new data, including receptors that consume terrestrial plants growing in soils of the former impoundments. Data in this update includes:

- Great horned owl (GHO) survey results from 2000
- GHO survey and productivity results from 2001
- GHO dietary composition from 2001
- Bald eagle productivity from 2000 and 2001
- Bald eagle dietary composition from 2000 and 2001
- Data for concentrations of PCBs in shrews from the former Trowbridge impoundment.

This report is to be used in conjunction with the Draft Interim Giesy Ecotoxicology ERA (2000) and detailed work plans developed by Michigan State University. It is expected that additional updates will follow as soon as additional data sets have been completed. Some of the information that is nearly complete and that will be included in subsequent updates includes:

- Shrews from Fort Custer (reference) locations
- Additional small mammal samples from the former Trowbridge and Fort Custer (reference) locations
- Great horned owl serum and egg samples
- Bald eagle serum and egg samples
- Miscellaneous food web items and dietary items for receptors

Previously submitted ERA documents for the KRAOC include the Michigan Department of Environmental Quality (MDEQ) / Camp Dresser and McKee (CDM) June 1999 document titled "Final Baseline Ecological Risk Assessment, Allied Paper, Inc. / Portage Creek / Kalamazoo River Superfund Site", and the addendum to this document dated August 15, 2000. Together, these documents are

hereafter referred to as the MDEQ ERA unless otherwise indicated. The overall risk characterization approach to be applied in this update is simply a revision of the MDEQ ERA hazard quotient (HQ) approach. Improvements of the Draft Interim Giesy Ecotoxicology ERA (2000) and subsequent updates over the previously submitted MDEQ ERA documents are presented in a comparative stepwise fashion, are clearly noted for transparency, and include: the addition of data sets consisting of site-specific, state-of-the-art congener-specific PCB analysis performed on terrestrial plants, refined site-specific exposure profiles, and refined effects levels (refer to section 3.0 for more details on the different exposure and risk characterization approaches that are utilized in this ERA for wildlife that consume terrestrial vegetation).

The results of the MDEQ ERA indicated that PCBs in the KRAOC theoretically could constitute a risk of harm to some terrestrial ecological receptors (i.e., the American robin and the white-footed mouse). These predicted risks were due primarily to predicted exposures of terrestrial receptors to PCBs from the consumption of plant tissue. However, when measured site-specific concentrations of PCBs in plants were substituted for literature-derived estimates in this update to the Draft Interim Giesy Ecotoxicology ERA (2000), the resulting HQs for each of these terrestrial receptors were substantially reduced to values that were equal to or less than 1.0. Separate from the MDEQ ERA methodology and assumptions, an alternative ERA was conducted in which multiple exposure scenarios and scientifically defensible toxicity reference values were utilized. While the degree of conservatism remained great, an attempt was made to make the exposure and toxicity assumptions more realistic. The resultant recalculated HQs in this alternative ERA were less than 1.0 for all of the terrestrial receptors considered in the MDEQ ERA.

In addition to the terrestrial receptors selected by MDEQ, the MSU-ATL collected data that can be used to assess potential risks to the short-tailed shrew, a receptor that would be expected to have maximal exposure to PCBs in the soils. The lines of evidence that were evaluated for the short-tailed shrew include: 1) presence and numbers of shrews in the study area compared to a reference location; 2) comparison of field-collected, whole body burdens of PCBs in shrews to literature-derived tissue residue-based toxicity reference values; and 3) comparison of estimated and/or measured site-specific dietary exposures for shrews to literature-derived toxicity reference values. Hazard quotients (HQs) were calculated for short-tailed shrews based on dietary exposures and based on tissue residues. The hazard quotients range from 0.06 to 0.84 depending on the approach and the assumptions that were utilized. The degree of uncertainty is greater for the dietary exposure model compared to the tissue residue model because of uncertainty in site use assumptions and because data are not yet available for many of the site-specific dietary items of shrews such as crickets, beetles, and snails. As this information becomes available, it will be used to refine the exposure calculations. An additional consideration is a comparison of field measured concentrations of PCBs in shrews collected from the former Trowbridge impoundment to concentrations of PCBs in shrews exposed in laboratory studies. The concentrations of PCBs in shrews from the field correspond most closely to those from a laboratory exposure of shrews to 0.55 mg PCBs/kg in the diet, a concentration that is well below comparative concentrations necessary to elicit adverse effects in other rodents, which are approximately 5-20 mg PCBs/kg in the diet. Thus, adverse effects in shrews would not be expected. Taking into consideration the conservative nature of the TRVs and the relatively small HQ values from multiple methods of evaluation, the risk posed to shrews by concentrations of PCBs in the soils of the former impoundments is minimal. This, coupled with observations of viable populations of shrews and other small mammal species collected from the former Trowbridge impoundment, indicates that there are no measurable, population-level effects. Thus, no effects would be predicted and none seem to be observed.

Great horned owls were evaluated in the MDEQ ERA and were found to have dietary-based hazard quotients less than 1.0 when compared to either the LOAEL or NOAEL. However, in the addendum to the MDEQ ERA, risks to owls were reconsidered based on concentrations of PCBs in three great horned owl eggs collected from the area downstream of Lake Allegan in the early 1990's. MDEQ utilized this data in a top-down analysis to provide the basis of a soil cleanup value. However, the paper from which their toxicity reference value (TRV) was derived does not provide support for the TRV value for great

horned owls in the MDEQ ERA (for further discussion, refer to the Draft Interim Giesy Ecotoxicology ERA (2000)). Furthermore, based on the locations from where the owl eggs were collected, it is unlikely that the flood plain soils were a major source of PCBs that were accumulated into the owl eggs. Until a link to the flood plain soils can be established, the top-down analysis based on great horned owls is inappropriate to determine a target soil threshold for protection of sensitive higher trophic level receptors.

Predicted risks to great horned owls from dietary exposure to PCBs in the KRAOC were previously found to be insignificant as discussed in the Draft Interim Giesy Ecotoxicology ERA (2000). To provide additional lines of evidence for consideration in the baseline ERA and to further address the potential risks of PCBs to great horned owls, field studies are being conducted to address site-specific productivity, dietary composition, and dietary exposure to PCBs. The data on dietary composition will be used in determining the site-specific exposure concentrations for great horned owls residing in the KRAOC. However, data on PCB concentrations for most of the individual dietary items are not currently available. It is important to note that the dietary pathways of great horned owls are not strictly based on terrestrial prey as demonstrated by the presence of muskrat remains. This evidence is consistent with several reports in the literature of great horned owls consuming biota linked to aquatic ecosystems such as fish, waterfowl, muskrats, and great blue herons. Thus, it may be inappropriate to determine a target soil threshold for protection of sensitive higher trophic level receptors based on great horned owls since their exposures are linked to both aquatic and terrestrial sources. The available dietary-based and tissue residue-based approaches, although preliminary, do not support the conclusion that PCBs are adversely affecting great horned owl populations in the KRAOC.

The overall conclusions of this update to the Draft Interim Giesy Ecotoxicology ERA (2000) indicate that there is no evidence of risk of harm to the terrestrial receptors considered in the MDEQ ERA.

In light of the need to provide decision-makers with the most up-to-date information available, additional data collected from MSU-ATL field studies are provided here that relate to aquatic exposure pathways. While many studies are preliminary (i.e., the first year of a three-year study), it may be helpful for decision-makers to understand the status and preliminary results of on-going studies. Currently available data include information on tree swallow nest box studies, PCB concentrations in mink liver, and bald eagle observational data. Although the data are not yet available for predicted risks from dietary exposure to PCBs or TEQs for mink, the available tissue residue data indicate that tissue residue-based hazard quotients for mink range from approximately 0.33 to 1.71 for mink captured from areas proximal to the former impoundments and approximately 0.21 to 0.64 for mink captured from reference locations. Preliminary bald eagle productivity data from the KRAOC includes one fledged chick and one addled egg from Ottawa Marsh in 2000, and one fledgling from the Swan Creek Highbanks nest in 2001. Data for bald eagle productivity from reference locations will be presented in a subsequent update.

2.0 INTRODUCTION

This report has been prepared to provide decision-makers with all available and relevant information in order to describe the potential risk of harm to terrestrial ecological receptors in the Kalamazoo River Area of Concern (KRAOC). This report has been issued as an updated supplement for the report entitled, "Phase I Kalamazoo River Area Of Concern Draft Interim Ecological Risk Assessment Of Former Impoundment Soils", which was issued in October of 2000 and which will be hereafter referred to as the Draft Interim Giesy Ecotoxicology ERA (2000). It should be noted that this update to the Draft Interim Giesy Ecotoxicology ERA (2000) includes only the first few data sets from a three-year site-specific study and should be considered preliminary until all of the data have been collected, analyzed, and interpreted. Furthermore, as additional data become available, additional updates to the Draft Interim Giesy Ecotoxicology ERA (2000) will be submitted. By submitting data updates in this way, decision makers will be provided with the most up-to-date and relevant information in a timely manner in order to describe the potential risk of harm to terrestrial receptors in the Kalamazoo River Area of Concern (KRAOC) from formerly impounded, now exposed sediments. Similarly, information necessary to describe the potential risk of harm to ecological receptors exposed through aquatic pathways in the KRAOC will be included in these updates.

A series of site-specific studies are being conducted within the KRAOC by the Michigan State University, Aquatic Toxicology Laboratory (MSU-ATL), the Department of Zoology, and National Food Safety and Toxicology Center (NFSTC). As mandated by the CERCLA process, a risk analysis of possible adverse effects on resident wildlife should be performed as part of the KRAOC Remedial Investigation and Feasibility Study (RI/FS). The aforementioned MSU-ATL studies were designed specifically to fill important data gaps vital to an accurate, site-specific risk analysis as mandated by CERCLA. The data collected from the MSU-ATL studies will be combined with previously obtained site-specific data utilized in the Michigan Department of Environmental Quality (MDEQ) / Camp Dresser and McKee (CDM) June 1999 document titled "Final Baseline Ecological Risk Assessment, Allied Paper, Inc. / Portage Creek / Kalamazoo River Superfund Site", and the Addendum to this document dated August 15, 2000 (MDEQ 1999, and 2000). Together, these documents are hereafter referred to as the MDEQ ERA. The overall risk characterization approach to be applied in this update and forthcoming updates is simply a revision of the MDEQ ERA hazard quotient (HQ) approach. Refined approaches in the Draft Interim Giesy Ecotoxicology ERA (2000) and associated updates over the previously submitted MDEQ ERA documents are included in a comparative stepwise fashion and are clearly noted for transparency. Refinements include site and congener-specific PCB analysis performed on floodplain soils and terrestrial plants, refined site-specific exposure profiles, and refined effect levels.

This update to the Draft Interim Giesy Ecotoxicology ERA (2000) only includes the first few data sets that are available from a three-year site-specific study presently being performed by the MSU-ATL. The primary focus of this update is on terrestrial exposure pathways. However, additional data collected from MSU-ATL studies are provided here that relate to aquatic exposure pathways. This ERA will be periodically updated, as data become available, to include a comprehensive multiple lines of evidence approach for key wildlife receptors.

2.1 Schedule and Purpose of Additional Data Sets

Our general strategy for this and subsequent ERA updates is to utilize a multiple lines of evidence approach incorporating all of the data available. The expected completion date for each data set and a description of how it will be used in the ERA is included (Table 2-1). The overall purpose of all of the data collected as part of these studies is to elucidate site-specific and PCB congener-specific exposure and

population health for those ecological receptors previously identified as being of greatest priority (refer to MSU-ATL work plan). In general, the data sets fall into one of two categories: (1) exposure studies or (2) effects studies.

Exposure studies have been designed to gather site-specific data on the concentrations of individual PCB congeners. This will better define PCB exposures, thereby minimizing the need for conservative exposure and effects assumptions and thereby reducing uncertainty in the baseline ERA. Exposure studies include site-specific dietary composition of key receptors, determination of PCB concentrations in site-specific prey items, tissue residue levels in key receptors, and data necessary to develop a food web model that can be used to address movement of PCBs within the KRAOC food web.

Effect studies have been designed to gather site-specific population data for key receptors based on historical data and receptor food chain position. The presence or absence of viable populations is an important line of evidence for understanding receptor population health. Populations of key receptors are being examined in terms of both physical (e.g., habitat suitability) and/or chemical (e.g., PCB congeners, DDT and metabolites) stressors. Tissue residue concentrations in key receptors (mink livers, bald eagle eggs, etc.) will be compared to tissue-based toxicity reference values developed from published studies in the peer-reviewed scientific literature. Each of the effect-based studies is based on population-level measurement endpoints and compared to a reference value or baseline area.

2.2 Report Organization

Organization of this and each subsequent update will be by receptor type rather than by components of the risk assessment process (e.g., problem formulation, exposure, effects, and risk assessment). The remainder of this report is organized as follows:

Section 3.0. Wildlife That Consume Terrestrial Plants

Section 4.0. Short-tailed Shrew

Section 5.0. Great Horned Owl

Section 6.0. Tree Swallow

Section 7.0. Mink

Section 8.0. Bald Eagle

Section 9.0. Aquatic Plants

Section 10.0 References

Appendix A. Dietary Exposure Risk Calculations and Related Documentation

Appendix B. Data sets on impoundment soils, earthworms, white-footed mice, waterfowl, and muskrats with descriptive statistics that are used in the interim ERA

Appendix C. Exposure profiles and risk characterization for the American robin, short-tailed shrew, and great horned owls based on interim ERA approaches.

Table 2-1. Status of Data Availability for Samples Collected from the Kalamazoo River Area of Concern by Michigan State University's Aquatic Toxicology Laboratory (MSU-ATL).

Sample Type	Total # of Samples	Status	Data Applicability
Terrestrial and Aquatic Plant Samples	37	Completed	Dietary Item Food Web Model
Trowbridge Shrew Samples	16	Completed; Undergoing Final QA	Receptor Dietary Item Food Web Model
Mink Livers	9	Completed; Undergoing Final QA	Receptor Compare to Mink Liver TRV Compare Study and Reference Sites
Owl Blood Sera	5	Undergoing QA	Receptor Develop Serum to Egg Relationship Compare to Egg TRV
Owl Eggs	3	Undergoing QA	Receptor Compare to Egg TRV Compare Study and Reference Sites
Eagle Eggs	1	Undergoing QA	Receptor Compare to Egg TRV Compare Study and Reference Sites
Eagle Blood Sera	12	Undergoing QA	Receptor Develop Serum to Egg Relationship Compare to Egg TRV
Fort Custer Small Mammal	38	Undergoing QA	Dietary Item Food Web Model
Great Horned Owl Pellet Analysis (non-chemical)	22	50% complete	Dietary Composition Food Web Model
Bald Eagle Prey Remains and Observation Data (non-chemical)	96	50% complete	Dietary Composition Food Web Model
Other Trowbridge Small Mammal Samples	20 random out of 89	Analysis in progress	Dietary Item Food Web Model
Soil Samples	36	Extracted	Food Web Model
Sediment Samples	43	Extracted	Food Web Model
Soil Worms: depurated and non-depurated	17	Extracted	Dietary Item Food Web Model
Crayfish	12	Extracted	Dietary Item Food Web Model
Aquatic Emergent Invertebrates	Subset of 132	Extraction in progress	Dietary Item Food Web Model
Soil Invertebrates	Subset of 96	Dec. 2001	Dietary Item Food Web Model
Benthic Invertebrates	32	Dec. 2001	Dietary Item Food Web Model
Fish	144	Subset under-going QA	Dietary Item Food Web Model
Turtles	8	Dec. 2001	Dietary Item Food Web Model

3.0 WILDLIFE THAT CONSUME TERRESTRIAL PLANTS

3.1 Concentration of PCBs in Plants

Concentrations of PCBs in terrestrial plants have been determined for multiple species at several locations in different seasons (Table 3-1) over a period of two years (Giesy Ecotoxicology, 2000). Plant sampling events can be separated into two categories. Those that were sampled throughout the year within the six biological sampling areas or BSAs (4 Trowbridge, 2 Fort Custer) and those that were sampled within the Trowbridge impoundment based on targeted species, irrespective of BSAs. BSA targeted sampling was based on location and occurred within the appropriate 35 x 35 meter small mammal sampling grid. A minimum of 10 g plant material was collected from the above ground portion of the plant (just above the root crown). Plant samples were not rinsed prior to storage, transport or analysis. The sampling area was a 1 m^2 randomly selected plot and included the simultaneous collection of co-located samples of soil, above ground terrestrial invertebrates, subterranean invertebrates and small mammals. In the rare instances where the 1 m^2 sampling grid contained more than a single species of plant with greater than 10 g biomass a second species was also sampled. Year 2000 BSA sampling began in late May and occurred three times at approximately 45 day intervals.

Some samples were collected within the Trowbridge impoundment but outside of BSAs based on criteria and targeted species developed in conjunction with US Fish and Wildlife Service. These criteria and targeted species include plants, seeds and/or fruits consumed by songbirds, terrestrial plants either submerged or partially submerged in the river basin for the majority of their growing season, and those plants species previously identified as PCB bioaccumulators based on a list generated by Dave Charters USEPA, Sullivan Ledge Superfund site (see footnote for Table 3-1). The concentrations of PCBs in bioaccumulating plants from the former Trowbridge impoundment were 0.0136 and 0.0183 for the mean and the 95% UCL of the mean, respectively. Co-located soils were also sampled simultaneously with most targeted species plant samples. Based on predetermined sample analysis priorities, only a select number of co-located soil samples were analyzed in conjunction with the plant samples included in this update. The remaining soil samples will be analyzed in conjunction with subterranean invertebrates including depurated and non-depurated worm samples and will be included in subsequent updates.

Concentrations of PCBs in plant tissues from all locations range from 0.0012 to 0.200 mg/kg, (wet weight). Plant tissues collected as part of the songbird diet contained the lowest PCB concentrations ranging from 0.0017 to 0.0073 mg/kg, (wet weight) with an arithmetic mean and 95% UCL of the arithmetic mean of 0.0020 and 0.0053 mg/kg, (wet weight), respectively. Plants tissues collected as potential PCB bioaccumulators were less than the arithmetic mean of all terrestrial plant samples collected from Trowbridge and ranged from 0.0029 to 0.00525 mg/kg, (wet weight) with an arithmetic mean and 95% UCL of the arithmetic mean of 0.0136 and 0.0183 mg/kg, (wet weight), respectively. For all of the terrestrial plants collected from Trowbridge, the concentrations of PCBs ranged from 0.0017 to 0.200 mg/kg (wet weight) with an arithmetic mean and 95% UCL of the arithmetic mean of 0.0174 and 0.0327 mg/kg (wet weight), respectively. This sample group includes both the single greatest value of 0.200 mg/kg from a sample of nettle (not a likely predominant food source for any receptor), as well as the May 2000 plant tissue samples. The May 2000 plant tissue group contained PCB concentrations which were 4.2 times greater than the next highest plant tissue group (July 2000 sampling event) and more than 200 times greater than the songbird diet sampling. This was likely due to sediments on the surface of the plant tissues as a result of spring flooding which receded only days before sampling (as noted by sampling personnel). Nevertheless to be conservative in nature, these data were included in the mean and 95% UCL values used to calculate average potential daily doses (refer to section 3.2). Plant tissue data are available in an electronic spreadsheet and a complete data package, including QA information and chromatograms which are available upon request.

