6 Ecological Risk Assessment

6.1 Introduction

The goal of the baseline risk assessment (BLRA) is to support the Fox River RI/FS process by evaluating whether sufficient ecological risks exist at the site to warrant a remedial action. Furthermore, the BLRA defines protective sediment quality thresholds for PCBs for ecological receptors in the Lower Fox River and Green Bay, from which a range of cleanup action levels can be selected for the FS.

The approach used for the ecological BLRA of the Lower Fox River site followed an established framework and guidelines for assessing ecological risks. Specifically, the EPA guidance for ecological risk assessment (1997a) established an eight-step process for assessing ecological risk. The first two steps are the screening level evaluation with the goals of determining if the site poses no or negligible ecological risk, and identifying which contaminants and exposure pathways require further evaluation. Steps three through seven detail the development of a BLRA. Step eight discusses risk management and will not be addressed as part of this BLRA report. Additionally, the WDNR (1992) has issued ecological risk assessment guidance that is compatible with the EPA (1997a, 1998b) guidance.

The BLRA evaluates the 10 COPCs identified in the SLRA with more site-specific information, and will expand on potential ecological concerns. This BLRA builds on the preliminary draft BLRA for the Lower Fox River and Green Bay (ThermoRetec, 1999), by incorporating new data released since the last draft, and responds to concerns raised by public comment to the previous draft. New information was only considered for evaluation up to October 1, 1999. Additionally, northern Green Bay (Zone 4), which was not previously evaluated, will be assessed for risk in this BLRA.

The EPA Environmental Response Team (ERT) has prepared a risk assessment for PCBs in Green Bay Zone 4 (EPA, 2000a) (Appendix C). Inclusion of Zone 4 in the present BLRA is a separate assessment. It was determined that inclusion of Zone 4 with the current BLRA had several advantages including:

- A larger database containing validated data,
- Interpolated sediment concentrations,

- Includes other COPCs in addition to PCBs, and

- Comparable criteria and modeling efforts between all Lower Fox River reaches and Green Bay zones.

Specifically, the following components of the SLRA will be further refined to address baseline risk:

- Chemical fate, transport, and degradation;
- Ecological receptors;
- Exposure routes;
- Extent of exposure;
- Extent and likelihood of threats or impacts; and
- Uncertainty associated with the calculation of risk.

A Biological Technical Assistance Group (BTAG) meeting was convened during the formulation of the SLRA to discuss the approach and procedures for performing the site-specific ecological risk assessment for the Lower Fox River/Green Bay system. BTAG review and consultation was also used in the completion of the BLRA. As mentioned in Section 3.3, the resource agencies, risk managers and biologists/ecologists in the BTAG included:

- Wisconsin Department of Natural Resources;
- U.S. Fish and Wildlife Service;
- U.S. Environmental Protection Agency, Region 5;
- U.S. Environmental Protection Agency, Environmental Response Team;
- National Oceanic and Atmospheric Administration;
- Menominee Nation; and
- Oneida Nation.

6.2 Problem Formulation

The Problem Formulation for the BLRA builds on the Problem Formulation presented in the SLRA (RETEC, 1998b), focusing on the site conceptual model and identifying appropriate standards and criteria to assess data collected. The conceptual site model establishes complete exposure pathways, and relates assessment and measurement endpoints for the Characterization of Exposure (Section 6.4) and Risk Characterization (Section 6.6) for this site. In addition, the species chosen for the conceptual model were also used in modeling to develop sediment quality thresholds (Section 7).

Specific areas of the SLRA Problem Formulation have been revised, including:

- **Contaminants of Potential Concern:** Based upon the SLRA risk analysis, a reduced list of chemicals will be evaluated in the BLRA.

- **Contaminant Fate and Transport:** The migration pathways through which COPCs may enter into the Lower Fox River and Green Bay ecosystems are defined.

- **Assessment and Measurement Endpoints:** The measurement endpoints are the means by which the risk to the assessment endpoints are evaluated.

- **Conceptual Model:** The integration of information on sources, exposure pathways, and ecological receptors to describe how receptors may become exposed to COPCs, and potentially be placed at risk. Three area-specific conceptual models will be used to evaluate risk in the Lower Fox River and Green Bay.

Each of these elements of the Problem Formulation are discussed in more detail in the sections below.