Table 3-1. Concentrations of total PCBs in terrestrial plants collected from the KRAOC.

Sample Description	Genus/species or common name	Lab ID	Total PCB Concentration (mg/kg, ww)* Individual sample	Mean	95% UCL
Fall Songbird Diet at Trowbridge (October 2000)				**0.0037**	**0.0053**
	Unkn. Seed Pod	TBPS005	0.0020		
	Ragweed	TBPS007	0.0042		
	Unkn. Seed Pod	TBPS008	0.0032		
	Rosehips	TBPS010	0.0039		
	Milkweed Pod	TBPS011	0.0017		
	Wild Grapes	TBPS017	0.0073		
Trowbridge (May 2000)				**0.0580**	**0.151**
	Grass**	TBPS018	0.0029		
	Quackgrass**	TBPS019	0.0071		
	Nettle	TBPS020	0.200		
	Grass**	TBPS021	0.0219		
Trowbridge (July 2000)				**0.0127**	**0.0229**
	Grass**	TBPS022	0.0345		
	Grass**	TBPS023	0.0078		
	Quackgrass**	TBPS024	0.0214		
	Quackgrass**	TBPS025	0.0072		
	Wild Grapes	TBPS026	0.0022		
	Co-Loc. Plant	TBPS033	0.0029		
Trowbridge (August 2000)				**0.0139**	**0.0196**
	Quackgrass**	TBPS027	0.0210		
	Quackgrass**	TBPS028	0.0179		
	Grass**	TBPS029	0.0080		
	Nettle	TBPS030	0.0024		
	Quackgrass**	TBPS031	0.0181		
	Rosehips	TBPS032	0.0157		
Trowbridge (October, 2000)				**0.0074**	**0.0095**
	Aspen**	TBPS009	0.0085		
	Sumac**	TBPS012	0.0086		
	Purple Loosetrife**	TBPS002	0.0052		
Fort Custer (July 2000)				**0.0036**	**0.0055**
	Grass**	FCPS001	0.0058		
	Grass**	FCPS002	0.0024		
	Nettle	FCPS003	0.0016		
	Quackgrass**	FCPS004	0.0047		
Fort Custer (August 2000)				**0.0041**	**0.0083**
	Grass**	FCPS005	0.0103		
	Yellow Rocket	FCPS006	0.0012		
	Co-Loc. Plant #1	FCPS008	0.0037		
	Co-Loc. Plant #2	FCPS009	0.0012		
	All Trowbridge Terrestrial Plants			**0.0174**	**0.0327**
	All Fort Custer Terrestrial Plants			**0.0038**	**0.0060**

*Concentrations of non-detectable PCB congeners were estimated by assuming one-half detection limit.

**Bioaccumulating plant species (see text for details).

3.2 Exposure Calculations for Wildlife That Consume Terrestrial Plants

The American robin and the white-footed mouse are the two previously identified receptors of concern (MDEQ ERA 1999) for which plant tissues comprise a significant portion of the diet. In the MDEQ ERA (MDEQ 1999), concentrations of PCBs were not available for terrestrial vegetation and, therefore, were modeled using a conservative bioaccumulation factor (BAF). The unnecessary uncertainty resulting in this conservative approach was recognized in the subsequent addendum ERA (Addendum MDEQ 2000) in which an attempt was made to develop a central tendency of published literature values. However, the studies cited were largely inappropriate for the evaluation of the uptake of PCBs from KRAOC soils into plants (for an in depth discussion of this issue, see Giesy Ecotoxicology, 2000). The modeled concentrations are considerably greater than the actual measured concentrations of PCBs in site-specific plant tissues (Table 3-2). Using the conservative 95% UCL concentration for PCBs measured in terrestrial plants, the average potential daily doses (ADD_{pot}) were only 4 and 8% of those predicted by MDEQ for American robin and white-footed mouse respectively based on the 1999 Final ERA (MDEQ 1999) or 16 and 29% based on the ERA Addendum (MDEQ 2000) (Table 3-3).

Previously, site-specific data on measured concentrations of PCBs in plants were available for only six plant samples (Giesy Ecotoxicology, 2000). In this update, a new data set consisting of five separate sampling events spread out over the entire growing season was utilized. To provide for worst case scenarios, one sampling event targeted previously identified PCB bioaccumulating plants, while the entire data set included plant tissue in which soil and sediment surface contamination was not removed. Despite its intrinsic conservatism, this new measured plant tissue PCB data set resulted in virtually identical average potential daily doses for both the American robin and white-footed mouse as the previously measured PCB plant tissue data (Table 3-3). An examination of the differences between the MDEQ calculations based on modeled plant PCB data and the measured PCB data sets clearly demonstrates the reduction in uncertainty when site-specific data are utilized.

Exposure scenarios and calculations are summarized for comparison in the remainder of this section for the following assumptions and data sets:

1) Calculations based on the exact same modeled data, exposure parameters, and toxicity reference values (TRVs) as MDEQ (MDEQ 1999 and 2000);

2) ERA calculations based on measured, site-specific concentrations of PCBs in plants replacing modeled estimates while all other calculations are based on the same exposure parameters and TRVs as MDEQ (MDEQ 1999 and 2000); and

3) Interim ERA calculations based on measured, site-specific concentrations of PCBs in plants and more appropriate exposure parameters (i.e., including surface water ingestion and incidental ingestion of soil) and TRVs.

Table 3-2. Concentration of total PCBs predicted and measured in KRAOC terrestrial vegetation.

	MDEQ ERA Modeled Data		2000 Measured data[A]	2001 Measured data[A]
	1999 version Total PCBs (mg/kg, ww)	**2000 version Total PCBs (mg/kg, ww)**	**Total PCBs (mg/kg, ww)**	**Total PCBs (mg/kg, ww)**
Mean	NC	NC	0.0159	0.0174
95% UCL (Mean)	22	4.7	0.0239	0.0327
Geometric mean	NC	NC	0.0135	0.0079
95% UCL (Geometric mean)	NC	NC	0.0223	0.0121
Range	0.3 – 39.3	0.06-8.5	0.0074 - 0.0312	0.0020 - 0.2000
n	NC	NC	6	25

[A]The samples collected and analyzed by the MSU-ATL were all collected at the former Trowbridge impoundment. Thirty-one samples of terrestrial vegetation have been analyzed to date. Concentrations of non-detectable congeners were estimated by assuming one-half the detection limit. Data are presented in Table 3-1 and Giesy Ecotoxicology, (2000).

NC – not calculated.

Table 3-3. Average potential daily doses (ADD_{pot}) of total PCBs for the American robin and white-footed mouse.

	Average Potential Daily Dose of Total PCBs (mg/kg/d)			
	MDEQ ERA Modeled Data 1999 version	**MDEQ ERA Modeled Data 2000 version**	**Measured data (Oct 2000)[A] (MSU-ATL) Trowbridge[C]**	**Measured data (July 2001)[B] (MSU-ATL) Trowbridge[C]**
American Robin	13.8	3.67	0.60	0.60
White-Footed Mouse	2.8	0.76	0.22	0.25

[A]Calculated using data from samples collected and analyzed in 2000 (refer to Table 3-2 for details).

[B]Calculated using data from samples collected in 2000 and analyzed in 2001 (refer to Table 3-2 for details).

[C]The measured exposure point concentration used in the calculation for each species was the 95% UCL of concentration of PCBs in terrestrial vegetation (using Trowbridge data). For the purposes of comparison, all other assumptions made in the exposure calculations are the same as found in the MDEQ ERA (MDEQ 1999) with the exception that the ingestion rates for robins were assumed to be 1.2 g/g/d for plants and 0.76 g/g/d for invertebrates (US EPA 1993). Refer to Appendix A for more details.

The availability of site-specific measured plant tissue PCB concentrations not only allows for more accurate assessment of exposure but also the opportunity to further refine exposure scenarios both spatially and temporally. In this ERA update, as in the Draft Interim Giesy Ecotoxicology ERA (2000), a more refined site wide hazard assessment is presented. In this more complete and defensible risk characterization, multiple exposure scenarios are presented in order to provide a comprehensive representation of current site conditions to risk managers. In this way risk managers can determine how sensitive the results are to changes in the underlying assumptions.

Characteristics of the American robin and white-footed mouse are presented in Table 3-4 and Table 3-5**,** including exposure assumptions for body weight, ingestion rates, dietary composition, etc. While exposure pathways such as consumption of water and incidental ingestion of soil were considered negligible and excluded from the MDEQ ERA, they were included in this more comprehensive ERA. Exposure estimates for both the American robin and the white-footed mouse were calculated based on site-specific measured PCBs values for plant tissues, soil and water. To minimize uncertainty, instead of using models with multiple assumptions to predict movement of PCBs from soils to receptors, measured concentrations of dietary PCBs were utilized in the Draft Interim Giesy Ecotoxicology ERA (2000) and its updates. Since data are available for typical dietary items for both the American robin and the white-footed mouse, there was no need to model exposure to the prey items at Trowbridge and at the reference site. Exposures at other locations in which certain data are lacking were assumed to be similar to that of Trowbridge as a worst-case scenario.

Table 3-4. Key characteristics of terrestrial receptors.

Receptor	Genus/Species	Body Weight BW (kg)	Food Ingestion Rate IR_{feed} (kg/day) [a]	Water Ingestion Rate IR_{water} (L/day)	Soil Ingestion Rate IR_{soil} (kg/day) [b]
American robin	*Turdus migratorius*	0.077[1]	0.092 (0.0585) [c]	0.011[1]	0.0005[2]
White-footed mouse	*Peromyscus leucopus*	0.021[1]	0.0055[1]	0.004[1]	0.00003[3]

[a]kg/day wet weight
[b]kg/day dry weight
[c]The ingestion rate for robins varies with the food type (IR_{plants}= 0.092 kg/day; $IR_{terr.\ inverts.}$ = 0.0585 kg/day). Ingestion rate of invertebrates in the diet of robins is calculated from the plant rate modified for differences in both assimilation efficiency and energy content of food (Appendix A; USEPA 1993).
[1]USEPA, 1993. Wildlife exposure factors handbook. EPA/600/R-93/187.
[2]Calculated based on percent soil in diet (dry weight basis) as given by (Beyer 1994), assuming 70% water content in food (USEPA 1993).
[3]Calculated based on percent soil in diet determined by best scientific judgement, assuming 70% water content in food.

Table 3-5. Dietary fraction for ecological receptors.

Receptor	Dietary Fraction (% of total diet on a wet weight basis)[1] Plants	Terrestrial invertebrates
American robin	49	51
White-footed mouse	44	56

[1]Dietary fraction is based on (USEPA 1993), and scientific judgement.

Estimates of daily contaminant exposure experienced by American robins and white-footed mice were calculated using a modification of a generalized exposure model (USEPA 1998). The resulting exposure model is depicted (Eq. 3-1):

$$ADDpot(mg/kgBW/d) = \sum_{k}^{m} (C_k \times FR_k \times NIR_k) \qquad \textbf{Eq. 3-1}$$

Where:

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

BW = Body weight

C_k = Average contaminant concentration in the kth type of food (*e.g.*, in mg/kg wet weight)

FR_k = Fraction of intake of the kth food type that is from the contaminated area (unitless)

NIR_k = normalized ingestion rate of the kth food type on a wet weight basis (*e.g.*, in kg food/kg body weight-day)

m = number of contaminated food types

The ADD_{pot} utilized in this interim ERA for the American robin and white-footed mouse were calculated in the same manner as those of the MDEQ ERA and are presented in Table 3-4. Note again that the calculated ADD_{pot} based on actual site-specific data for these species are considerably less than the modeled estimates for concentrations of PCBs in plants presented in previous MDEQ ERAs. Point estimates of exposure will be generated for each species-subreach combination using point estimates of values for species-specific life history parameters and the 95% upper confidence limit (UCL) of the mean contaminant concentration in prey.

An area use factor of 50% was included in the exposure model for this interim ERA for the American robin. American robins would not likely forage at the site more than 50% of the time because of migratory behavior during the winter season. Also, the former impoundment sediments do not provide much suitable foraging habitat for American robins (refer to Appendix D of the Draft Interim Giesy Ecotoxicology ERA, (2000); also personal communication, Dr. Lisa Williams to Dr. John Giesy, June 2000). It is also evident, based on site-specific field data, that the white-footed mouse would not likely forage at the site for a substantial portion of the spring and fall due to substantial water inundation. However, an area use factor was not included in the white-footed mouse exposure scenarios.

3.3 Toxicity Reference Values for American Robin

The TRVs for American robins are based on a reproduction study in which penned pheasants were exposed by gelatin capsule to three doses of Aroclor 1254 (0, 12.5, and 50 mg/week) once per week for 112 days through a critical reproductive lifestage (Dahlgren et al., 1972). Conversion of concentrations in diet to a daily dose are based on a body weight of 1 kg for pheasants (USEPA, 1995). No adverse effects were observed on egg fertility or chick growth at either dose level. However, at 12.5 mg PCBs/week (or 1.8 mg PCBs/kg/day), adverse effects were observed for egg hatchability. Since the study considered dietary exposure during reproduction, the 1.8 mg PCBs/kg/d dose was considered to be a chronic LOAEL. A chronic NOAEL of 0.6 mg PCBs/kg/d was estimated by dividing the chronic LOAEL by an uncertainty factor of three (a factor of ten would be excessively conservative because for the chicken, which is the most sensitive bird species tested, the NOAEL is 0.98 mg/kg/d for the same endpoint).

Furthermore, since the estimated NOAEL is less than the most sensitive bird species tested, no additional uncertainty factors were applied to account for potential intertaxon variability. In other words, it is unlikely that the chronic NOAEL for other bird species is any lower than for chickens. Despite this and to maintain conservatism, the calculated NOAEL used in these calculations is lower than the chicken NOAEL.

3.4 Toxicity Reference Values for White-footed Mouse

The TRVs for the white-footed mouse are based on a reproduction study in which white-footed mice were exposed by diet to 10 mg Aroclor 1254 /kg diet (in feed) for up to 540 days through a critical reproductive lifestage (Linzey, 1987 and 1988). Conversion of concentrations in diet to a daily dose was based on a normalized ingestion rate of 0.135 g food/g body weight/d that was provided in the study. Adverse effects were observed at a dose level of 10 mg PCBs/kg in feed (or 1.35 mg PCBs/kg-day) including a reduction in numbers of young at weaning. Since the study considered dietary exposure during reproduction, the 1.35 mg/kg/d dose was considered to be a chronic LOAEL. A chronic NOAEL of 0.45 mg PCBs/kg/d was estimated by dividing the chronic LOAEL by an uncertainty factor of three. Since the study was conducted on the same species as used in the ERA, there was no need to apply an uncertainty factor to account for potential intertaxon variability.

3.5 Comparison of Toxicity Reference Values for the MDEQ ERA and the Interim ERA

TRVs in this interim ERA have been modified from those in the MDEQ ERA. Modifications were incorporated for those instances in which TRVs with higher associated confidence were located. In short, TRVs associated with the No Observable Adverse Effects Levels remained about the same for the two ERAs (Table 3-6), while TRVs associated with the Lowest Observable Adverse Effect Levels were significantly lower in this interim ERA as compared to those in the MDEQ ERA (Table 3-7).

Table 3-6. Comparison of NOAEL TRVs between the MDEQ ERA and the interim ERA.

	NOAEL Toxicity Benchmarks (mg PCBs/kg/d)	
Receptor	**MDEQ ERA**	**Interim ERA**
American Robin	0.6	0.6
White-Footed Mouse	0.5	0.45

Table 3-7. Comparison of LOAEL TRVs between the MDEQ ERA and the interim ERA.

	LOAEL Toxicity Benchmarks (mg PCBs/kg/d)	
Receptor	**MDEQ ERA**	**Interim ERA**
American Robin	6.0	1.8
White-Footed Mouse	6.5 – 26.1	1.35

3.6 Risk Calculations for Wildlife That Consume Terrestrial Plants

A risk characterization was performed based on measures of exposure (section 3.2) compared to measures of effects (sections 3.3 and 3.4). Hazard quotient (HQ) values were calculated for both the American robin and the white-footed mouse under different exposure scenarios and PCB concentrations in prey by the use of equations 3-2 and 3-3:

$$HQ = \frac{ADD_{pot}\ (\text{mg/kg BW/d})}{\text{Toxicity Reference Value (mg/kg BW/d)}} \qquad \textbf{Eq. 3-2}$$

When tissue residue data are available for receptors, the HQs will be calculated by the following equation:

$$HQ = \frac{\text{Tissue concentration (mg/kg)}}{\text{Toxicity Reference Value (mg/kg)}} \qquad \textbf{Eq. 3-3}$$

The following sections present two risk characterizations with progressive revisions of the MDEQ ERA approach. The first risk characterization, presented in Table 3-8, simply replaces modeled concentrations of PCBs in terrestrial vegetation with measured concentrations, without changing any of the assumptions or TRVs of the MDEQ-ERA. The second, more complete, risk characterization (presented in section 3.6.2) is an interim ERA in which multiple exposure scenarios and scientifically defensible toxicity reference values were utilized.

3.6.1 Hazard Quotients Based on MDEQ Approach

Table 3-8. Predicted risk of adverse effects of PCBs on receptors that consume terrestrial vegetation.

	NOAEL based hazard quotient				**LOAEL based hazard quotient**			
	MDEQ ERA (Modeled data)		**Updated ERA[A] (Measured data)**		**MDEQ ERA (Modeled data)**		**Updated ERA[A] (Measured data)**	
	1999[B]	2000[C]	2000[D]	2001	1999[B]	2000[C]	2000[D]	2001
American Robin	23	6.1	0.99	1.00	2.3	0.61	0.10	0.10
White-Footed Mouse	5.6	1.5	0.43	0.45	0.43	0.12	0.03	0.03

[A]The calculations and assumptions are those of the MDEQ ERA for exposure and effect except that actual values of PCB concentrations have been used instead of predicted values. Values are based on 95% UCL of arithmetic mean of the measured (MSU-ATL) concentrations of total PCBs in terrestrial vegetation (using Trowbridge data) with the exception that the ingestion rates for robins were assumed to be 1.2 g/g/d for plants and 0.76 g/g/d for invertebrates (US EPA 1993). Refer to Appendix A for more details.
[B]MDEQ, 1999
[C]MDEQ, 2000
[D]Giesy Ecotoxicology, 2000

3.6.2 Results of Interim ERA

The results of an ERA are often evaluated with alternative assumptions, data, and/or toxicity data. The purpose of such an evaluation is to determine how sensitive the results are to changes in the underlying assumptions of the ERA (refer to section 3.2 for more details on the various exposure assumptions and data sets used in this ERA). An interim ERA was conducted in which multiple exposure scenarios, including the incidental ingestion of site soil and water, and more stringent toxicity reference values were used. While the degree of conservatism remained great, an attempt was made to make the exposure and toxicity assumptions more realistic including the addition of a site use factor for the American robin based on migratory behavior and habitat suitability. The sitewide HQs from this interim ERA are summarized in Table 3-9 (for more details and calculations for this interim ERA, refer to Appendices A, B, and C).

All of the HQ values (both NOAEL-based and LOAEL-based) were less than 1.0 for both the American robin and the white-footed mouse. Based on the conservative parameterization of multiple assessment methods and scenarios the overall conclusion of this section of the interim ERA indicates that there is no evidence of risk of harm either the American robin or the white-footed mouse within the Kalamazoo River Area of Concern.

Table 3-9. Sitewide hazard quotients for receptors evaluated in the interim ERA.

Receptor	NOAEL-Based Hazard Quotient (maximum for any one site)*	LOAEL-Based Hazard Quotient (maximum for any one site)*
American Robin	0.2 (0.4)	< 0.1 (0.1)
White-footed Mouse	< 0.1 (< 0.1)	< 0.1 (< 0.1)

*Based on the most likely exposure scenario as noted for American robins in Appendix C (also Appendix G of Giesy Ecotoxicology, 2000 for similar calculations for the white-footed mouse).

4.0 SHORT-TAILED SHREW

4.1 Introduction

The short-tailed shrew (*Blarina brevicauda*) lives on and in the soil and eats soil invertebrates, including insect larvae of the orders Diptera, Lepidoptera, Orthoptera, and Hymenoptera and adults of the order Coleoptera. It also eats beechnuts (Hamilton 1930, 1941), earthworms (Babcock 1914; Whitaker and Ferraro, 1963), and snails (Shull, 1907). This coupled with its relatively high metabolism means that it is likely to be one of the most exposed terrestrial species in the areas of the former impoundments. Because of its small size, the short-tailed shrew has the greatest weight-specific rate of metabolism among mammals (Lawlor, 1979). Due to its relatively great ingestion rate, relatively small home range (varies from 0.39 ha to 0.96 ha; Buckner, 1966; Faust et al., 1971), and its relatively short lifespan (approximately 1 year; Russell, 1998), the short-tailed shrew has been recommended as a sentinel species of contaminant exposure based on trophic level and value as a biomonitor (Talmage and Walton, 1991). The shrew was selected as a surrogate species for wildlife that would be expected to have maximal exposure to PCBs in the soils.

Based on the currently available data, the lines of evidence that can be evaluated include: 1) presence and numbers of shrews in the study area compared to a reference location; 2) comparison of field-collected, whole body burdens of PCBs in shrews to literature-derived tissue residue-based toxicity reference values; and 3) comparison of estimated and/or measured site-specific dietary exposures for shrews to literature-derived toxicity reference values.

4.2 Derivation of Dietary-Based Toxicity Reference Values (TRVs) for PCB Exposure to Shrews

A few studies have reported concentrations of PCBs in shrews collected from PCB-contaminated sites. However, actual field verification of population-level effects due to exposure to PCBs or laboratory studies on reproductive effects due to exposure to PCBs have not been conducted with shrews. Controlled laboratory studies on the reproductive effects of PCBs have been conducted with other rodent species such as rats. From such studies, it is possible to derive no observable (NOAEL) and lowest observable adverse effect levels (LOAEL). An attempt was made to identify studies that evaluated sensitive ecologically relevant endpoints (primarily effects on reproduction), similar exposure routes (primarily dietary), exposure duration (chronic and/or during sensitive life stages) conditions as similar as possible to those expected at this site. An additional consideration was the use of the same or very similar species for derivation of both a dietary-based TRV and a tissue-based TRV. For this reason, the TRV for the white-footed mouse (discussed previously) was not deemed appropriate since matched dietary and tissue residue concentrations are not available in the peer-reviewed scientific literature.

A two-generation reproduction study was reviewed in which Sherman-strain rats were exposed by diet to five doses of Aroclor 1254 (0, 1, 5, 20, and 100 mg/kg in feed) for up to 274 d through a critical reproductive lifestage (Linder et al 1974). Conversions of concentrations in the diet to a daily dose were not necessary because this information was provided in the study. No adverse effects were observed at a dose level of 5 mg/kg in feed (or 0.32 mg/kg/d). While the TRVs are based on 2-generation reproductive studies, it should be pointed out that reproductive effects were not observed at dietary concentrations of up to 10 mg/kg in a three-generation reproductive study with rats cited as personal communication in Linder et al., (1974). Furthermore, since it is well known that Aroclor 1254 (from the laboratory study) is more toxic than Aroclor 1242 (the predominant Aroclor found in the KRAOC), the TRV is based on the more toxic of the potential Aroclors (Harris et al., 1993). Thus, the NOAEL value of 5 mg/kg (or 0.32

mg/kg/d) of Aroclor 1254 in feed should be considered a conservative estimate of the NOAEL. At 20 mg/kg in feed (or 1.5 mg/kg/d), adverse effects were observed including a reduction in litter size. Since the study considered dietary exposure during the sensitive and ecologically relevant time period of reproduction, the 0.32 and 1.5 mg/kg/d doses were considered to be chronic dietary-based NOAELs and LOAELs, respectively. However, studies have shown that numerous physiological functions such as metabolic rates, as well as responses to toxic chemicals, are functions of body size. For example, smaller animals usually have greater metabolic rates and are usually more resistant to toxic chemicals because of more rapid rates of detoxification or elimination (Sample et al., 1996).

NOAEL and LOAEL TRVs from rats (test species) were adjusted for differences in body size of shrews (wildlife species) according to equation 4-1 using methodology from Sample et al., (1996) and USEPA (1992). Using the mean body weight of control rats in the Linder et al., (1974) study (mean = 425 g) and the mean body weight of control shrews in the Russell (1998) study (mean = 19.4 g), the adjusted dietary-based TRVs for shrews are 0.69 and 3.2 mg/kg/d for the NOAEL and LOAEL, respectively.

$$NOAEL_{wildlife} = NOAEL_{test\ species} * [\text{body weight}_{test\ species}/\text{body weight}_{wildlife}]^{(1-0.75)} \quad \textbf{[Equation 4-1]}$$

4.3 Derivation of Tissue-Residue Based Toxicity Reference Values (TRVs) for PCB Exposure to Shrews

The next task was to identify studies from which a tissue residue-based TRV could be derived. One might ask the question: Why is it necessary to determine a tissue residue-based TRV when a dietary TRV is available? This was done to reduce the uncertainty on the exposure estimate. The tissue concentration is the most proximal measure of exposure and obviates the need to know the relative portions of dietary items, concentration of PCBs in these items, and assimilation efficiency. Furthermore, by knowing the actual body burden obviates the need to apply site use factors. This is important because the floodplain is flooded or frozen much of the year and shrews are not present. If data are available only for potential dietary items, there is considerably more uncertainty in the predicted exposure compared to a known exposure as determined by a site-specific whole body burden concentration of PCBs. Moreover, site-specific tissue residues take into account other factors such as reduced site use due to periodic flooding of the former impoundment floodplain.

While the study by Linder et al., (1974) did not measure tissue residues after exposure, tissue residues were measured in multiple tissues from a similar chronic study of rats exposed to Aroclor 1254 in the diet at dosage levels of 0, 2, 20, and 100 mg/kg for 246 d (Grant et al., 1974). The results of the two studies can be used to estimate tissue residue concentration of PCBs in rats chronically exposed to PCBs that correspond to the dietary NOAELs and LOAELs (Table 4-1). The rat data are based on individual organ concentrations in adipose, kidney, liver, blood, heart, and brain (Grant et al., 1974). Since shrew data from the field are based on whole body concentrations of PCBs, the best-matched comparison appears to be between lipid-normalized PCB concentrations in adipose tissue of rat [rat adipose tissue is composed of approximately 85% lipid (Haddad et al., 2000)] and lipid-normalized whole body PCB concentrations in field-collected shrews. Thus, the lipid-normalized tissue residue based LOAEL TRV for shrews exposed to PCBs is 73.3 mg/kg (Table 4-1). However, since tissue residue concentrations of PCBs were not measured at a concentration corresponding to the dietary-based NOAEL, a tissue residue-based NOAEL can not be established. Despite this, the tissue residue-based NOAEL is bounded in the study and can be assumed to fall between 30.7 and 73.3 mg/kg.

Table 4-1. Tissue residues from chronic, laboratory exposures of rats to varying concentrations of Aroclor 1254 in the diet.

	Total PCB Concentration (mg/kg)						
Diet[1]	**Blood[2]**	**Liver[2]**	**Brain[2]**	**Heart[2]**	**Adipose[2]**	**Adipose[3]**	**Comments[4]**
0	0.01	0.14	<0.00	0.03	3.3	3.9	
2	0.08	1.13	0.10	0.42	26.1	30.7	
5							NOAEL
20	0.05	4.77	1.13	1.59	62.3	73.3	LOAEL
100	0.35	57.04	3.50	5.98	347.5	408.8	

[1]Dietary concentrations of PCBs are from Linder et al., (1974) and Grant et al., (1974).
[2]Tissue residue concentrations of PCBs in rat are from Grant et al., (1974) reported on a wet weight basis.
[3]Lipid-normalized tissue residue concentrations of PCBs in rat adipose tissue normalized to 85% lipid based on lipid concentrations reported in rat adipose tissue (Haddad et al., 2000).
[4]NOAEL and LOAEL are from Linder et al., (1974). See text for details.

Another issue addressed was the relative uncertainty regarding the species sensitivity of shrews compared to other rodents, such as mice and rats. From the scientific literature, available data indicate that shrews are less responsive than other rodents to PCB-mediated effects (Villeneuve et al., 1972; Russell, 1998). In both of these studies, the endpoints included pentobarbital-induced sleeping times in PCB-exposed animals compared to reference animals, a non-invasive, yet sensitive and dose-dependent endpoint relating to induction of cytochrome P450 metabolizing enzymes. Pentobarbital-induced sleeping times are inversely proportional to the PCB exposure concentration. The pentobarbital-induced sleeping time-based LOAEL for rats, mice, and shrews indicate that shrews are less sensitive to PCBs than rats and mice (Table 4-2). Furthermore, for other endpoints, shrews have been shown to be consistently less sensitive to PCBs than mice in the only published study that has attempted to investigate effects of PCBs in shrews (Russell, 1998). In this study, short-tailed shrews were exposed daily to a mixture of Aroclors 1242 and 1254 (2:1 ratio) for 31 d at dosage levels of 0, 0.55, 9.78, and 214 mg/kg as measured in the diet. Also, in this study, white-footed mice were exposed daily to a mixture of Aroclors 1242 and 1254 (2:1 ratio) for 31 d at dosage levels of 0, 0.11, 1.30, and 25 mg/kg in the diet. Since two species were evaluated, comparisons of species sensitivities can be made. Endpoints that were measured include multiple enzyme activities, pentobarbital-induced sleeping times, protein concentrations of individual enzymes, body weight, relative liver weight, and relative spleen weight. The results from this study indicate that there were no significant differences in pentobarbital-induced sleeping times for shrews (there was a significant difference for white-footed mice at 25 mg/kg in diet), no significant differences in final body weight, body weight gain, relative liver weight for shrews (there was a significant difference for white-footed mice at 25 mg/kg in diet), and no significant differences in relative spleen weight or hepatic microsomal protein content for shrews (there was a significant difference in white-footed mice at 25 mg/kg in diet). The only statistically significant differences observed in shrews exposed to PCBs were induced monooxygenase (EROD, BROD, and MROD) activities in liver microsomes of short-tailed shrews at 214 mg/kg in the diet. For comparison, statistically significant induction of monooxygenase activity occurred at concentrations as low as 1.3 mg/kg in the diet of white-footed mice. It is important to note that the behavioral and biochemical endpoints (i.e., phenobarbital-induced sleep time, EROD, BROD, and MROD activities) discussed here are not necessarily ecologically relevant and are used only to demonstrate species sensitivity. Thus, taken together, the available evidence demonstrates that short-tailed shrews are less sensitive to the effects of PCBs than other rodents. Despite this indication of less sensitivity of shrews relative to other rodents, the TRVs were maintained at the same level as other rodents except for the allometric adjustment to account for differences in body weights of shrews and rats.

Table 4-2. Differences in species sensitivity to PCB-induced changes in pentobarbital-induced sleeping time.

Species	LOAEL	Reference
Rat	20 mg/kg in diet	Villeneuve et al., (1974)
White-footed mouse	25 mg/kg in diet	Russell (1998)
Short-tailed shrew	> 214 mg/kg in diet	Russell (1998)

4.4 Small Mammal Trapping Results

The trapping success for small mammals was greater at the former Trowbridge impoundment locations (0.0446 small mammals per trap night) compared to the Fort Custer (reference) locations (0.0125 small mammals per trap night) when normalized to the level of effort (trap nights) at each location (Table 4-3). It should be noted that trapping was stopped after collecting the pre-specified sample size of $\geq$ 12 samples per location and per sampling period. Of all of the small mammals, 17 shrews were collected from the former Trowbridge impoundment and 16 shrews were collected from the Fort Custer (reference) locations. Approximately 60 % of the shrews were collected in Sherman live traps and the remainder were collected in pitfall traps.

4.5 Risk Characterization for Short-Tailed Shrews

Hazard quotients (HQs) were calculated for short-tailed shrews based on dietary exposures and based on tissue residues (Table 4-4). The hazard quotients range from 0.06 to 0.84 depending on the approach and the assumptions that were utilized. The degree of uncertainty is greater for the dietary exposure model compared to the tissue residue model because of uncertainty in site use assumptions and because data are not yet available for many of the site-specific dietary items of shrews such as crickets, beetles, and snails. As this information becomes available, it will be used to refine the exposure calculations.

An additional consideration is a comparison of field measured concentrations of PCBs in shrews collected from the former Trowbridge impoundment (Table 4-5) to concentrations of PCBs in shrews exposed in laboratory studies (Table 4-6). The concentrations of PCBs in shrews from the field correspond most closely to those from a laboratory exposure of shrews to 0.55 mg PCBs/kg in the diet, a concentration that is well below comparative concentrations necessary to elicit adverse effects in other rodents, which are approximately 5-20 mg PCBs/kg in the diet (Linder et al., 1974; Grant et al., 1974). Similarly, the predicted dietary exposure of 0.18-0.33 mg/kg/d (based on concentrations of PCBs in dietary items) yields a predicted tissue residue concentration of approximately 3.9-30.7 mg/kg (whole body, lipid basis) which is very similar to the observed range of 1-109 mg/kg (whole body, lipid basis) in field-collected shrews. Thus, the available lines of evidence are consistent and adverse effects in shrews would not be expected.

Taking into consideration the conservative nature of the TRVs and the relatively small HQ values from multiple methods of evaluation, the risk posed to shrews by concentrations of PCBs in the soils of the former impoundments is minimal. This, coupled with observations of viable populations of shrews and other small mammal species collected from the former Trowbridge impoundment, indicates that there are no measurable, population-level effects. Thus, no effects would be predicted and none seem to be observed.

Table 4-3. Results of small mammal trapping at the former Trowbridge impoundment and at Fort Custer (reference) locations collected during the 2000 summer season.

Location	# Trap nights[1]	Shrews[2]	Meadow Voles	Deer Mice	White-footed Mice	Oldfield Mice	Jumping mice	Chipmunks and squirrels	Total # of small mammals
Trowbridge 1	784	4	16	6			1		
Trowbridge 2	392	3	6	17					
Trowbridge 3	588	4	14	6			1		
Trowbridge 4	588	6	11	7	1		1	1	
All Trowbridge	**2352**	**17**	**47**	**36**	**1**	**0**	**3**	**1**	**105**
Fort Custer 1	2610	6		12		1	1	4	
Fort Custer 2	1305	10		3	3		4	5	
All Fort Custer	**3915**	**16**	**0**	**15**	**3**	**1**	**5**	**9**	**49**

[1]Number of trap nights was calculated as the number of traps set out multiplied times the number of nights that they were in place.
[2]Shrews include both the short-tailed shrew and the masked shrew.

Table 4-4. Hazard quotients for shrews calculated based on dietary exposure modeling and based on whole body concentrations of PCBs in shrews from the former Trowbridge impoundment within the KRAOC.

	Dietary-Based Hazard Quotient		Tissue Residue-Based Hazard Quotient
Exposure Level	**NOAEL-Based**	**LOAEL-Based**	**LOAEL-Based**
Geometric mean	0.26	0.06	0.43
95% UCL geometric mean	0.48	0.10	0.84

Refer to Appendices B and C for more details on the dietary exposure model and risk calculations.

Table 4-5. Summary of shrew data collected from the former Trowbridge impoundment with the KRAOC.

Species	Lab ID	% Lipid	Total PCB Concentration (mg/kg, ww)	Lipid-Normalized Total PCB Concentration (mg/kg, lipid)
Short-tailed shrew	TBSM053	5.23	0.132	2.52
Masked shrew	TBSM061	4.3	0.887	20.6
Short-tailed shrew	TBSM072	2.22	2.43	109
Short-tailed shrew	TBSM092	3.16	2.34	74.1
Short-tailed shrew	TBSM100	1.88	3.04	162
Short-tailed shrew	TBSM013	1.42	1.12	110
Short-tailed shrew	TBSM024	1.46	1.33	97.9
Short-tailed shrew	TBSM028	2.8	3.12	111
Short-tailed shrew	TBSM046	1.67	0.411	24.6
Short-tailed shrew	TBSM081	2.18	1.05	48.2
Short-tailed shrew	TBSM086	1.51	1.18	78.1
Short-tailed shrew	TBSM090	4.26	1.2	28.2
Short-tailed shrew	TBSM091	4.03	1.18	29.3
Short-tailed shrew	TBSM099	3.3	0.726	22.0
Short-tailed shrew	TBSM094	2.9	1.91	65.9
Short-tailed shrew	TBSM054	2.52	0.0251	1.00
Masked shrew	TBSM052	4.72	0.221	4.68
Geometric mean			0.84	31.4
95% UCL geometric mean			1.52	61.8

Table 4-6. Body burden concentrations of PCBs in short-tailed shrews from a controlled, laboratory study in which shrews were exposed to different concentrations of PCBs in the diet for 31 d.

Total PCB Concentration (mg/kg)		
Dietary Exposure (in feed)	**Mean whole body**	**Whole body (lipid-normalized)**
Control	0.31	14.3
0.55	1.88	82.1
9.78	27.4	1,390
214	291	14,600

Data from Russell (1998)

5.0 GREAT HORNED OWL

5.1 Introduction

Great horned owls were evaluated in the MDEQ ERA and were found to have dietary-based hazard quotients less than 1.0 when compared to either the LOAEL or NOAEL. However, in the addendum to the MDEQ ERA, risks to owls were reconsidered based on concentrations of PCBs in three great horned owl eggs collected from the area downstream of Lake Allegan in the early 1990's. MDEQ utilized this data and a tissue-based toxicity reference value (TRV) in a top-down analysis to provide the basis of a soil cleanup value. However, the paper from which their TRV was derived does not provide support for the TRV value for great horned owls in the MDEQ ERA (for further discussion, refer to the Draft Interim Giesy Ecotoxicology ERA (2000)). Furthermore, based on the locations from where the owl eggs were collected (i.e., Allegan State Game Area), it is unlikely that the former impoundment flood plain soils were a major source of PCBs that were accumulated into the owl eggs. Until a link to the former impoundment flood plain soils can be established, the top-down analysis based on great horned owls is inappropriate to determine a target soil threshold for protection of sensitive higher trophic level receptors.