6.2.1 Contaminants of Potential Concern

A description of the contaminants known to exist in the Lower Fox River and Green Bay system was provided in Section 1.1. The identified COPCs to be carried forward in the BLRA was provided in Section 2.5.1 (see Appendix A). These COPCs are: PCBs (total and PCB congeners), 2,3,7,8-TCDD/2,3,7,8-TCDF, DDT/DDE/DDD, dieldrin, mercury, lead, and arsenic. PCBs were carried forward in the BLRA as the primary COPC because SLRA-calculated sediment hazard quotients (HQs) ranged from 1,514 to 5,872, generally several orders of magnitude greater than HQs for other COPCs. Although 2,3,7,8-TCDD is the most toxic dioxin congener, all structurally related dioxin and furan congeners will be evaluated for toxicity based on the toxicity equivalency method, further described in Section 6.3.2. The dioxin and furan congeners that will be evaluated are those that have been measured in site media and those that have toxic equivalency factors (TEFs). The only PCB congeners that will be evaluated for dioxin-like toxicity are those that most structurally resemble dioxin and have the greatest potential for bioaccumulation: congeners 77, 81, 105, 118, 126, and 169, as further discussed in Section 6.3.3.

6.2.2 Contaminant Fate and Transport

General chemical fate and transport processes within the Lower Fox River and Green Bay were previously described (Section 2.5). This section describes, for each COPC, the unique chemical and physical properties that govern the mobility,

and hence, the fate and transport of each COPC. Table 6-1 identifies fate and transport properties of potentially bioaccumulating chemicals of concern.

As previously discussed in Section 2.4, contaminants are found in the Lower Fox River and Green Bay as a result of releases from point and non-point sources. Currently, the principal source for COPCs is the contaminated sediment deposits found throughout the system. The principal transport mechanism is sediment resuspension, with transport occurring by downstream currents in the Lower Fox River, and by discrete resuspension-transport-deposition events within Green Bay (WDNR, 1998b, 1998c). The fate of these contaminants, following their release into the water column, depends on the chemical properties of the contaminant, abiotic factors within the receiving environment (e.g., organic carbon in sediments, pH, surface water hardness), and interaction with the biotic environment. This interaction can result in degradation, transformation, or bioconcentration of the contaminant. The fate of a contaminant is not fixed, and the degree of contaminant exchange between surface water, sediment, sediment pore water, and biota varies. The predicted transport and fate for each COPC is described below.

Organic Constituents

Organically-contaminated sediments are often complex mixtures of numerous compounds that will separate and partition into sediments and water, based on the properties of the individual chemicals present. The primary property that governs the extent to which a chemical will partition between sediment, pore water, and surface water is its solubility limit. Water solubility limits set the maximum dissolved-phase concentrations for pure compounds or compounds present in dilute solutions. The organic COPCs for the Lower Fox River and Green Bay are all non-polar organic compounds. Such compounds generally show a higher affinity for partitioning to sediment rather than water. Dissolution of non-polar organic chemicals are further controlled by their affinity for organic carbon phases in sediments or water. Affinity for organic carbon is generally determined from the octanol-water partitioning coefficient (K_{ow}) of the chemical, where the higher the K_{ow}, the greater the affinity for partitioning to organic carbon and the lesser potential for dissolution in water. Vapor pressure and Henry's Law Constant are indicators of a chemical's tendency to partition between water and the atmosphere.

Bioaccumulation of non-polar organic compounds occurs as a result of uptake by a receptor, followed by partitioning of the compounds into the receptor's organic carbon compartment—the lipids. Therefore, bioaccumulation is highly dependent upon an organism's lipid content and on the affinity of the compound to partition into the organic phase, as measured by its K_{ow}. Generally, the relationship

between K_{ow} and lipids is assumed to be linear, except for extremely hydrophobic compounds (i.e., log $K_{ow} > 6$) (Bertelsen *et al.*, 1998).

Once chemicals are accumulated within an organism's lipid fraction, biomagnification may occur when organisms at lower trophic levels are preyed upon by receptors higher in the food chain. The net result is an aggregate increase in tissue body burdens of the chemicals at higher trophic levels. Non-polar organic compounds with log K_{ow}s between 4 and 6 have the greatest tendency to biomagnify within ecological food webs (Oliver and Niimi, 1988; Thoman, 1989).