Predicted risks to great horned owls from dietary exposure to PCBs in the KRAOC from both terrestrial and aquatic exposure pathways were previously found to be insignificant as discussed in the Draft Interim Giesy Ecotoxicology ERA (2000). To provide additional lines of evidence for consideration in the baseline ERA and to further address the potential risks of PCBs to great horned owls, field studies are being conducted to address site-specific productivity, dietary composition, and dietary exposure to PCBs.

5.2 Preliminary Data

Twelve great horned owl nesting cones (nests) have been located at Fort Custer (reference site) and twelve at the former Trowbridge impoundment (study site). MSU-ATL personnel placed all nests in the field during 2000 and early 2001. Preliminary field observations indicate that at least one nest is occupied at Fort Custer with one 6 wk old nestling. Two nests are occupied at the former Trowbridge impoundment, each with two 6 wk old nestlings. MSU-ATL personnel collected blood samples (with oversight provided by USFWS) from each of the nestlings. Additional information on these nests is provided (Table 5-1). Also during 2001, two great horned owl eggs were collected from nests at Allegan State Game Area and a single egg was collected from a nest near Ceresco. PCB concentrations will be determined in these eggs and reported in a future update.

Each study area was searched on foot from December through February to confirm nest occupancy. Hooting call surveys were utilized to locate additional territorial owl pairs. Preliminary survey data indicates that great horned owl populations are similar between the former Trowbridge impoundment and Fort Custer (reference) locations (Table 5-2)

In addition, PCB analyses for dietary items during 2000 and 2001 is complete and data analysis is nearly complete for site-specific dietary composition for great horned owls, great horned owl survey data, and concentrations of PCBs in great horned owl eggs collected this year. A preliminary summary of confirmed, site-specific dietary items for great horned owls residing in the KRAOC has been compiled (Table 5-3 and Table 5-4). Note that dietary items for the great horned owl have been identified by three different methods: 1) prey remains observed in nest during serum collection from great horned owl fledglings; 2) prey remains collected at the base of a known great horned owl nest tree; and 3) prey remains collected at the base of a known great horned owl perch tree.

The data on dietary composition will be used in determining the site-specific exposure concentrations for great horned owls residing in the KRAOC. Data on PCB concentrations from most of the individual dietary items are not currently available. It is important to note, however, that the dietary pathways of the great horned owls are not strictly based on terrestrial prey as demonstrated by the presence of muskrat remains. This evidence is consistent with several reports in the literature of great horned owls consuming biota linked to aquatic ecosystems such as fish, waterfowl, muskrats, and great blue herons (Wink et al., 1987; Devine and Stevens, 1985; Gretch, 1987). This is the likely reason that the results of the "top down" approach and "bottom-up" approaches used by CDM/MDEQ could not be reconciled (MDEQ, 2000). Thus, it may be inappropriate to determine a target soil threshold for protection of sensitive higher trophic level receptors based on great horned owls since their exposures are linked to both aquatic and terrestrial sources.

5.3 Preliminary Risk Characterization

As described in the Draft Interim Giesy Ecotoxicology ERA (2000), the available dietary-based (see Table C-3 of Appendix C) and tissue residue-based approaches, although preliminary, do not support the conclusion that PCBs are adversely affecting great horned owl populations in the KRAOC.

Table 5-1. Locations of great horned owl nests and associated data for Spring, 2001.

Location	GPS Coordinates	Nest ID	Nest type	Sampling Date	# Nestlings	Data Collected
Fort Custer	N 42 19.322 W 85 21.872	FC#04CO	Cone nest	04/24/2001	1	Blood sample
Trowbridge	N 42 28.245 W 85 45.149	TB#02CO	Cone nest	04/24/2001	2	Blood sample
Trowbridge	N 42 29.059 W 85 46.825	TB#10CO	Cone nest	04/24/2001	2	Blood sample

Table 5-2. Owl survey data in the KRAOC for years 2000-2001.

Location	# of GH Owls Responding Per Survey	# of Occupied GH Owl Breeding Territories Per Survey
Fort Custer	3.25	2.1
Trowbridge	3.83	2.5

Table 5-3. Confirmed, site-specific dietary items for great horned owls residing in the KRAOC for years 2000-2001.

Dietary Item	Sample Type[1]
Aves (Birds)	
Grackle (*Quiscalus*)	Prey remains in nest
Crow (*Corvus*)	Prey remains in nest, feeding perch
Starling (*Sturnus*)	Prey remains in nest, base of nest tree
Chicken (*Gallus*)	Feeding perch
Unknown	Prey remains in nest, base of nest tree, feeding perch
Mammalia (Mammals)	
Shrew (*Blarina*)	Base of nest tree, feeding perch
Mole (*Talpidae*)	Base of nest tree, feeding perch
Mice (*Peromyscus*)	Prey remains in nest, base of nest tree, feeding perch
Vole (*Microtus*)	Base of nest tree, feeding perch
Muskrat (*Ondata*)	Prey remains in nest, base of nest tree, feeding perch
Squirrel (*Sciurus*)	Prey remains in nest, base of nest tree, feeding perch
Rabbit (*Sylvilagus*)	Prey remains in nest, base of nest tree, feeding perch
Unknown	Prey remains in nest, base of nest tree, feeding perch

[1]Prey remains in nest refers to visual confirmation of dietary items in nest at time of blood sampling from great horned owl nestlings. If no edible portion was remaining on the dietary item, the remains of the dietary item were collected. Great horned owl pellets were collected from either the base of the nest tree or from a feeding perch located less than 30 m from the nest tree.

Table 5-4. Prey item occurrence determined for great horned owls residing in the KRAOC during the years 2000-2001.

Dietary Item	Percent Occurrence[1]
Aves (Birds)	27%
Mammalia (Mammals)	73%

[1]Percent occurrence determined as composite from all sample types (e.g., prey remains in nest, pellets from the base of nest tree, and pellets from the feeding perch).

6.0 TREE SWALLOW

6.1 Introduction

During the summer of 2000, productivity measurements were conducted on migratory songbirds, specifically tree swallows, nesting in the Ft. Custer State Recreation Area and the former Trowbridge impoundment. Nest boxes at each site are located within the same proximity to the Kalamazoo River, and a similar level of effort was conducted at each site. Birds at Ft. Custer experienced more human disturbance than those at the former Trowbridge impoundment. Ft. Custer eggs were also handled by a larger number of study personnel, contributing to the loss of 6 eggs broken during the study, as opposed to the loss of one egg at the former Trowbridge impoundment.

6.2 Preliminary Data

All data taken during the 2000 field season regarding passerine productivity is considered preliminary in nature with the statistical analysis pending (Table 6-1). One confounding factor in data interpretation is that the tree swallow nesting population at Ft. Custer was established almost 30 years ago, while that at the Trowbridge site was established in April 2000. Lower productivity from less established sites is likely related to female age-related differences that have been well documented in the literature (De Steven, 1978; Stutchbury and Robertson, 1988; Wheelwright and Schultz, 1994). This is the most likely the reason for the number of active nests at Trowbridge being half that of Ft. Custer. The well-established population at Ft. Custer also contains older females that are returning to the area. The older females will often have larger broods because they are better skilled at nest building, hatching, and nestling care. In order to eliminate the confounding effects of female experience on the interpretation of the data, the proportion of after hatch year females to second year females must be quantified. The age of the females was not recorded during the 2000 season.

The 2001 season will be used to make comparable productivity measurements between the sites. The establishment of the nesting population at the former Trowbridge impoundment and the identification of female age will eliminate most physiological explanations for differences in productivity between the sites. Data that is now being gathered is expected to be a more reliable indicator of PCB related effects on passerine productivity.

Table 6-1. Preliminary tree swallow bird box data from the 2000 field season in the KRAOC.

Productivity Measurements	Trowbridge[5]	Fort Custer[5]
Hatching Success [1]	81.3%	76.6%
Fledging Success [2]	92.3%	97.2%
Egg to Fledging Success [1,3]	75.0%	74.5%
Avg. # eggs/Nest	3.7 (1.4)	5.0 (1.1)
Avg. # Nestlings/Nest [4]	4.0 (1.5)	3.8 (1.2)
# Active Nests	10	20
# Abandoned Nests	2	1
Avg. Egg Weight	1.9 (0.2)	1.7 (0.23)
Avg. # Eggs Hatched/Nest [1]	2.9 (2.0)	3.6 (1.4)
Avg. # Nestlings Fledged/Nest [4]	3.4 (1.5)	3.7 (1.1)
# Adult Visits/30 min.	2.5 (2.0)	3.3 (3.2)
Avg. Nestling Age During Visits	7.5 (5.0)	9.2 (5.7)

[1] Excluding eggs broken by study personnel
[2] Percent of chicks to fledge
[3] Percent of eggs to fledge
[4] Excluding nests abandoned during egg laying
[5] Values followed by parentheses are mean values and standard deviations (in parentheses)

7.0 MINK

7.1 Introduction

Mink are one of the most sensitive organisms to the effects of PCBs (Giesy *et al*., 1994). Available literature was reviewed and evaluated for the most appropriate comparison to mink exposed to PCBs from the KRAOC. The most sensitive endpoint in mink exposed to PCBs is reproduction.

7.2 Exposure Models for Mink

At least four methods of estimating exposure to mink will be evaluated in the BERA, including two dietary models, one for total PCBs and the other for TEQs, and one trophic transfer model (Figure 7-1). Note that data are not currently available to utilize each of these methods. However, as data become available, the results from each of these methods will be incorporated as multiple lines of evidence. The results of each of these exposure models will then be compared to corresponding reference doses for calculation of hazard quotients. For mink, a diet composition was assumed to be 33.3% fish (approximately same as 30% value in Heaton et al, 1995), 33.3% crayfish, and 33.3% small mammals.

In addition to the determination of total concentrations of PCBs and TEQs in the diets of mink, concentrations of these residues have been measured in tissues of mink. Thus, measured tissue residue concentration data will serve as a fourth method of assessing exposure and as a means to compare the modeled values. The justification for the use of tissue concentrations is the same as that given for shrews.

7.2.1 Dietary Intake of Total PCBs for Mink

Exposure to total PCBs in fish and other prey items (ww basis) will be calculated as for other receptors by calculating an ADD_{pot}. Site-specific concentrations of PCBs in dietary items of mink from the KRAOC from MSU-ATL field studies will be available at a later date.

7.2.2 Dietary Intake of TEQs for Mink

Another model of exposure will be dietary intake of TEQs from fish. Site-specific concentrations of TEQs in dietary items of mink from the KRAOC from MSU-ATL field studies will be available at a later date. This model of exposure will be accomplished by multiplying congener concentrations in fish and other prey items (on a wet weight basis) times congener-specific toxic equivalency factors (TEFs; van den Berg *et al*., 1998) and then summing the products to get total TEQs as shown (equation 7-1):

$$\text{ng TEQ/kg fish} = \Sigma[\text{conc. of PCB \#}n \text{ in fish (ng/kg, ww) x TEF for PCB \#}n] \qquad \textbf{Equation 7-1}$$

where n = any PCB congener for which there are both data for concentrations in fish and a TEF value

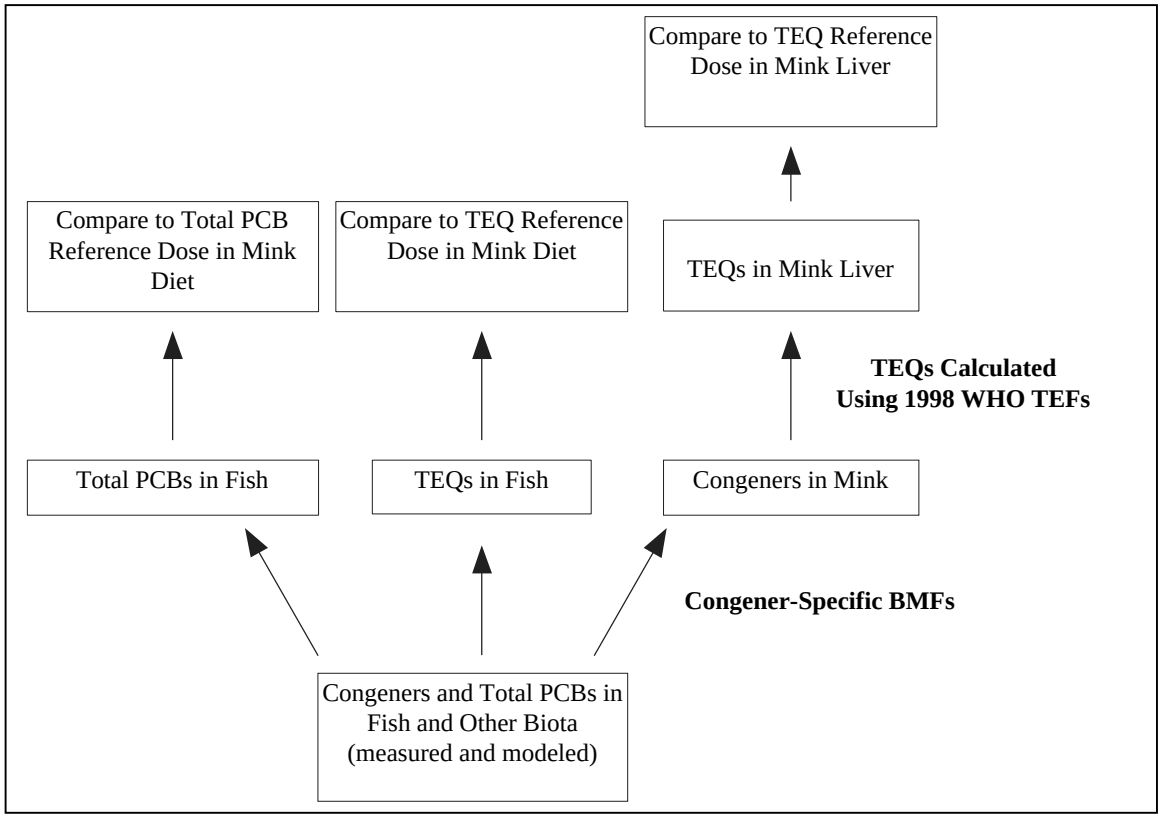

Figure 7-1. Conceptual model for congener-specific uptake of PCBs into mink from fish and other prey items. Abbreviations are as follows: TEQs, TCDD-equivalents; WHO, World Health Organization; TEFs, toxic equivalency factors; BMFs, biomagnification factors.

7.2.3 Congener-Specific Trophic Transfer Model for Mink

The trophic transfer model utilizes congener-specific biomagnification factors (BMFs) to model the movement of dioxin-like PCB congeners from fish and other prey items into mink liver. The accumulation of dioxin-like PCB congeners will be conducted on a lipid-normalized basis, which is the most appropriate method of modeling hydrophobic compounds, such as PCBs, through a food web. When dioxin-like PCB congener data becomes available for field collected mink from the KRAOC, the trophic transfer model can be compared to measured concentrations as a means of "ground truthing" the model. Concentrations of dioxin-like PCB congeners in fish and other prey items will be utilized (when data becomes available) and congener-specific BMFs from a mink feeding study (Tillitt et al., 1996), to calculate the concentrations of dioxin-like PCB congeners in mink liver. These concentrations will then be multiplied by their respective TEFs and summed to yield mink liver TEQs as shown (equation 7-2):

$$\text{Dose} = \text{Diet Fraction} * \sum_{77}^{189} (\text{PCB}_n * \text{BMF}_n \text{ Lipid Fraction} * \text{TEF}_n) \qquad \textbf{Equation 7-2}$$

where:

Dose = total concentration of TEQs in mink liver lipid (pg TEQ/g mink liver, ww);

Diet Fraction = portion of mink diet that is comprised of fish or other prey species (0.333 g fish, ww/g food, ww);

[PCB_n] = concentration of each dioxin-like PCB congener in whole fish (pg PCB_n/g fish lipid);

BMF_n = lipid-normalized fish-to-mink liver biomagnification factor (Tillitt et al., 1996);

Lipid Fraction = lipid fraction of mink liver (g lipid/g liver, ww); and

TEF_n = 2,3,7,8-TCDD equivalency factor (van den Berg *et al*., 1998)

n = specific dioxin-like PCB congener

7.3 Concentrations of PCBs in Mink Liver from the KRAOC

PCB concentrations in livers of mink collected from various locations within the KRAOC range from 0.039 to 4.87 mg/kg (wet weight; Table 7-1). Only one sample (Mink0010) exhibited any abnormal pathology. In this sample, fatty liver was apparent. The lipid content of this sample was 53.7 %, which confirms the presence of elevated lipids. Field notes from the trapper relating to this sample indicate that its appearance was distinct from other samples by the coloration of the fur (trapper notes: “Black mink-? Ranch.”). The nearest mink ranch is located in Shelbyville, which is approximately 10 miles north of Plainwell and the owners acknowledge that mink occasionally do escape and they are not tagged. While it can not be confirmed whether or not this sample is from a mink ranch lineage, it does raise the question of why this mink exhibited fatty liver. Since fatty liver is a well-known condition associated with PCB exposure, it is noteworthy that this sample contained a relatively low concentration of PCBs (0.04 mg/kg wet weight) compared to other samples that did not exhibit fatty liver. Fatty liver can also be caused by excessively high fat content of the diet and is often observed in ranch-raised mink (D. Aulerich, pers. comm.).

Table 7-1. Summary of mink liver data collected from the KRAOC.

Lab ID	Location	% Lipid	Total PCB Concentration (mg/kg, ww)	Total PCB Concentration (mg/kg, lipid)
Mink0001	"D" Ave. Bridge	4.48	3.36	75.0
Mink0006	Near "B" Ave., S. of Silver Creek	4.19	4.81	115
Mink0007	Between "B" Ave. and Baseline Rd., ~ 200 m upstream from Oxbow	11.92	0.953	7.99
Mink0008	Between "B" Ave. and Baseline Rd., ~ 200 m upstream from Oxbow	7.72	5.98	77.5
Mink0009	Between "B" Ave. and Baseline Rd., off 4th St., near Oxbow	5.21	0.0428	0.821
Mink0010	200 m downstream of Plainwell Dam	53.7	0.0404	0.075
Mink0011	Upstream edge of Otsego Dam	4.01	1.62	40.4
Mean	**Study area combined**	13.0	2.40	45.2
95% UCL Mean	**Study area combined**	26.5	4.15	78.6
Mink 1001	Appx. SW corner of Ft. Custer State Park (reference site)	4.04	1.54	38.1

7.4 Derivation of Dietary-Based Toxicity Reference Values (TRVs) for PCB Exposure to Mink

Pending

7.5 Derivation of Tissue-Residue Based Toxicity Reference Values (TRVs) for PCB Exposure to Mink

The tissue residue-based reference doses for mink are those Halbrook et al., (1999). The TRV from Halbrook et al., (1999) is based on reproductive toxicity in mink from dietary exposure of an environmentally weathered PCB mixture in fish from Poplar Creek (located on the Oak Ridge Reservation in Tennessee). Halbrook et al., (1999) reported that a mean concentration of PCBs in mink liver of 7.25 mg/kg from a chronic dietary study was the LOAEL. The only statistically significant adverse effect at this liver concentration was a decrease in male kit weight. All other measured endpoints were not statistically significant such as female kit weights, litter size, number born alive, and number alive at six weeks (Halbrook et al., 1999). Since the values from this study are suitable for comparison in the current analyses (i.e., same species, and same exposure route, similar mixture, etc.), no uncertainty factors have been applied to determine a LOAEL. However, a NOAEL was not available from the same study. In situations where a chronic NOAEL value for an ecologically relevant endpoint is not available, USEPA guidance states that a NOAEL can be estimated by a LOAEL to NOAEL uncertainty factor of between 1 and 10 (USEPA, 1995). In this ERA for mink, a factor of 3 was selected in order to convert the LOAEL value of 7.25 mg/kg in mink liver to a NOAEL value of 2.42 mg/kg in mink liver. A factor of 3 was selected rather that a greater factor since the LOAEL is based on conservative endpoints that are protective of effects on reproduction.

Another study that was evaluated for tissue residue-based TRVs for mink exposed to PCBs was that of Tillitt et al., (1996) which was based on reproductive toxicity in mink exposed through the diet to an

environmentally weathered PCB mixture in carp from Saginaw Bay, MI. However, there is substantial uncertainty regarding TRVs for total PCBs from Tillitt et al., (1996) because other contaminants were present in fish from Saginaw Bay which may not be present at the same concentrations in dietary items of mink from the KRAOC. For example, the contribution of polychlorinated dibenzo-p-dioxins (PCDDs) and polychlorinated dibenzofurans (PCDFs) towards the tissue residue-based TRVs in mink liver reported in Tillitt et al., (1996) was greater than 90%. In other words, the TRVs from Tillitt et al., (1996) may be substantially influenced by the presence of relatively great concentrations of PCDDs and PCDFs in Saginaw Bay fish from this study. Thus, since there is considerable uncertainty in tissue residue-based effect concentrations in mink tissue for PCBs from the Tillitt et al., (1996) study, the TRVs for this ERA are those derived from the Halbrook et al., (1999) study.

7.6 Preliminary Risk Characterization for Mink

Although the data are not yet available for predicted risks from dietary exposure to PCBs or TEQs, the available tissue residue data indicate that tissue residue-based hazard quotients for mink range from approximately 0.33 to 1.71 for mink captured from areas proximal to the former impoundments and approximately 0.21 to 0.64 for mink captured from reference locations (Table 7-2).

Table 7-2. Hazard quotients for mink calculated based on concentrations of PCBs in mink liver collected from the KRAOC and tissue residue-based effect levels.

Exposure Level	Hazard Quotients (NOAEL)	Hazard Quotients (LOAEL)
Study Area Combined		
Mean	0.99	0.33
95% UCL Mean	1.71	0.57
Fort Custer	0.64	0.21

8.0 BALD EAGLE

8.1 Introduction

Field data collection for years 2000 and 2001 is complete and data analysis is nearly complete for site-specific dietary composition for bald eagles. A preliminary summary of confirmed, site-specific dietary items for bald eagles residing in the KRAOC has been compiled (Table 8-1). Note that dietary items for the bald eagle have been identified by two different methods: 1) prey remains observed in nest during serum collection from bald eagle fledglings; 2) observations of prey items delivered to the bald eagle nest. The data on dietary composition will be used in determining the site-specific exposure concentrations for bald eagles residing in the KRAOC. Data on PCB concentrations from most of the individual dietary items are not currently available.

8.2 Preliminary Data

Preliminary bald eagle productivity data from the KRAOC includes one fledged chick and one addled egg from Ottawa Marsh in 2000, and one fledgling from the Swan Creek Highbanks nest in 2001. Data from reference locations will be presented in a subsequent update.

Table 8-1. Confirmed, site-specific dietary items for bald eagles (Ottawa Marsh nest) residing in the KRAOC for year 2000.

Prey Item Observation[1]		Prey Remains from Nest[2]	
n	**Description**	**n**	**Description**
16	**Fish**	27	**Fish**
	Smallmouth bass (1)		N. Pike (1)
	Sunfish (1)		Catfish/bullhead (21)
	Bullhead (1)		Unconfirmed, but likely
	Unknown (13)		bowfin, bass, and sunfish (5)
1	**Birds**	2	**Birds**
	Unknown (1)		Mallard (1)
			Unknown (1)
3	**Mammals**	3	**Mammals**
	Muskrat (2)		Muskrat (2)
	Unknown (1)		Raccoon (1)
2	**Reptiles**	8	**Reptiles**
	Turtles – unknown species (2)		Musk turtles (5)
			Map turtles (2)
			Painted turtles (1)
10	**Unidentified items**		
32	**Total # prey items**	**40**	**Total # prey items**

[1]Prey items delivered to nest (observed by field personnel equipped with spotting scopes).
[2]Prey remains from nest collected during serum collection from bald eagle fledglings.

9.0 AQUATIC PLANTS

Data on aquatic plants will be utilized in the food web modeling (Table 9-1).

Table 9-1. PCB concentrations in aquatic plants.

Plant	Lab ID	Total PCB conc. (µg/g wet wt.)	Sample size
Cattail	TBPS001	0.0097	1
Loosetrife	TBPS002	0.0053	1
Lilly Pad	TBPS003	0.0466	1
Arrowhead	TBPS004	0.0405	1
Milfoil	TBPS005	0.0364	1
Mean conc.	TBPS006	**0.0277**	

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Appendix A

Dietary exposure risk calculations and related documentation.

Derivation of Ingestion Rates of Invertebrates and Fruit in the Diet of the American Robin

American robins (*Turdus migratorius*) and other wildlife are typically exposed to residues from multiple dietary items with different concentrations of residues in each item. To estimate food ingestion rates for the calculation of total doses, the information that is needed for each dietary item includes the residue concentration, the caloric content, and the assimilation efficiency. This information, taken together with information from the USEPA Exposure Factors Handbook and the peer-reviewed scientific literature, can be used to calculate exposure.

> Robins, as well as other fruit-eating birds, exhibit a low digestive efficiency for fruits. Karasov and Levey (1990) estimated the metabolizable energy coefficient (MEC) (i.e., the proportion of food energy that actually is assimilated) for robins eating a mixed fruit diet to be only 55 percent, perhaps because of the low retention time of the digested matter in the gut (Levey and Karasov, 1992). The short retention time might actually be an adaptation to eating fruit because large quantities of fruit must be processed to obtain an adequate protein intake. In contrast, when eating insects, robins (as well as other bird species) exhibit a higher digestive efficiency of approximately 70 percent (Levey and Karasov, 1989). Moreover, the energy content of insects tends to be higher than that of most fruits, particularly on a wet weight basis. [from USEPA, 1993]

The USEPA Exposure Factors Handbook reports values of 0.89 and 1.52 g/g/d for normalized ingestion rates for robins based on two studies in which robins were maintained on a fruit-only diet. From these two values, the average normalized ingestion rate for robins on a fruit-only diet is 1.2 g/g/d (USEPA, 1993). Using the gross energy content and assimilation efficiency data from the USEPA Exposure Factors Handbook, the metabolizable energy (ME) was calculated for fruits and invertebrates in the diet of robins (Table A-1). Thus, since the normalized ingestion rate is known for robins on a fruit-only diet and since the ME of fruit is known for robins and since the free-living metabolic needs of robins can be assumed to be independent of the dietary composition, then the ingestion rate for invertebrates can be determined by multiplying the ingestion rate for fruit by the ratio of ME of fruit to the ME of invertebrates as shown in the following equations:

$$NIR_{invertebrates}\ (g/g/d) * ME_{invertebrates}\ (kcal/g) = NIR_{fruit}\ (g/g/d) * ME_{fruit}\ (kcal/g)$$

$$NIR_{invertebrates}\ (g/g/d) = NIR_{fruit}\ (g/g/d) * [ME_{fruit}\ (kcal/g)/ME_{invertebrates}\ (kcal/g)]$$

$$NIR_{invertebrates}\ (g/g/d) = 1.2\ (g/g/d) * [0.61\ (kcal/g)/0.96\ (kcal/g)]$$

$$NIR_{invertebrates}\ (g/g/d) = 0.76\ (g/g/d)$$

Where:

NIR = normalized ingestion rate (g/g/d)
GE = gross energy content (kcal/g, wet weight)
AE = assimilation efficiency (%)
ME = metabolizable energy (kcal/g, wet weight) = GE*AE (for each dietary item)

Table A-1. Calculation of the metabolizable energy of invertebrates and fruit in the diet of the American robin (*Turdus migratorius*).

Food Type	GE (kcal/g, ww)	AE (%)	ME (kcal/g, ww)
earthworms	0.8[A]	0.72	0.58
crickets	1.7	0.72	1.2
beetles	1.5	0.72	1.1
invertebrates (average)			0.96
fruit	1.1	0.55	0.61

[A]Value is in the middle of the range (0.78-0.83) stated in the USEPA Wildlife Exposure Factors Handbook (1993).

Supporting Information for Risk Calculations Using MDEQ Assumptions

Exposure profiles, data sets with descriptive statistics, and risk characterization for the American robin, white-footed mouse, short-tailed shrew, and great horned owls are provided in the following tables.

Table A-2. Exposure profile and risk characterization for American robins (using assumptions of CDM/MDEQ).

HQ calculations for robins using CDM/MDEQ exposure assumptions and modeled concentrations of PCBs in terrestrial plants

Receptor	Dietary item	Dietary fraction (DF)	Dietary item PCB conc. (mg/kg)	Total dietary PCB conc. (mg/kg FW diet)	IR (g/d)	BW (g)	APDD[1] (mg/kg/d)	Toxicity Reference Value NOAEL	Toxicity Reference Value LOAEL	Hazard Quotient[2] NOAEL-based	Hazard Quotient[2] LOAEL-based
Robin	Terr. plants	0.49	4.7	2.303	92	77	2.75				
	Terr. inverts	0.51	1.5	0.77	92	77	0.91				
	Total						3.67	0.6	6.0	6.11	0.61

HQ calculations for robins using CDM/MDEQ exposure assumptions* and measured concentrations of PCBs in terrestrial plants**

Receptor	Dietary item	Dietary fraction (DF)	Dietary item PCB conc. (mg/kg)	Total dietary PCB conc. (mg/kg FW diet)	IR (g/d)	BW (g)	APDD[1] (mg/kg/d)	Toxicity Reference Value NOAEL	Toxicity Reference Value LOAEL	Hazard Quotient[2] NOAEL-based	Hazard Quotient[2] LOAEL-based
Robin	Terr. plants	0.49	0.033	0.02	92	77	0.02				
	Terr. inverts	0.51	1.5	0.77	92	77	0.91				
	Total						0.93	0.6	6	1.56	0.16

HQ calculations for robins using corrected CDM/MDEQ exposure assumptions* and measured concentrations of PCBs in terrestrial plants****

Receptor	Dietary item	Dietary fraction (DF)	Dietary item PCB conc. (mg/kg)	Total dietary PCB conc. (mg/kg FW diet)	IR (g/d)	BW (g)	APDD[1] (mg/kg/d)	Toxicity Reference Value NOAEL	Toxicity Reference Value LOAEL	Hazard Quotient[2] NOAEL-based	Hazard Quotient[2] LOAEL-based
Robin	Terr. plants	0.49	0.033	0.02	92	77	0.02				
	Terr. inverts	0.51	1.5	0.77	58.5	77	0.58				
	Total						0.60	0.6	6	1.00	0.10

*CDM/MDEQ exposure assumptions are from the report entitled, "Final Baseline Ecological Risk Assessment - Allied Paper Inc./Portage Creek/Kalamazoo River Superfund Site - June 1999"

**Measured concentrations of PCBs in terrestrial plants are from Michigan State University's Aquatic Toxicology Laboratory (MSU/ATL) from samples collected in 2000.

***Exposure assumptions were corrected for differences in intake rates for invertebrates versus plants in diet of robins (USEPA, 1993). The average consumption rate of plants in the diet of robins is 1.2 g/g BW/d (92 g/d; USEPA, 1993) consumption rate of invertebrates in diet of robins is calculated from the plant rate modified for differences in both assimilation efficiency and energy content of food = 0.76 g/g BW/d (58.5 g/d)

[1]Average potential daily doses (APDD) were calculated according to the following equation:

$$ADD_{pot}\ (\text{mg/kg BW/d}) = \frac{\sum_{k}^{m} (C_k \times FR_k \times IR_k)}{BW}$$

[2]Hazard quotients were calculated according to the following equation:

$$HQ = \frac{ADD_{pot}\ (\text{mg/kg BW/d})}{\text{Toxicity Reference Value}\ (\text{mg/kg BW/d})}$$

Table A-3. Exposure profile and risk characterization for the white-footed mouse (using assumptions of CDM/MDEQ).

HQ calculations for robins using CDM/MDEQ exposure assumptions and modeled concentrations of PCBs in terrestrial plants											
		Dietary fraction	Dietary item PCB conc.	Total dietary PCB conc.	IR	BW	APDD[1]	Toxicity Reference Value		Hazard Quotient[2]	
Receptor	Dietary item	(DF)	(mg/kg)	(mg/kg FW diet)	(g/d)	(g)	(mg/kg/d)	NOAEL	LOAEL	NOAEL-based	LOAEL-based
WF-mouse	Terr. plants	0.44	4.7	2.068	5.5	21	0.54				
	Terr. inverts	0.56	1.5	0.84	5.5	21	0.22				
	Total						0.76	0.5	6.5	1.5	0.12

HQ calculations for robins using CDM/MDEQ exposure assumptions* and measured concentrations of PCBs in terrestrial plants**											
		Dietary fraction	Dietary item PCB conc.	Total dietary PCB conc.	IR	BW	APDD[1]	Toxicity Reference Value		Hazard Quotient[2]	
Receptor	Dietary item	(DF)	(mg/kg)	(mg/kg FW diet)	(g/d)	(g)	(mg/kg/d)	NOAEL	LOAEL	NOAEL-based	LOAEL-based
WF-mouse	Terr. plants	0.44	0.033	0.01	5.5	21	0.00				
	Terr. inverts	0.56	1.5	0.84	5.5	21	0.22				
	Total						0.22	0.5	6.5	0.45	0.03

*CDM/MDEQ exposure assumptions are from the report entitled, "Final Baseline Ecological Risk Assessment - Allied Paper Inc./Portage Creek/Kalamazoo River Superfund Site - June 1999"

**Measured concentrations of PCBs in terrestrial plants are from Michigan State University's Aquatic Toxicology Laboratory (MSU/ATL) from samples collected in 2000.

[1]Average potential daily doses (APDD) were calculated according to the following equation:

$$ADD_{pot}\ (\text{mg/kg BW/d}) = \frac{\sum_{k}^{m} (C_k \times FR_k \times IR_k)}{BW}$$

[2]Hazard quotients were calculated according to the following equation:

$$HQ = \frac{ADD_{pot}\ (\text{mg/kg BW/d})}{\text{Toxicity Reference Value}\ (\text{mg/kg BW/d})}$$

Appendix B

Supporting Information for Risk Calculations Using Interim ERA Assumptions

Data sets on impoundment soils, earthworms, white-footed mice, waterfowl, and muskrats with descriptive statistics that are used in the interim ERA.

Table B-1. Summary of Former Impoundment Floodplain Soil Data with Different Methods of Treating Non-dectable PCB Concentrations.*

Summary of Former Impoundment PCB Data (mg/kg)						Non-detects** (ND) removed from statistical analysis					
	sample size	mean	minimum	median	maximum	geometric mean	standard deviation	95% UCL interval	95% UCL (mean)	95% UCL (geometric mean)	mean percent organic carbon
Plainwell Dam	32	13.332	0.230	5.600	85.000	5.544	18.377	6.367	19.699	9.264	8.869
Otsego Dam	29	11.664	0.048	8.000	36.000	5.215	11.023	4.012	15.676	9.787	8.501
Trowbridge Dam	57	15.735	0.026	11.000	81.100	6.510	16.306	4.233	19.969	10.438	7.645

Summary of Former Impoundment PCB Data (mg/kg)						ND= 0					
	sample size	mean	minimum	median	maximum	geometric mean	standard deviation	95% UCL interval	95% UCL (mean)	95% UCL (geometric mean)	mean percent organic carbon
Plainwell Dam	42	10.158	0.000	3.175	85.000	0.712	16.981	5.136	15.293	2.339	8.869
Otsego Dam	41	8.250	0.000	2.260	36.000	0.426	10.673	3.267	11.517	1.540	8.501
Trowbridge Dam	76	11.802	0.000	6.547	81.100	0.725	15.671	3.523	15.325	1.837	7.645

Summary of Former Impoundment PCB Data (mg/kg)						ND=1/2 Detection Limit					
	sample size	mean	minimum	median	maximum	geometric mean	standard deviation	95% UCL interval	95% UCL (mean)	95% UCL (geometric mean)	mean percent organic carbon
Plainwell Dam	42	10.165	0.029	3.175	85.000	1.597	16.977	5.134	15.299	3.502	8.869
Otsego Dam	41	8.261	0.028	2.260	36.000	1.196	10.665	3.264	11.525	2.765	8.501
Trowbridge Dam	76	11.816	0.014	6.547	81.100	1.811	15.660	3.521	15.337	3.363	7.645

Summary of Former Impoundment PCB Data (mg/kg)						ND= Detection Limit					
	sample size	mean	minimum	median	maximum	geometric mean	standard deviation	95% UCL interval	95% UCL (mean)	95% UCL (geometric mean)	mean percent organic carbon
Plainwell Dam	42	10.172	0.057	3.175	85.000	1.884	16.973	5.133	15.305	3.824	8.869
Otsego Dam	41	8.272	0.048	2.260	36.000	1.464	10.656	3.262	11.534	3.124	8.501
Trowbridge Dam	76	11.832	0.026	6.547	81.100	2.158	15.648	3.518	15.350	3.796	7.645

*Data calculated from original data in BBL's Draft Technical Memorandum 12 (1994).