Polychlorinated Biphenyls (PCBs). PCBs are a general class of chemically inert, non-polar, synthetic, halogenated hydrocarbons, of which there are 209 different compounds (congeners) (Eisler and Belisle, 1996). PCB congeners vary between one and 10 chlorine atoms substituted on the biphenyl ring, and are named according to the position of the chlorine substitution and the number of substitutions (e.g., 3,3',4,4',5-pentachlorobiphenyl). By convention, PCB congeners are usually referred to by the numerical designation given by the International Union of Pure and Applied Chemistry (IUPAC). For example, 3,3',4,4',5-pentachlorobiphenyl is commonly referred to as PCB 126.

PCBs in the environment are stable and persistent; cycling rather than degradation represents the predominant fate. Partitioning of PCBs in an aquatic system is dependent on sorption reactions which in turn are dependent on PCB characteristics such as solubility, vapor pressure, partition coefficients, structure, degree of chlorination, and media (sediment, tissue) characteristics such as lipid and organic carbon content.

PCBs are highly lipophilic and, therefore, more readily bind to sediments or accumulate in tissues rather than remain in the water column (Eisler and Belisle, 1996). PCB congeners that are less chlorinated are more soluble than highly chlorinated congeners; however, even less chlorinated congeners have a strong affinity for organic carbon (dissolved or particulate) in the water column. Therefore, it is the lower chlorinated congeners that tend to be transported by the water column while higher chlorinated congeners sorb more readily to sediments. Partitioning of lower chlorinated congeners between the suspended organic phase and the dissolved organic phase is generally seasonally dependent, because levels of suspended solids vary seasonally.

PCB sorption to sediments is the primary mechanism of removing PCBs from the water column, and is dependent on PCB congener K_{ow}. Generally, K_{ow} values increase with increasing chlorination and increasing the K_{ow} values increases the hydrophobicity and binding affinity. PCB log K_{ow}s reported by Eisler and Belisle

(1996) range from 4.15 (a trichlorobiphenyl) to 9.60 (a decachlorobiphenyl). PCB K_{ow} values for the specific planar PCBs of concern (IUPAC Nos. 77, 81, 105, 118, 126, and 169) range from 6.37 (congener 81) to 7.43 (congener 169). A potentially equally important mechanism for removing PCBs from the water column is volatilization. Lower chlorinated congeners may volatilize to the atmosphere depending primarily on wind speed and water column concentration.

Once PCBs are bound to sediment particles, they may be dispersed and diluted with clean sediments, deposited into quiescent zones in the river or bay, resuspended by high water or storm events, or buried by the addition of cleaner sediments. For the Lower Fox River and Green Bay, the principal means of reduction in PCB concentrations is by dilution or burial. Transformation processes such as volatilization, photo-oxidation, and hydrolysis are minor, but biodegradation by bacteria and fungi, although slow, may be significant (RETEC, 2002b). PCB metabolism and rate of metabolism by microorganisms is generally congener-specific. Both aerobic and anaerobic degradation can occur, but while aerobic degradation leads to intermediate compounds that can be further degraded by other microorganisms, anaerobic degradation results in dechlorination that reduces the amount of chlorine present, but does not change the overall concentration of total PCBs. For the Lower Fox River, both aerobic and anaerobic dechlorination processes have been demonstrated, but occur at very slow rates that do not result in any appreciable decrease in PCB concentrations in the system.

Compounds with high K_{ow}s, such as PCBs, not only have affinity for organic carbon in sediments, but also for lipids in organisms. The more lipophilic and hydrophobic a substance, the more concentrated the substance will be in the sediment and the phytoplankton of an aquatic system (Loizeau and Menesguen, 1993). Aquatic organisms can be exposed to PCBs through the water column, through ingesting sediments, and through consuming prey. For invertebrates, both aquatic and benthic, exposure to PCBs through contact with the water column or pore water contributes significantly to the total body burden of total PCBs. For most species, however, particularly those at high trophic levels, prey consumption is likely the primary route of exposure. MacDonald (1993) examined the distribution of PCB congeners in seven lake systems to determine the role of sediment/biota partitioning and food web transport. Results of this study indicated that food web transport is a greater determining factor for the concentration of PCBs in higher trophic levels. Biological uptake of PCBs by aquatic organisms appears to be species-specific. Rates of accumulation vary depending on species, age, sex, and size. Generally, when equally exposed, fish accumulate two to three times more PCBs than aquatic invertebrates (Eisler,

1986). Much of this uptake likely comes from prey consumption, but gill uptake of PCBs also contributes and is generally rapid (Bruggeman *et al.*, 1981).