**The number of non-detects for each impoundment are as follows:

Plainwell Dam (10 non-detects)
Otsego Dam (12 non-detects)
Trowbridge Dam (20 non-detects)

Table B-2. Summary of Earthworm Data with Different Methods of Treating Non-Detectable PCB Concentrations.*

	Summary of Earthworm PCB Data (mg/kg)					Non-detects** (ND) removed from statistical analysis					
	sample size	mean	minimum	median	maximum	geometric mean	standard deviation	95% UCL interval	95% UCL (mean)	95% UCL (geometric mean)	mean percent lipid
Downstream of Allegan Dam (TBSA 1)	3	0.024	0.023	0.024	0.025	0.024	0.001	0.001	0.025	0.025	1.297
Trowbridge Dam (TBSA 3&5)	6	2.257	1.310	2.200	3.200	2.182	0.611	0.489	2.746	2.757	1.890
Plainwell Dam (TBSA 10)	-	-	-	-	-	-	-	-	-	-	1.590
Upstream reference (TBSA 11)	3	0.460	0.13	0.590	0.660	0.370	0.288	0.326	0.786	1.033	2.070

	Summary of Earthworm PCB Data (mg/kg)					ND= 0.001					
	sample size	mean	minimum	median	maximum	geometric mean	standard deviation	95% UCL interval	95% UCL (mean)	95% UCL (geometric mean)	mean percent lipid
Downstream of Allegan Dam (TBSA 1)	3	0.024	0.023	0.024	0.025	0.024	0.001	0.001	0.025	0.025	1.297
Trowbridge Dam (TBSA 3&5)	6	2.257	1.310	2.200	3.200	2.182	0.611	0.489	2.746	2.757	1.890
Plainwell Dam (TBSA 10)	3	0.001	0.001	0.001	0.001	0.001	0.000	-	-	-	1.590
Upstream reference (TBSA 11)	3	0.460	0.130	0.590	0.660	0.370	0.288	0.326	0.786	1.033	2.070

	Summary of Earthworm PCB Data (mg/kg)					ND=1/2 Detection Limit					
	sample size	mean	minimum	median	maximum	geometric mean	standard deviation	95% UCL interval	95% UCL (mean)	95% UCL (geometric mean)	mean percent lipid
Downstream of Allegan Dam (TBSA 1)	3	0.024	0.023	0.024	0.025	0.024	0.001	0.001	0.025	0.025	1.297
Trowbridge Dam (TBSA 3&5)	6	2.257	1.310	2.200	3.200	2.182	0.611	0.489	2.746	2.757	1.890
Plainwell Dam (TBSA 10)	3	0.027	0.025	0.025	0.030	0.026	0.003	0.003	0.029	0.029	1.590
Upstream reference (TBSA 11)	3	0.460	0.130	0.590	0.660	0.370	0.288	0.326	0.786	1.033	2.070

	Summary of Earthworm PCB Data (mg/kg)					ND= Detection Limit					
	sample size	mean	minimum	median	maximum	geometric mean	standard deviation	95% UCL interval	95% UCL (mean)	95% UCL (geometric mean)	mean percent lipid
Downstream of Allegan Dam (TBSA 1)	3	0.024	0.023	0.024	0.025	0.024	0.001	0.001	0.025	0.025	1.297
Trowbridge Dam (TBSA 3&5)	6	2.257	1.310	2.200	3.200	2.182	0.611	0.489	2.746	2.757	1.890
Plainwell Dam (TBSA 10)	3	0.053	0.050	0.050	0.059	0.053	0.005	0.006	0.059	0.059	1.590
Upstream reference (TBSA 11)	3	0.460	0.130	0.590	0.660	0.370	0.288	0.326	0.786	1.033	2.070

*Data calculated from original data in BBL's Draft Technical Memorandum 14 - Biota Investigation (1994).

**The number of non-detects for each impoundment are as follows:

Downstream of Allegan Dam (TBSA 1) (0 non-detects)
Trowbridge Dam (TBSA 3&5) (0 non-detects)
Plainwell Dam (TBSA 10) (3 non-detects)
Upstream reference (TBSA 11) (0 non-detects)

Table B-3. Summary of Whitefooted Mouse Data with Different Methods of Treating Non-Detectable PCB Concentrations.*

Summary of Whitefooted Mouse PCB Data (mg/kg)						Non-detects** (ND) removed from statistical analysis					
	sample size	mean	minimum	median	maximum	geometric mean	standard deviation	95% UCL interval	95% UCL (mean)	95% UCL (geometric mean)	mean percent lipid
Downstream of Allegan Dam (TBSA 1)	8	0.078	0.011	0.036	0.350	0.043	0.112	0.078	0.155	0.090	2.365
Trowbridge Dam (TBSA 3&5)	18	0.209	0.017	0.195	0.450	0.164	0.124	0.057	0.266	0.240	1.266
Plainwell Dam (TBSA 10)	7	0.119	0.057	0.110	0.280	0.103	0.077	0.057	0.176	0.155	1.098
Upstream reference (TBSA 11)	-	-	-	-	-	-	-	-	-	-	1.286

Summary of Whitefooted Mouse PCB Data (mg/kg)						ND= 0.001					
	sample size	mean	minimum	median	maximum	geometric mean	standard deviation	95% UCL interval	95% UCL (mean)	95% UCL (geometric mean)	mean percent lipid
Downstream of Allegan Dam (TBSA 1)	10	0.062	0.001	0.028	0.350	0.020	0.104	0.065	0.127	0.064	2.365
Trowbridge Dam (TBSA 3&5)	20	0.188	0.001	0.160	0.450	0.099	0.134	0.059	0.247	0.212	1.266
Plainwell Dam (TBSA 10)	10	0.083	0.001	0.067	0.280	0.026	0.085	0.053	0.136	0.105	1.098
Upstream reference (TBSA 11)	10	0.001	0.001	0.001	0.001	0.001	-	-	-	-	1.286

Summary of Whitefooted Mouse PCB Data (mg/kg)						ND=1/2 Detection Limit					
	sample size	mean	minimum	median	maximum	geometric mean	standard deviation	95% UCL interval	95% UCL (mean)	95% UCL (geometric mean)	mean percent lipid
Downstream of Allegan Dam (TBSA 1)	10	0.067	0.011	0.029	0.350	0.039	0.101	0.063	0.130	0.071	2.365
Trowbridge Dam (TBSA 3&5)	20	0.192	0.017	0.160	0.450	0.142	0.129	0.056	0.248	0.210	1.266
Plainwell Dam (TBSA 10)	10	0.092	0.029	0.067	0.280	0.072	0.076	0.047	0.139	0.113	1.098
Upstream reference (TBSA 11)	10	0.039	0.025	0.038	0.055	0.038	0.011	0.007	0.046	0.045	1.286

Summary of Whitefooted Mouse PCB Data (mg/kg)						ND= Detection Limit					
	sample size	mean	minimum	median	maximum	geometric mean	standard deviation	95% UCL interval	95% UCL (mean)	95% UCL (geometric mean)	mean percent lipid
Downstream of Allegan Dam (TBSA 1)	10	0.073	0.011	0.045	0.350	0.045	0.100	0.062	0.134	0.080	2.365
Trowbridge Dam (TBSA 3&5)	20	0.196	0.017	0.160	0.450	0.152	0.124	0.055	0.250	0.217	1.266
Plainwell Dam (TBSA 10)	10	0.102	0.057	0.069	0.280	0.088	0.069	0.042	0.144	0.121	1.098
Upstream reference (TBSA 11)	10	0.078	0.050	0.076	0.110	0.075	0.022	0.013	0.091	0.089	1.286

*Data calculated from original data in BBL's Draft Technical Memorandum 14 - Biota Investigation (1994).

**The number of non-detects for each impoundment are as follows:

Downstream of Allegan Dam (TBSA 1) (2 non-detects)
Trowbridge Dam (TBSA 3&5) (2 non-detects)
Plainwell Dam (TBSA 10) (3 non-detects)
Upstream reference (TBSA 11) (10 non-detects)

Table B-4. Summary of Waterfowl Data with Different Methods of Treating Non-Detectable PCB Concentrations.*

	Summary of Waterfowl PCB Data (mg/kg)					Non-detects** (ND) removed from statistical analysis					
	sample size	mean	minimum	median	maximum	geometric mean	standard deviation	95% UCL interval	95% UCL (mean)	95% UCL (geometric mean)	mean percent lipid
Sitewide (all of KRAOC)	14	3.34	0.29	1.27	28.0	1.42	7.18	3.76	7.10	2.54	6.21

	Summary of Waterfowl PCB Data (mg/kg)					ND= 0.001					
	sample size	mean	minimum	median	maximum	geometric mean	standard deviation	95% UCL interval	95% UCL (mean)	95% UCL (geometric mean)	mean percent lipid
Sitewide (all of KRAOC)	25	1.88	0.01	0.55	28.0	0.16	5.55	2.18	4.05	0.45	6.21

	Summary of Waterfowl PCB Data (mg/kg)					ND=1/2 Detection Limit					
	sample size	mean	minimum	median	maximum	geometric mean	standard deviation	95% UCL interval	95% UCL (mean)	95% UCL (geometric mean)	mean percent lipid
Sitewide (all of KRAOC)	25	1.91	0.03	0.55	28.0	0.35	5.54	2.17	4.08	0.74	6.21

	Summary of Waterfowl PCB Data (mg/kg)					ND= Detection Limit					
	sample size	mean	minimum	median	maximum	geometric mean	standard deviation	95% UCL interval	95% UCL (mean)	95% UCL (geometric mean)	mean percent lipid
Sitewide (all of KRAOC)	25	1.94	0.05	0.55	28.0	0.48	5.53	2.17	4.11	0.89	6.21

*Data calculated from original data in USFWS data sheets

**There were 11 samples with non-detectable concentrations of PCBs for the waterfowl data.

Table B-5. Summary of Muskrat Data*

	Concentrations of PCBs in Muskrat Carcasses (mg/kg)									
	sample size	mean	minimum	median	maximum	geometric mean	standard deviation	95% UCL interval	95% UCL (mean)	95% UCL (geometric mean)
Plainwell (ABSA 5)	6	1.00	0.08	1.01	2.00	0.68	0.71	0.57	1.57	1.76
Otsego Dam (ABSA 7)	6	0.25	0.14	0.20	0.45	0.23	0.13	0.10	0.35	0.60
Trowbridge (ABSA 8)	6	2.13	0.16	0.58	8.40	0.80	3.23	2.58	4.71	1.40
Allegan (ABSA 9)	6	1.61	0.17	1.90	3.10	1.03	1.16	0.93	2.54	6.39
Reference (ABSA 1)	6	0.15	0.12	0.15	0.18	0.15	0.02	0.02	0.17	0.69

*Data calculated from original data in CDM data sheets

Table B-6. Input Data for the Interim ERA.

Interim ERA input data will be revised as additional site-specific data become available.

Sample type	Upstream Reference ABSA 1 includes TBSA 11		Fmr. Plainwell Impoundment ABSA 5 includes TBSA 10		Fmr. Otsego Impoundment ABSA 7		Fmr. Trowbridge Impoundment ABSA 8 includes TBSA 3 & 5	
	geomean	95% UCL (geomean)	geomean	95% UCL (geomean)	geomean	95% UCL (geomean)	geomean	95% UCL (geomean)
surface water[1]	0.000043	0.000043	0.000043	0.000043	0.000043	0.000043	0.000043	0.000043
soil[2]	0.39	0.39	1.60	3.5	1.18	2.77	1.81	3.36
muskrat[3]	0.15	0.15	0.68	1.25	0.36	0.61	0.49	1.46
white-footed mouse[4]	0.038	0.045	0.072	0.11	0.24	0.30	0.14	0.21
earthworm (point values)[4]	0.024			0.026			2.18	
earthworm (impoundment-wide estimate)[5]	0.06	0.06	0.24	0.53	0.18	0.42	0.27	0.50
waterfowl[6]	1.42	2.54	1.42	2.54	1.42	2.54	1.42	2.54
plants (point values)[7]	0.0030	0.0051	0.0079	0.0231	0.0079	0.0231	0.0079	0.0231

Note - units for mice, muskrats, earthworms, and plants are mg/kg PCBs (wet weight basis); units for soil are mg/kg (dry weight basis); units for water are mg/L.

[1]Data from MDEQ, 1999 (95% UCL of arithmetic mean used as a worst-case for all potential exposures).

[2]Data calculated from original data in BBL's Draft Technical Memorandum 12 (1994) using one-half of the detection limit for samples with non-detectable concentrations of PCBs.

[3]Data from CDM's Final Technical Memorandum Mink/Muskrat Biota Sampling (1997).

[4]Data calculated from BBL's Draft Technical Memorandum 14 - Biota Investigation (1994).

[5]Predicted earthworm concentrations were calculated by multiplying the exposure point concentration in soil times the sitewide maximum site-specific bioaccumulation factor (soil to earthworm BAF; BBL's Draft Technical Memorandum 14 - Biota Investigation, 1994;). BAFs were used to estimate impoundment-wide concentrations in earthworms because data are not available for all locations within each impoundment.

[6]Data from USFWS

[7]Data from MSU-ATL studies (2000). Data from Trowbridge were used for other impoundments since data were not collected from other impoundments.

Appendix C

Supporting Information for Risk Calculations Using Interim ERA Assumptions

Exposure profiles and risk characterization for the American robin, short-tailed shrew, and great horned owls are provided in the following tables.

Table C-1. Exposure profile and risk characterization for American robin

Interim ERA (will be revised when additional site-specific data become available)

	Upstream Reference ABSA 1 includes TBSA 11		Fmr. Plainwell Impoundment ABSA 5 includes TBSA 10		Fmr. Otsego Impoundment ABSA 7		Fmr. Trowbridge Impoundment ABSA 8 includes TBSA 3 & 5			
	geomean	95% UCL (geomean)	geomean	95% UCL (geomean)	geomean	95% UCL (geomean)	geomean	95% UCL (geomean)	max	sitewide
prey item	**Concentrations in prey (mg PCBs/kg, wet weight)**									
surface water (95% UCL, from MDEQ, 1999)	0.000043	0.000043	0.000043	0.000043	0.000043	0.000043	0.000043	0.000043		
plants (MSU-ATL 2000 data)	0.0030	0.0051	0.0079	0.0231	0.0079	0.0231	0.0079	0.0231		
soil (ABSA-wide concentration)	0.39	0.39	1.6	3.5	1.18	2.77	1.81	3.36		
predicted earthworms (using max site-spec. BAF)	0.06	0.06	0.24	0.53	0.18	0.42	0.27	0.50		
exposure scenarios (including incidental soil ingestion at 2% of invertebrate ingestion rate)	**Potential average daily dose (ADD_{pot}; mg PCBs/kg body weight/day)**									
1:1 inverts:plants	0.03	0.03	0.11	0.24	0.08	0.19	0.12	0.23		
1:1 inverts:plants (50% of diet from Kalamazoo R)	0.01	0.01	0.05	0.12	0.04	0.09	0.06	0.11		
scenarios	**Hazard Quotients**									
*1:1 inverts:plants (NOAEL)	0.04	0.05	0.18	0.39	0.13	0.32	0.20	0.38	0.39	0.21
*1:1 inverts:plants (LOAEL)	0.01	0.02	0.06	0.13	0.04	0.11	0.07	0.13	0.13	0.07
**1:1 inverts:plants (50% of diet from Kalamazoo R; NOAEL)	0.02	0.02	0.09	0.20	0.07	0.16	0.10	0.19	0.20	0.11
**1:1 inverts:plants (50% of diet from Kalamazoo R; LOAEL)	0.01	0.01	0.03	0.07	0.02	0.05	0.03	0.06	0.07	0.04

Notes:

Potential average daily dose (ADDpot; mg PCBs/kg body weight/day) was calculated according to the equation (see text for details):

$$ADD_{pot}\ (mg / kg\ BW / d) = \sum_{k}^{m} (C_k \times FR_k \times NIR_k)$$

Intake rates differ for invertebrates versus plants in diet (USEPA, 1993)
average consumption rate of plants in diet of robins is 1.2 g/g/d (USEPA, 1993)
consumption rate of invertebrates in diet of robins is calculated from the plant rate modified for differences in both assimilation efficiency and energy content of food = 0.76 g/g/d

Hazard quotients (HQ) were calculated according to the following equation:

$$HQ = \frac{ADD_{pot}\ (mg / kg\ BW / d)}{\text{Toxicity Reference Value}\ (mg / kg\ BW / d)}$$

The study TRV for robins is from Dahlgren et al., (1972) as a LOAEL in pheasants = 1.8 mg/kg/d (for Aroclor 1254) - see GLWQI.

The study TRV for robins is derived from Dahlgren et al., (1972) as a NOAEL in pheasants = 0.6 mg/kg/d (for Aroclor 1254).
LOAEL to NOAEL correction of 0.3 (a factor of 0.1 would be excessively conservative because for the chicken, which is the most sensitive bird species tested, the NOAEL is 0.98 mg/kg/d for the same endpoint). Thus, it is unlikely that the chronic NOAEL for other bird species is any lower than for chickens.
Despite this and to maintain conservatism, the calculated NOAEL used in these calculations is lower than the chicken NOEL.

Earthworm concentrations were predicted using two independent methods: 1) maximum site-specific [using field measured bioaccumulation factors (BAFs); and 2) a literature-based relationship between soil (OC-normalized) and earthworms (lipid-normalized). However, the exposure calculations and HQs are based on the site-specific data.

Note that the diet of the robin has been simplified to only earthworms, the actual diet of robins is likely to include beetles, caterpillars, crickets, etc. that may have significantly different concentrations of PCBs

* Denotes the exposure scenarios that were utilized to support presentation to MDEQ on 3/25/99.
**The most likely exposure scenario based on available life history data for this species and site-specific information on the existing habitats and biological communities associated with Kalamazoo River.

Table C-2. Exposure profile and risk characterization for short-tailed shrews

Interim ERA (will be revised when site-specific data become available)

	Upstream Reference ABSA 1 includes TBSA 11		Fmr. Plainwell Impoundment ABSA 5 includes TBSA 10		Fmr. Otsego Impoundment ABSA 7		Fmr. Trowbridge Impoundment ABSA 8 includes TBSA 3 & 5		
	geomean	95% UCL (geomean)	geomean	95% UCL (geomean)	geomean	95% UCL (geomean)	geomean	95% UCL (geomean)	max
Dietary item	**Concentrations in prey (mg PCBs/kg, wet weight)**								
soil	0.39	0.39	1.6	3.5	1.18	2.77	1.81	3.36	
predicted earthworm conc. (impoundment-wide)[1]	0.06	0.06	0.24	0.53	0.18	0.42	0.27	0.50	
plants (MSU-ATL data)	0.003	0.0051	0.0079	0.0231	0.0079	0.0231	0.0079	0.0231	
crickets, etc. (need to enter) assume equal to pred. earthworm[2]	0.06	0.06	0.24	0.53	0.18	0.42	0.27	0.50	
	Potential average daily dose (ADD_{pot}; mg PCBs/kg body weight/day)[3]								
exposure scenarios (including 2% incidental soil ingestion)									
90% (predicted) invertebrates, 10% plants, incidental soil ingestion	0.04	0.04	0.16	0.34	0.12	0.27	0.18	0.33	0.34
	Hazard Quotients[4]								
dietary-based hazard quotients									
90% invertebrates and incidental soil ingestion (NOEL = 0.69)[5]	0.06	0.1	0.2	0.5	0.2	0.4	0.3	0.5	0.50
90% invertebrates and incidental soil ingestion (LOEL=3.2)[5]	0.01	0.01	0.05	0.11	0.04	0.08	0.06	0.10	0.11

[1]Predicted earthworm concentrations were calculated by multiplying the exposure point concentration in soil times the sitewide maximum site-specific bioaccumulation factor (soil to earthworm BAF; BBL's Draft Technical Memorandum 14 - Biota Investigation, 1994;). BAFs were used to estimate impoundment-wide concentrations in earthworms because data are not available for all locations within each impoundment.

[2]Note that data are not yet available for all site-specific prey items in the diet of the shrew. Diets of shrews are greatly variable (see text for details). At this point, it is assumed that all invertebrate prey items have concentrations of PCBs that are equivalent to earthworms. As data become available for additional dietary items, this data will be incorporated into the exposure calculations. Plant data comes from plants collected during 2000 and analyzed by MSU.

[3]Potential average daily dose (ADDpot; mg PCBs/kg body weight/day) was calculated according to the equation (see text for details):

$$ADD_{pot}\,(mg/kg\,BW/d) = \sum_{k}^{m} (C_k \times FR_k \times NIR_k)$$

Normalized ingestion rate (NIR) for shrews taken as the weighted average of all data from USEPA (1993), which is 0.629 g/g/d.