Once ingested, PCBs may be metabolized by the cytochrome P450 enzyme system found in birds and mammals, but the degree of metabolism is specific to both the amount of chlorination and the species, and elimination rates are slower than uptake rates. It is generally the higher chlorinated congeners (tri- to penta-chlorinated PCBs) that are bioaccumulated. PCB congeners with eight or more chlorine atoms are structurally limited, because of their size, from passing biological membranes. Equilibrium partitioning, proposed as the principal mechanism of PCB bioaccumulation, suggests that PCB concentrations in the adipose tissue of predators are proportional to those levels present in their environment and prey (Foley *et al.*, 1988).

Although high PCB residue levels have been detected in fish, mammals, and birds worldwide (Eisler and Belisle, 1996), high concentrations alone may not be predictive of adverse effects. Some organisms are capable of storing extremely high concentrations of PCBs in their fat without any apparent detrimental effect (Olafsson *et al.*, 1983), yet when fat stores are used for energy, mobilized PCBs may cause adverse effects (Landis and Yu, 1995).

Dioxins and Furans. Polychlorinated dibenzodioxins (PCDD), as a group, represent 75 different positional isomers, while polychlorinated dibenzofurans (PCDF) comprise over 135 compounds (ATSDR, 1998a). These two chemical classes are generally referred to as dioxins. Tetra-chloro dibenzodioxins (TCDD) and tetra-chloro dibenzofurans (TCDF) are a subset of PCDD and PCDF compounds, respectively. Unlike PCBs, dioxins have never been purposely manufactured, but are found as trace impurities in chlorophenols, chlorinated herbicides, and commercial Aroclor mixtures, or are incidental byproducts of some bleached kraft paper processes or combustion (e.g., in fly ash or produced in forest fires) (Hoffman *et al.*, 1996).

The fate and transport of dioxins are analogous to PCBs. Dioxins preferentially associate with particulate or organic matter because of their high lipophilicity and low water solubility (Boening, 1998; McKim *et al.*, 1985). Once sorbed to particulate matter or bound in the sediment organic phase, they exhibit little potential for leaching or volatilization. They are highly stable in all environmental media, with persistence measured in decades. The only environmentally significant transformation process for these congeners is believed to be photodegradation of chemicals not bound to particles in the gaseous phase or at the soil- or water-air interface (EPA, 1994a). Bacterial degradation of dioxins and furans is possible, but is a very slow process.

Dioxins and furans have been found to highly bioconcentrate in aquatic food webs (ATSDR, 1998a). Thus, the principal route of exposure through the Lower Fox River/Green Bay food web is via ingestion of contaminated food, as opposed to respiration across gill surfaces for fish or aquatic invertebrates.

Dichlorodiphenyl-trichloroethane (DDT). Dichlorodiphenyl-trichloroethane (DDT), having DDD and DDE as principal metabolites, is an organochlorine compound used as insecticide until banned for use in the United States in 1973 because of adverse toxicity to wildlife. Both DDD and DDE are stable and biologically active, although DDE is non-insecticidal (Montgomery, 1996). In soils, under aerobic conditions, DDT is rapidly converted to DDD and very slowly converted to DDE via reductive dechlorination (Montgomery, 1996). In sediments, however, DDE is the major metabolite formed from DDT (Montgomery, 1996). Although DDE is slightly soluble in water, DDT is less soluble and strongly adheres to suspended sediment particles (ATSDR, 1998b).

DDT is not readily metabolized by animals, but is primarily stored in lipids. Its biological half-life is approximately 8 years. Biologically accumulated DDT may be metabolized to another form (i.e., DDT may be transformed to DDE). When lipid reserves are metabolized, the DDT or metabolites are released into the system, where a toxic response may result. DDT may act as a direct toxin to some receptors; however, because of its tendency to concentrate in biological tissues, higher trophic level receptors may be at increased risk through ingestion of contaminated food sources.

Dieldrin. Dieldrin is a non-systemic and persistent cyclodiene insecticide. It was broadly used in the United States until 1974, when the EPA restricted its use to termite control via direct soil injection, and to non-food seed and plant treatment. Dieldrin is no longer produced commercially in the United States.

Dieldrin has a low volatility, sorbs readily to sediment organic matter, and has a high potential for bioaccumulation (bioaccumulation factor [BAF] = 4,670) (EPA, 1992b). Dieldrin is persistent in sediments and surface water, with half-lives of 3 and 6 years, respectively (Howard *et al.*, 1991). Direct photolysis of dieldrin can occur, creating a half-life of about 2 months (EPA, 1992b). Dieldrin's degradation is unaffected by aerobic or anaerobic conditions (Montgomery, 1996), but can be biotransformed by soil microbes to a substance more toxic to insects (EPA, 1992b).

Inorganic Constituents

The fate of metals in the aquatic environment is determined by the interaction of many variables. The primary factor influencing the fate and transport of metals

is their speciation and adsorption capacity. When metals are released into the environment, their speciation and adsorption capacity are affected by, and changed with, the geochemistry of the environment.

Several factors influence adsorption and speciation of metals. The first of these factors is the presence of competing ions. In instances where metals are present in solution with other ions, competition for sorption sites on soil particles or on organic material may enhance the mobility of weakly sorbed metals. Adsorption of metals is also strongly influenced by pH. This is due, in part, to increased competition between protons (H^+) and metal ions for the same binding sites. Furthermore, pH affects the speciation and solubility of metals through the formation of hydroxide complexes. Speciation of metals is also controlled by the reduction/oxidation (redox) potential of the environment, which determines the oxidation state of the metal. For example, in an oxidized environment, arsenic is generally present as arsenate (As^{5+}), which ionically binds to soil and is immobilized. However, under reduced conditions, arsenate is transformed to arsenite (As^{3+}), which is water soluble and, therefore, more mobile.

Because of these complex interactions, total metals concentrations are generally not predictive of the bioavailability. One measure of metals bioavailability is the dissolved fraction of metals in surface water or pore water. However, consideration must still be given to the residual chemistry, including pH and dissolved organic carbon.

In summary, the degree to which a metal will adsorb to organic matter depends on the presence of competing ions, water chemistry, and metal speciation, which is, in turn, affected by such factors as pH and redox potential. The interaction among these factors is complex. Fate and transport of individual metals are discussed below.

Arsenic. Arsenic in water can react through oxidation, reduction, or methylation. Generally, arsenic preferentially binds to sediments and naturally occurs as sulfides of iron, nickel, and cobalt (Eisler, 1988a). Binding to sediments is dependent on the concentration of arsenic and sediment characteristics such as pH, ionic strength, Eh, and the presence of other compounds in sediments.

Arsenic in water exists primarily as a dissolved ionic species. Particulates account for less than 1 percent of the total measurable arsenic. Arsenates are more strongly adsorbed to sediment than are other arsenic forms. In bodies of water that become stratified in summer, arsenic released from sediments accumulates in the hypolimnion until turnover, when it is mixed with epilimnetic waters. This

mixing may result in a 10 to 20 percent increase in arsenic concentrations (Eisler, 1988a).

Arsenic exists in four oxidation states as inorganic or organic forms; its bioavailability and toxic properties are significantly modified by numerous biological and abiotic factors. In general, inorganic arsenic compounds are more toxic than organic compounds, and trivalent species are more toxic than pentavalent species. Arsenic is accumulated in a variety of organisms from the water; however, there is no evidence of biomagnification through the food chain. Bioconcentration factors are low in aquatic organisms, except for algae (Eisler, 1988a).

Lead. Lead in aquatic environments often precipitates out of solution by binding to carbonate or phosphate ions, and can be readily sorbed to either organic or inorganic components in sediments. Factors affecting the degree of sorption include: the sediment type, pH, organic carbon content, cation exchange capacity, the form of lead, and other constituents in the sediment, such as metal oxides, aluminum silicates, and carbonates. Sorption is higher in sediments containing clay, and lower in sediments containing a higher percentage of sand or sand and loam (Eisler, 1988b). Bioavailable lead in sediments is also governed by the amount of acid volatile sulfides within the sediment pore water (Ankley, 1996; Di Toro *et al.*, 1990).