[4]Hazard quotients (HQ) were calculated according to the following equation:

$$HQ = \frac{ADD_{pot}\,(mg/kg\,BW/d)}{\text{Toxicity Reference Value}\,(mg/kg\,BW/d)}$$

[5]TRV for shrews is from Linder et al., 1974, as a NOEL (in rats = 0.32 mg/kg/d for Aroclor 1254; very sensitive 2-generational study) and LOEL = 1.5 mg/kg/d TRV allometrically adjusted to account for differences in body weight of shrews and rats. The shrew NOAEL and LOAEL are 0.69 and 3.2 mg/kg/d, respectively.

Table C-3. Exposure profile and risk characterization for great horned owl

Interim ERA (will be revised when site-specific data become available)

	Upstream Reference ABSA 1 includes TBSA 11		Fmr. Plainwell Impoundment ABSA 5 includes TBSA 10		Fmr. Otsego Impoundment ABSA 7		Fmr. Trowbridge Impoundment ABSA 8 includes TBSA 3 & 5			
	geomean	95% UCL (geomean)	geomean	95% UCL (geomean)	geomean	95% UCL (geomean)	geomean	95% UCL (geomean)	max	sitewide
prey item	**Concentrations in prey (mg PCBs/kg, wet weight)**									
surface water (95% UCL, from MDEQ, 1999)	0.000043	0.000043	0.000043	0.000043	0.000043	0.000043	0.000043	0.000043		
soil (ABSA-wide concentration)	0.39	0.39	1.6	3.5	1.18	2.77	1.81	3.36		
waterfowl (blanks equal max)	1.42	2.54	1.42	2.54	1.42	2.54	1.42	2.54		
muskrats	0.15	0.15	0.68	1.25	0.36	0.61	0.49	1.46		
mice (blanks equal max)	0.038	0.045	0.072	0.11	0.24	0.3	0.14	0.21		
exposure scenarios (including soil and water ingestion)	**Potential average daily dose (ADD_{pot}; mg PCBs/kg body weight/day)**									
100% waterfowl	0.01	0.03	0.01	0.03	0.01	0.03	0.01	0.03		
100% small mammals (1:1 mice:muskrats)	0.01	0.01	0.04	0.07	0.03	0.05	0.03	0.08		
9:1 mammals:waterfowl	0.02	0.03	0.05	0.09	0.04	0.07	0.04	0.10		
9:1 mammals:waterfowl (50% of diet from Kalamazoo R)	0.01	0.02	0.03	0.05	0.02	0.04	0.02	0.05		
scenarios	**Hazard Quotients**									
100% waterfowl (NOAEL)	0.03	0.06	0.03	0.06	0.03	0.06	0.03	0.06	0.06	0.05
100% small mammals (NOAEL)	0.02	0.02	0.09	0.17	0.07	0.11	0.08	0.20	0.20	0.10
9:1 mammals:waterfowl (NOAEL)	0.06	0.08	0.12	0.22	0.10	0.17	0.11	0.25	0.25	0.14
9:1 mammals:waterfowl (estimated LOAEL)	0.02	0.03	0.04	0.07	0.03	0.06	0.04	0.08	0.08	0.05
*9:1 mammals:waterfowl (50% of diet from Kalamazoo R; NOAEL)	0.03	0.04	0.06	0.11	0.05	0.09	0.05	0.13	0.13	0.07
*9:1 mammals:waterfowl (50% of diet from Kalamazoo R; LOAEL)	0.01	0.01	0.02	0.04	0.02	0.03	0.02	0.04	0.04	0.02

Notes:

Potential average daily dose (ADDpot; mg PCBs/kg body weight/day) was calculated according to the equation (see text for details):

$$ADD_{pot}\ (mg / kg\ BW / d) = \sum_{k}^{m} (C_k \times FR_k \times NIR_k)$$

Hazard quotients (HQ) were calculated according to the following equation:

$$HQ = \frac{ADD_{pot}\ (mg / kg\ BW / d)}{\text{Toxicity Reference Value}\ (mg / kg\ BW / d)}$$

The study TRV for great horned owls is from McLane and Hughes, 1980 as a chronic NOAEL =0.41 mg/kg/d (for Aroclor 1254) in screech owls (reproductive endpoints).
The study LOAEL was estimated by multiplying the NOAEL by an uncertainty factor of three (e.g., NOAEL x 3 = 1.23 mg/kg/d).

"blanks equals max" refers to a conservative approach used in instances when data are unavailable, the maximum sitewide ABSA geometric mean and 95% UCL values were used

* The most likely exposure scenario based on the available life history data for this species and site-specific information on the existing habitats and biological communities associated with Kalamazoo River.

ATTACHMENT 2:

PRESENTATION OF SHREW RISK EVALAUATION (BLANKENSHIP ET AL., 2002)

Accumulation of PCBs and an Evaluation of Ecological Risk for Shrews and Other Small Mammals at the Kalamazoo River Superfund Site, Michigan

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Outline

- Objectives
- Site map
- Conceptual site model - food web
- Foodweb samples with PCB data
- Congener patterns
- Risk characterization for shrews
- Conclusions

Specific Study Objectives

Focus on wildlife that are expected to have the greatest exposures and/or sensitivity to PCBs such as:

- mink and bald eagle as representative receptors for aquatic-based exposure pathways
- **shrews** and great horned owls as representative receptors for terrestrial-based exposure pathways

Specific Study Objectives (continued)

Evaluate risk of adverse health effects in wildlife that are exposed to PCBs by:

- Determining site-specific dietary exposure for wildlife
  - Dietary composition and PCB content of each dietary item
  - Compare exposure to dietary effect levels (TRVs)
- Determining PCB concentrations in tissues of target species
  - Comparison to tissue residue effect levels
- Population levels/Abundance/Productivity

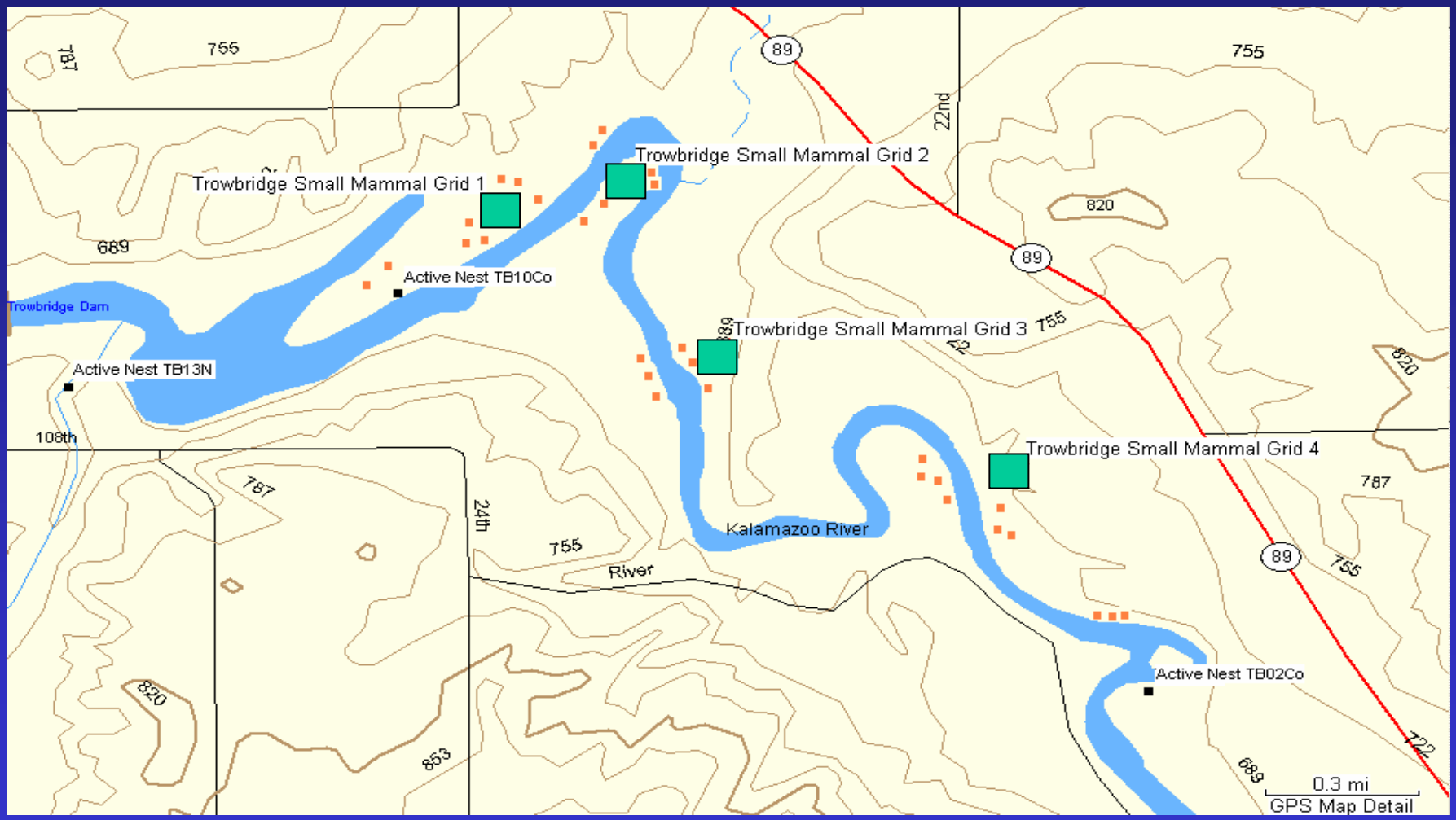
Additional Objectives

- Bioaccumulation patterns of individual congeners and total PCBs
  - Data from other studies of the Kalamazoo River reported Aroclor-equivalent concentrations; however the degree of environmental weathering is unknown
  - Few studies have evaluated PCB congener uptake in terrestrial systems

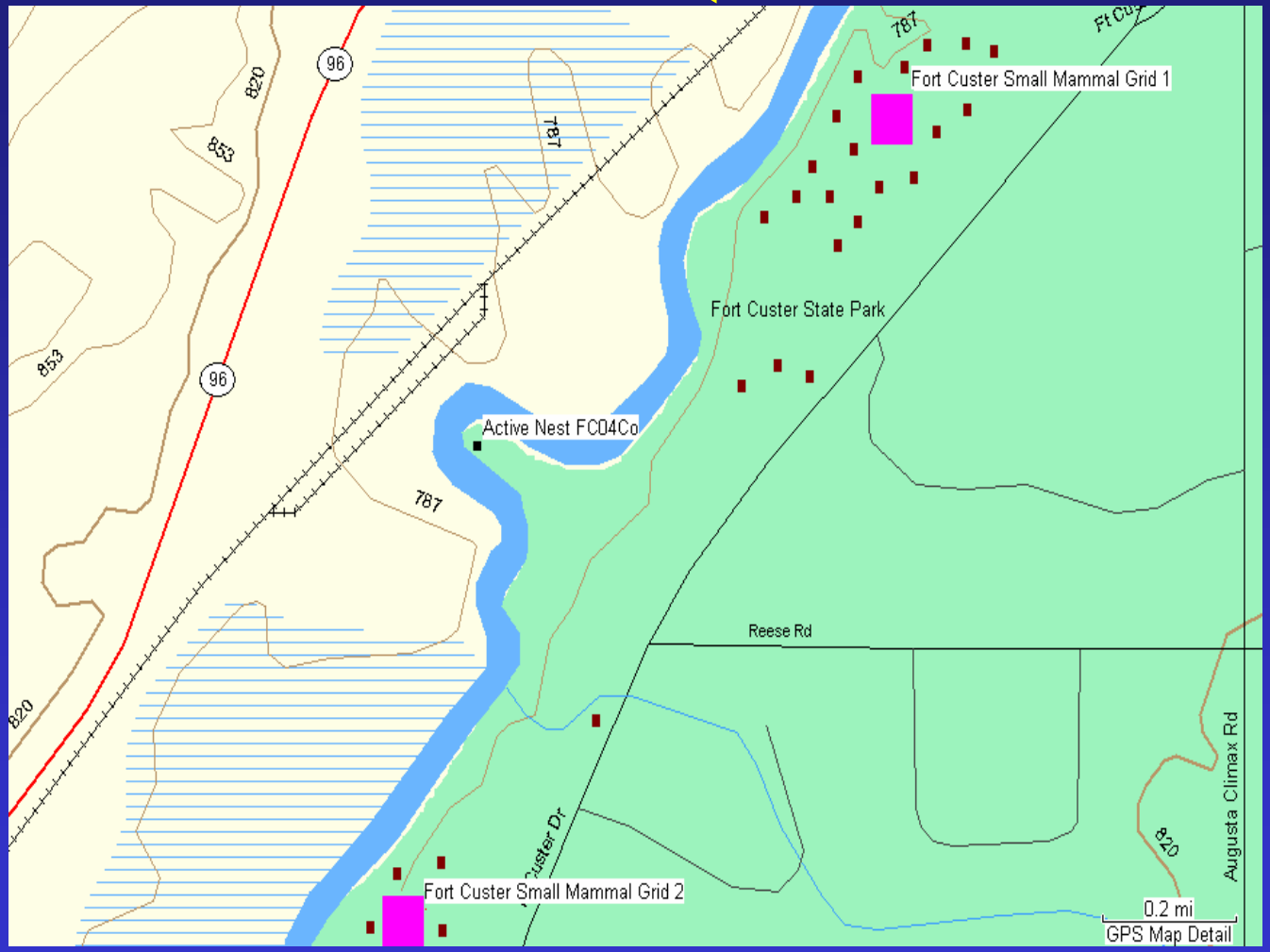
Specific Study Components and Rationale

- Focus on two locations for food web analysis
  - Former Trowbridge impoundment as a worst-case and selected because:
    - it is the largest of the impoundments (~ 327 acres of former sediments and ~172 acres of existing impounded water)
    - has the greatest mass of PCBs
    - has the greatest surficial mean concentration of PCBs in soils (~11 mg/kg, dw)
  - Fort Custer Recreation Area as a reference
    - selected because it is upstream of KRAOC and relatively uncontaminated with PCBs

Site Map - Trowbridge Grids



Site Map - Fort Custer (Reference Area) Grids



- Reference area ~30 miles upstream of Trowbridge

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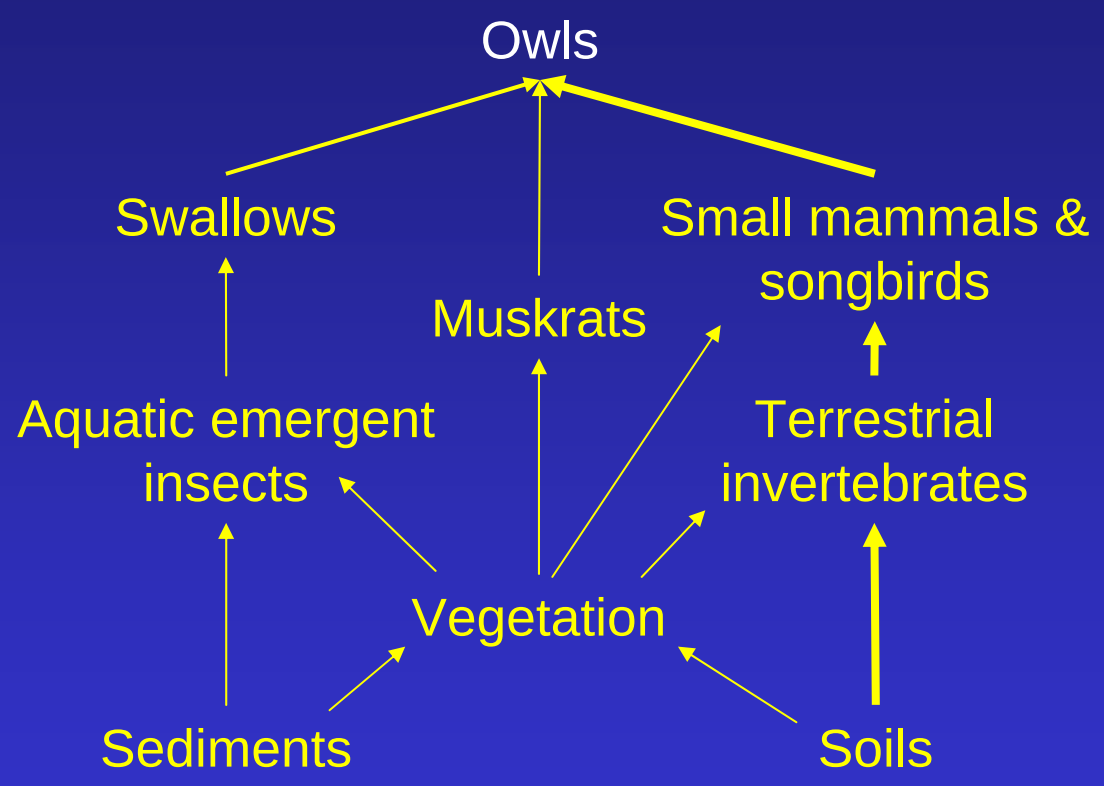
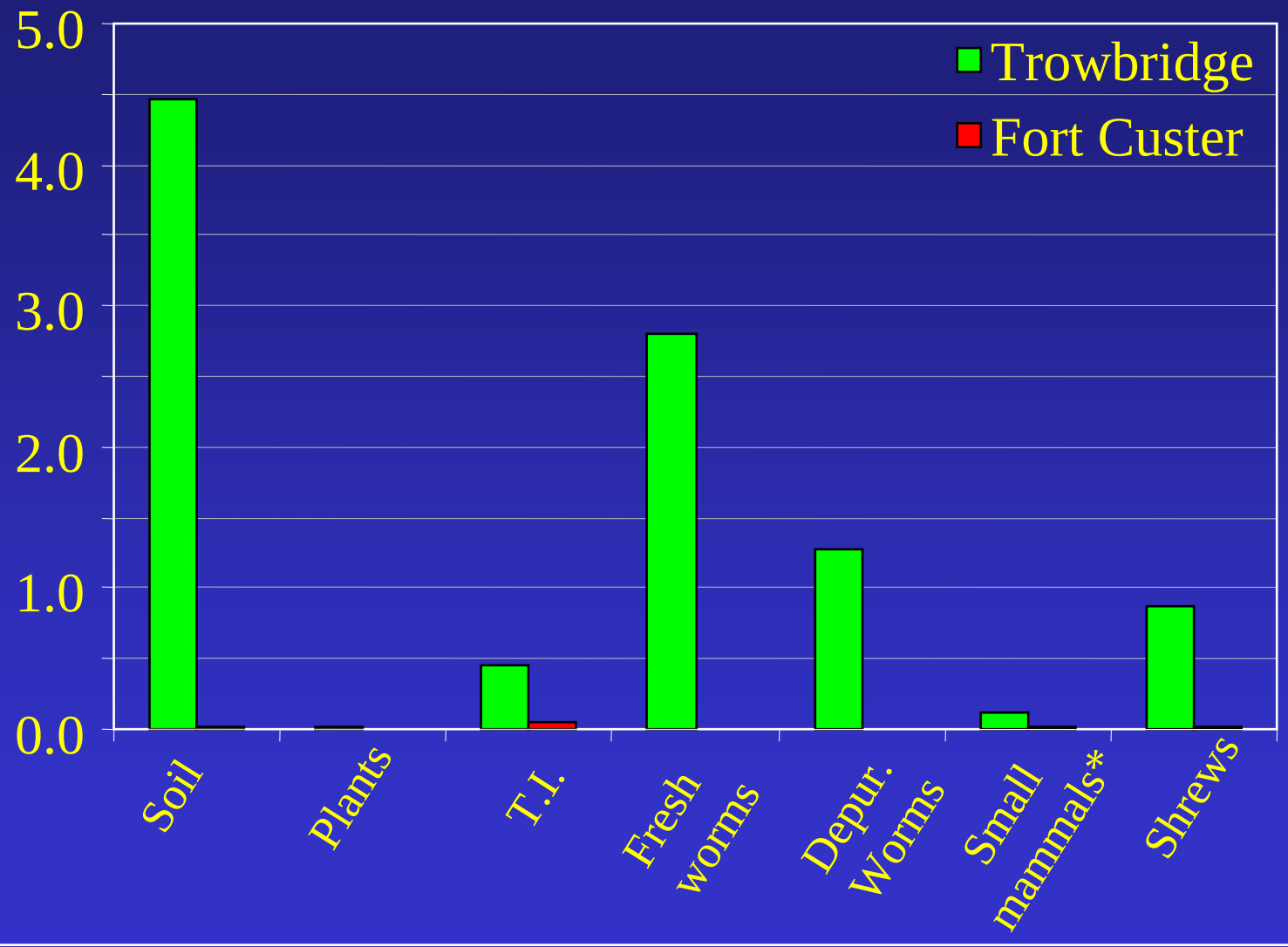
Terrestrial Food Web Samples Collected for PCB Analysis