Lead does not biomagnify to a great extent in food chains, although accumulation by plants and animals has been extensively documented (Wixson and Davies, 1993; Eisler, 1988b). Older organisms typically contain the highest tissue lead concentrations, with the majority of accumulation occurring in the bony tissue of vertebrates (Eisler, 1988b).

Predicting the accumulation and toxicity of lead is difficult since its effects are influenced to a large degree, relative to other metals, by interactions among physical, chemical, and biological variables. In general, organo-lead compounds are more toxic than inorganic lead compounds, and young, immature organisms are more susceptible to its effects (Eisler, 1988b).

Mercury. Mercury may be present in the environment in a number of forms and can exist in three oxidation states: elemental mercury (Hg^0), mercurous ion (Hg_2^{2+}), and mercuric ion (Hg^{2+}). Of all the inorganic forms, Hg^{2+} is the most toxic. Nonvolatile inorganic forms of mercury compounds sorb readily to sediments, particularly those sediments containing high organic carbon levels. Mercury forms stable complexes with organic compounds and are not easily removed from sediments (Eisler, 1987). Mobilization of sorbed mercury can be caused by

bioreduction to elemental mercury and bioconversion to more volatile and soluble forms, such as methylmercury.

The most toxic and bioavailable form of mercury is an organic form, methylmercury, which is highly stable, lipophilic, and accumulates in food chains. The majority of mercury detected in biological tissues is present in the form of methylmercury (Huckabee *et al.*, 1979).

Mercury can become methylated biologically or chemically. Microbial methylation of mercury occurs most rapidly under anaerobic conditions, common in wetlands and aquatic sediments. Mercury methylation in ecosystems depends on mercury loadings, microbial activity, nutrient content, pH, redox conditions, suspended sediment load, sedimentation rates, and other variables (Eisler, 1987). Conversion of inorganic mercury to methylmercury is favored by low pH and low dissolved organic carbon levels.

Mercury bioaccumulation and biomagnification have been demonstrated in the aquatic food chain: elevated levels have been found in piscivorous fish as compared with organisms lower on the food chain. Almost all mercury accumulated is in the methylated form, primarily as a result of the consumption of prey containing methylmercury (Eisler, 1987).

6.2.3 Assessment and Measurement Endpoints

As the culmination of the Problem Formulation phase of this risk assessment, endpoints have been derived to assess the risks posed by COPCs to the Lower Fox River and Green Bay biological receptors. This section presents those endpoints.

Assessment endpoints are explicit expressions of the environmental values (e.g., ecological resources) that are to be protected. Four principal criteria are used to select ecological values that may be appropriate for assessment endpoints: 1) ecological relevance, 2) susceptibility to known or potential stressors, 3) commercial or social value, and 4) relevance to management goals (EPA, 1998b). Adverse risk to assessment endpoints drive any potentially necessary risk management decisions.

Assessment endpoints generally are populations or communities (e.g., invertebrates or birds). Populations or communities may be deemed at risk if reproduction or survival of individuals are determined to be significantly impacted.

While the assessment endpoints (and conceptual model) help risk assessors identify measurable attributes to quantify for risk estimation, often the assessment

endpoints cannot be measured directly. Measures must be selected to determine whether the assessment endpoint is at risk. There are three categories of measures: measures of effects (measurement endpoints), measures of exposure, and measures of ecosystem receptor characteristics (EPA, 1998b). This BLRA selected only measurement endpoints (measures of effects) for its risk analysis.

Measurement endpoints are quantifiable ecological characteristics, through laboratory or field experimentation, that are related to the valued characteristic chosen as the assessment endpoint (EPA, 1992a). The measurement endpoint should be sensitive, and represent the same exposure pathway and mechanism of toxicity as the assessment endpoint it represents.

Both the assessment and measurement endpoints used for the BLRA were determined through iterative discussions with both the WDNR and with the BTAG. The assessment and measurement endpoints discussed below were refined after review of agency, industry, and public comments on the 1999 Preliminary Draft BLRA.

Assessment Endpoints

Appropriate selection and definition of assessment endpoints, which focus the risk assessment design and analysis, are critical to the utility of risk assessment. It is not practical, nor possible, to directly evaluate risks to all of the individual components of the ecosystem at the site. Assessment endpoints were selected for the risk assessment based on particular components of the ecosystem that could be adversely affected by the contaminants present.

A review of the habitat and ecology of the Lower Fox River and Green Bay, as discussed in Section 2, provided information for the selection of assessment endpoints. As noted in Section 2, the Lower Fox River and Green Bay provide habitat function for a variety of invertebrates, fish, birds, and mammals that inhabit or use this watershed for foraging, reproducing, and rearing.

Eight assessment endpoints were developed to evaluate the risk of contaminants in the Lower Fox River and Green Bay (Table 6-2). By evaluating and protecting these assessment endpoints, it is assumed that this ecosystem as a whole would also be protected. Each assessment endpoint is discussed in detail below.

Invertebrates

Invertebrate communities constitute a vast portion of the basis of the food chain in aquatic ecosystems. Since invertebrates process organic material and are prey items for other invertebrates, fish, and birds, they are important in nutrient and energy transfer in an aquatic ecosystem. Alterations in invertebrate functions may

consequently affect nutrient and energy transfer, and bird and fish populations. Also, COPCs in invertebrates may be passed along through the food chain. Therefore, upper trophic levels can be affected not only by reduced prey abundance, but also by trophic transfer of accumulated contaminants in invertebrate prey.

Functioning Water Column Invertebrate Communities. Pelagic communities inhabit the water column and include both phytoplankton and zooplankton. Phytoplankton are small uni- or multi-cellular algae and form the base of the pelagic food chain. Zooplankton in turn consume phytoplankton and, depending on populations levels, phytoplankton levels can either be limited or overabundant. If phytoplankton become overabundant (i.e., they are not sufficiently grazed by zooplankton) then they eventually die, settle to the sediment surface, and, as detritus, become part of the benthic food chain.

Functioning Benthic Invertebrate Communities. Benthic invertebrate communities are heterogeneous assemblages of organisms that inhabit bottom substrates and, like pelagic invertebrates, constitute a vast portion of the basis of the food chain in aquatic ecosystems. Benthic invertebrates are susceptible to COPC exposure because they live and feed directly in the sediment, where most contaminants are concentrated.

Benthic invertebrates play several important roles in the aquatic community, including the mineralization and recycling of organic matter and, therefore, nutrient and energy cycling which supports the productivity of the entire ecosystem. Also, benthic invertebrates are important trophic links in aquatic communities because they consume bacteria, plankton, and detritus, and are a dominant prey base for certain species of fish, birds, and other benthic organisms. Examples of important benthic invertebrates in the Lower Fox River system include chironomids (midges) and oligochaetes (segmented worms).

Fish

Fish have many roles in the aquatic ecosystem, including the transfer of nutrients and energy, and are prey for mammals, birds, and predatory fish. In fact, several predators rely solely, or primarily, on fish for survival. Fish typically constitute a large proportion of the biomass in aquatic systems. Additionally, fish have social and economic value; impaired fish communities would adversely affect commercial and recreational fishing.

Benthic Fish Reproduction and Survival. Benthic fish are those fish that live in contact with and forage for food directly in the sediments. As such, they represent a unique exposure pathway because of their foraging behavior (i.e., high

exposure to sediments) and prey items (i.e., predominately benthic invertebrates). Impairment to benthic fish communities could have strong impacts on nutrient and energy cycling, and on instream and nearby upland biological communities. Examples of benthic fish in the Lower Fox River include carp, catfish, and bullhead.

Pelagial Fish Reproduction and Survival. Pelagial fish were selected as an assessment endpoint because they have a different exposure pathway than benthic fish. Pelagial fish are those species that live and feed principally in the water column (as opposed to being in direct contact with sediment). Pelagial fish represent many trophic levels with prey items predominately in the water column (e.g., zooplankton and other fish). Upper trophic level pelagial fish may be strongly impacted by food chain transfers of COPCs. As with the benthic fish, several pelagial fish species are of commercial and/or recreational importance and a decline in these fish species could have economic impacts on the region. Examples of important pelagial fish in the Lower Fox River include shiners, shad, alewife, perch, and walleye. Pelagial fish important to Green Bay include the same species as are found in the river, in addition to lake trout and other salmonids in the upper bay.