- 6 grids (4 at Trowbridge & 2 at Reference area)
- Up to 4 different sampling rounds
- 28 soil samples
- 36 plant samples (terrestrial)
- > 150 groups of terrestrial invertebrates (fresh earthworms, depurated earthworms, crickets, beetles)
- 154 small mammals (mice, moles, voles and shrews)
- Passerine bolus samples, eggs, and nestlings
- Great horned owl serum

Preliminary Exposure Pathways



PCB Concentrations in Food Web



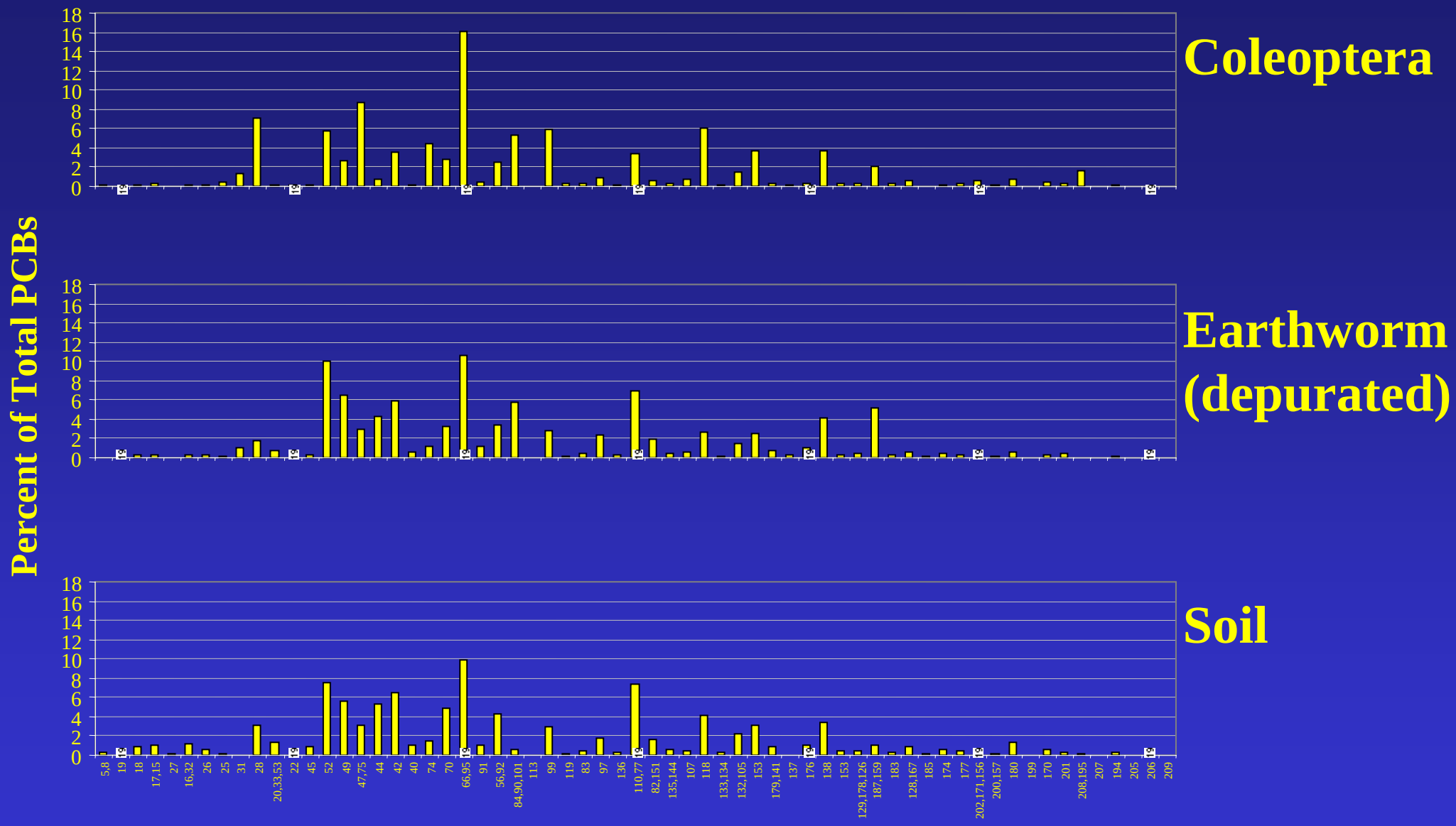
Key Findings - Total PCBs in Food Web

- All plant, invertebrate, and small mammal samples collected from grids had mean concentrations of PCBs that were less than that of the co-located soil
- Depurated earthworms generally had less PCBs than fresh earthworms (exceptions occurred at low PCB concentrations)
- Shrews had approximately 10-fold greater concentrations of PCBs than other small mammals (deer mice, meadow voles, etc.)

PCB Congener-Specific Patterns

- Few studies have attempted to evaluate PCB congener patterns in terrestrial food webs
- Are there differences in patterns among lower trophic levels?

- The next three slides depict relative congener patterns for samples selected from a single sampling grid at Trowbridge
- Concentrations of individual congeners were normalized to total PCBs



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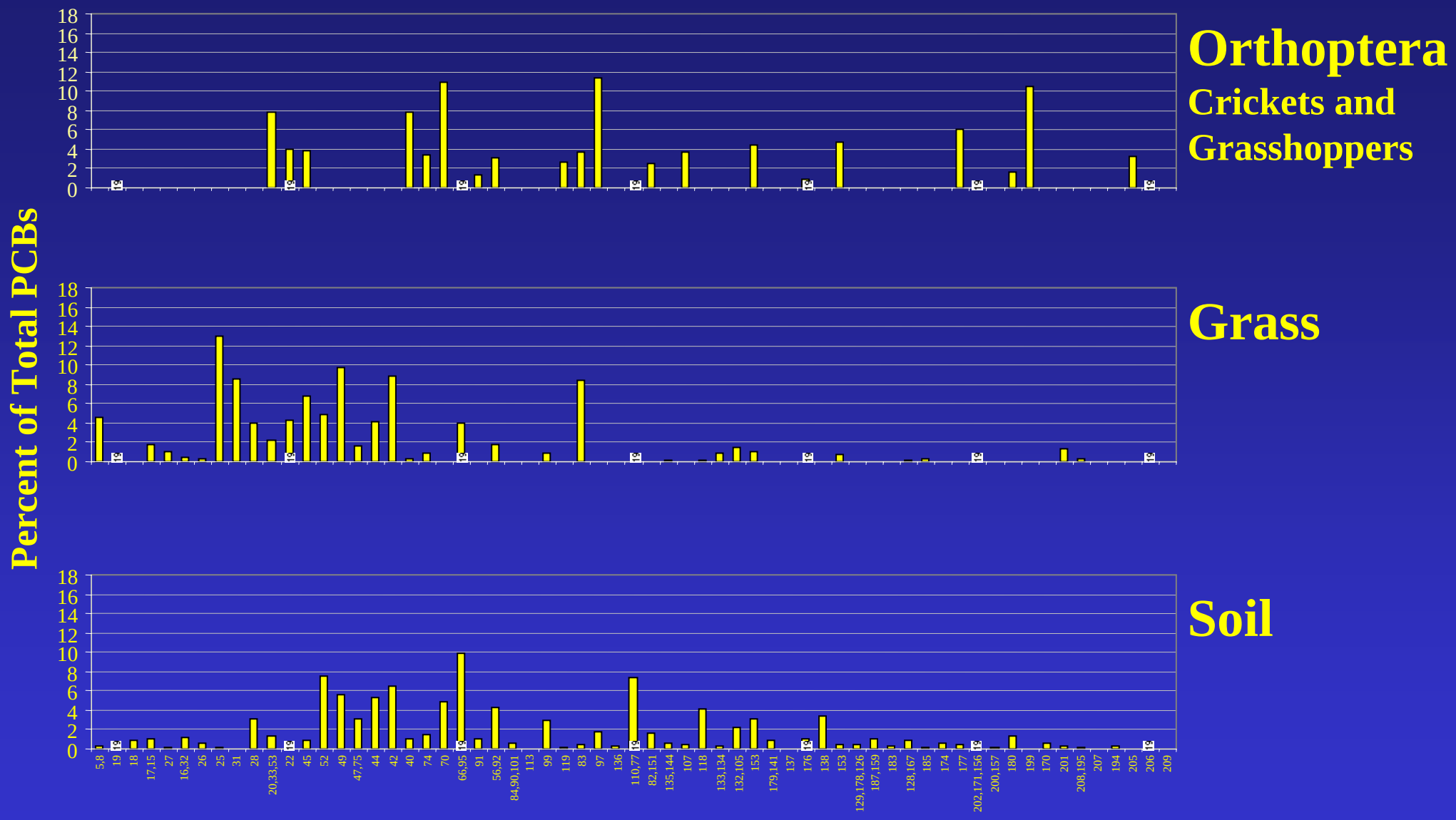

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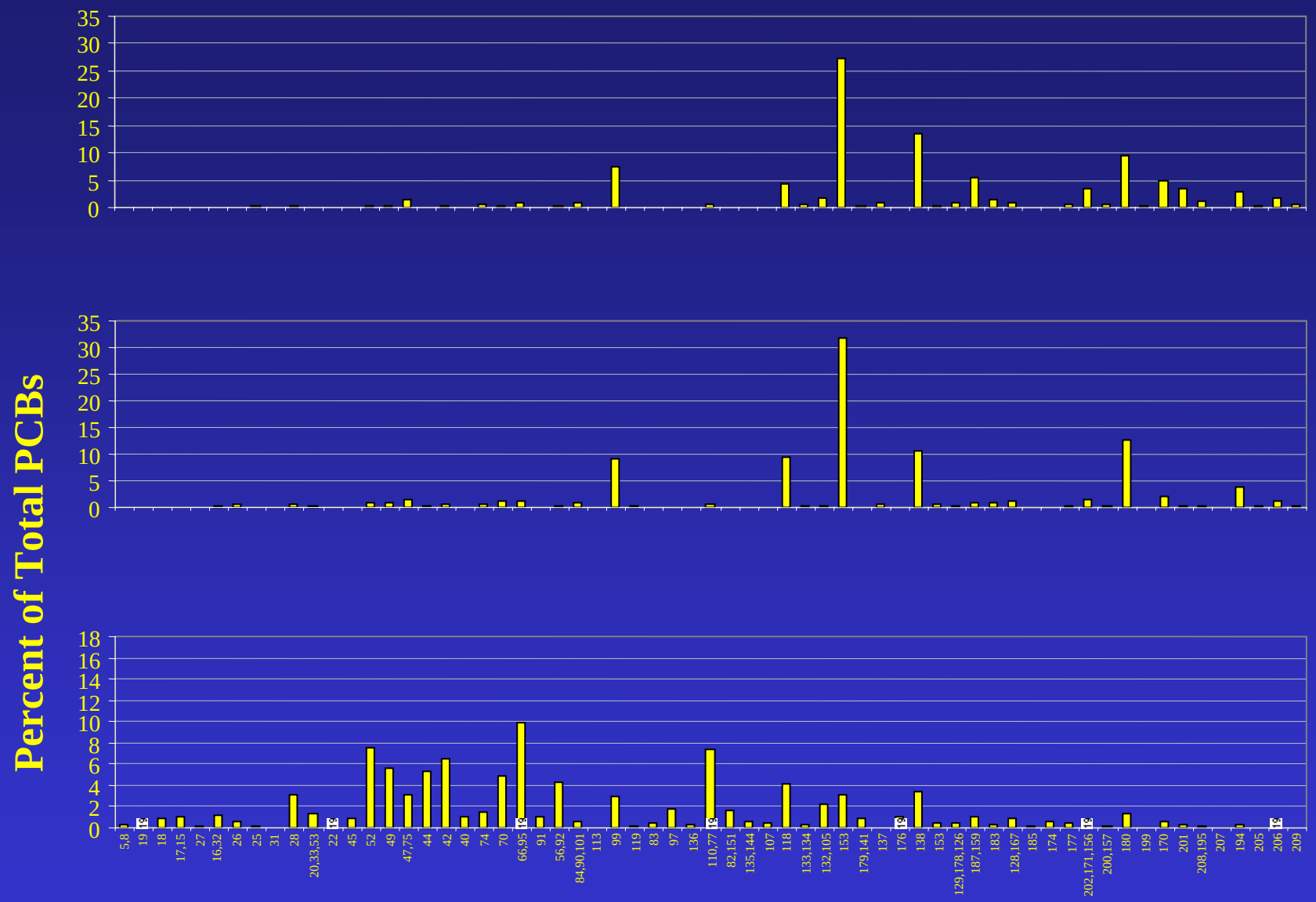
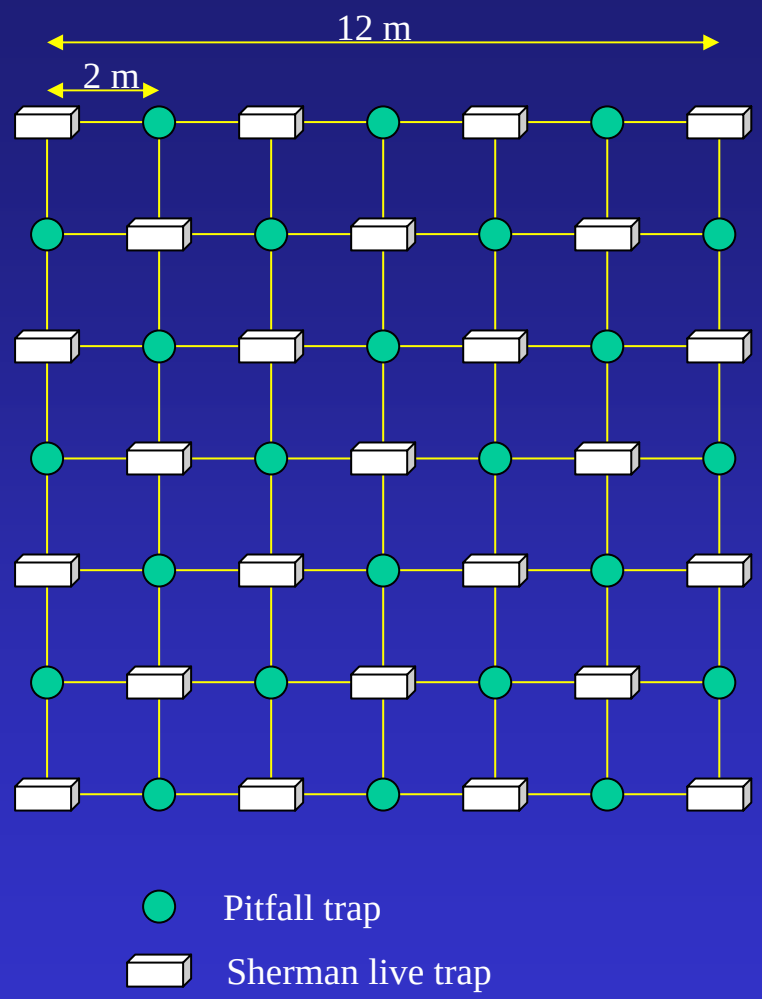

Percent of Total PCBs
Deer mouse
Shrew
Soil
35
30
25
20
15
10
5
0
18
16
14
12
10
8
6
4
2
0
5,8
19
18
17,15
27
16,32
26
25
31
28
20,33,53
22
45
52
49
47,75
44
42
40
74
70
66,95
91
56,92
84,90,101
113
99
119
83
97
136
110,77
82,151
135,144
107
118
133,134
132,105
153
179,141
137
176
138
153
129,178,126
187,159
183
128,167
185
174
177
202,171,156
200,157
180
199
170
201
208,195
207
194
205
206
209

Key Findings - PCB Congener Patterns in Food Web

- The earthworm and beetle composite samples exhibited the most similar pattern to co-located soil
- PCB concentrations in plants were primarily composed of less chlorinated PCBs (e.g., mono-, di-, tri-, and tetra-CBs) which are generally more volatile
- The orthoptera sample and especially the small mammal sample showed substantial differences from soil - notably enrichment of PCB 153 in mammals

Small Mammal Sampling



Short-Tailed Shrew

- Rationale
  - Expected to have greatest exposure among small mammals (e.g., worst case)
  - Meets the criteria of a good sentinel species since it occupies a small home range, has a short lifespan, and has high potential exposure
- Data Gaps
  - Tissue residue concentrations of PCB congeners including the most toxic, non-*ortho* coplanar PCBs
  - Dietary exposure to PCBs
  - Effects thresholds

Risk Characterization for Shrews

dietary exposure	Dietary-Based Hazard Quotient* NOAEL	Dietary-Based Hazard Quotient* LOAEL	Tissue Residue-Based Hazard Quotient*
mean	0.9	0.2	0.8
95% UCL mean	1.4	0.3	1.1

*dietary NOAEL and LOAEL of 0.69 and 3.2 mg/kg-d; tissue residue-based NOAEL of 73.3 mg/kg (lipid-normalized) Basis of TRVs: two-generation reproduction study in rats (Linder et al., 1974; Grant et al., 1974); also Russell 1998 study

Key Findings - Shrews

- The dietary-based HQs and tissue-residue based HQs are less than 1.0 or very close to 1.0
- Trapping results indicated that viable populations of shrews and other small mammals are present at the former Trowbridge impoundment

Conclusions

- Concentrations of PCBs in food web items collected from the former Trowbridge impoundment are substantially greater than at the Fort Custer Reference site
- Concentrations of PCBs in food web items are less than in soil from the same locations
- Congener patterns are considerably different within different parts of the terrestrial foodweb
- Dietary-based and tissue residue-based hazard quotients suggest that PCB risk to shrews is minimal

Acknowledgments

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