Birds

Bird populations, in general, present one of the most significant biological components of the river/bay system and occupy several trophic levels. Birds play an important role in energy transfer and nutrient processing between the aquatic and terrestrial communities. Given the potential for some contaminants to biomagnify, birds, as upper trophic level receptors, may concentrate, and be affected by, contaminants in their tissues to a greater degree than lower trophic level species. In addition to their ecological importance, birds are socially valued because of recreational activities and aquatic aesthetics.

Insectivorous Bird Reproduction and Survival. Insectivorous birds rely predominately on insects for food. In the Lower Fox River system, this is limited to birds that forage on insects which spend a large part of their life history in contact with contaminated sediments, and are preyed upon during their brief adult hatch-out period. Contaminants may be transferred from sediments to insects to the birds. Insectivorous birds serve an important function in regulating insect populations. Examples of insectivorous birds in the Lower Fox River and Green Bay region include swallows and blackbirds.

Piscivorous Bird Reproduction and Survival. Piscivorous birds rely primarily on fish for food. Of the bird populations present at the site, piscivorous birds represent a high trophic level and, therefore, are more at risk than insectivores from

contaminants transferred through the food chain. Examples of piscivorous birds on the Lower Fox River and Green Bay include cormorants and terns.

Carnivorous Bird Reproduction and Survival. Carnivorous birds were selected for evaluation because of their diverse forage, which can include consumption of fish, piscivorous birds, or even small mammals. Prey preferences of carnivorous birds can influence population levels of prey species. Examples of carnivorous birds on the Lower Fox River and Green Bay include eagles, osprey, and other raptors.

Mammals

Mammals are high trophic level predators and can link nutrient transfer between aquatic and terrestrial systems. Mammals can also be prey for other mammals or birds. The mammal species of concern for the Lower Fox River system are those that are aquatic-based.

Piscivorous Mammal Reproduction and Survival. Piscivorous mammals represent the upper trophic level of the riverine corridor ecosystem and, therefore, are potentially highly exposed to contaminants that bioaccumulate or biomagnify. Piscivorous mammals rely primarily on fish as food, but may also consume amphibians, invertebrates, crayfish, clams, and mussels. The foraging behavior of these mammals represents a pathway through which energy is transferred from the aquatic to terrestrial ecosystem. Mink and river otter are piscivorous mammals found in the Lower Fox River and Green Bay area.

Risk Questions

Risk questions have been formulated based upon the assessment endpoints. Risk questions serve to provide a focal point for evaluating the specific measurement endpoints to the assessment endpoints. The measurement endpoints are evaluated to answer the specific risk questions. Based on information collected during this problem formulation phase, the following risk questions have been formulated:

- **For the Assessment Endpoint—Functioning Water Column Invertebrate Communities:** *Are levels of site contaminants in surface water sufficient to cause adverse alterations to the functioning of water column invertebrate communities?*

- **For the Assessment Endpoint—Functioning Benthic Invertebrate Communities:** *Are levels of site contaminants in sediment sufficient to cause adverse alterations to the functioning of benthic invertebrate communities?*

- **For the Assessment Endpoint—Benthic Fish Survival and Reproduction:** *Are levels of site contaminants sufficient to cause survival or reproductive impairment in benthic fish?*

- **For the Assessment Endpoint—Pelagial Fish Survival and Reproduction:** *Are levels of site contaminants sufficient to cause survival or reproductive impairment in pelagial fish?*

- **For the Assessment Endpoint—Insectivorous Bird Survival, Physiology, and Reproduction:** *Are levels of site contaminants sufficient to cause survival or reproductive impairment or deformity in insectivorous birds?*

- **For the Assessment Endpoint—Piscivorous Bird Survival, Physiology, and Reproduction:** *Are levels of site contaminants sufficient to cause survival or reproductive impairment or deformity in piscivorous birds?*

- **For the Assessment Endpoint—Carnivorous Bird Survival, Physiology, and Reproduction:** *Are levels of site contaminants sufficient to cause survival or reproductive impairment or deformity in carnivorous birds?*

- **For the Assessment Endpoint—Piscivorous Mammal Survival and Reproduction:** *Are levels of site contaminants sufficient to cause survival or reproductive impairment in piscivorous mammals?*

Table 6-2 presents the assessment and measurement endpoints selected to test the above risk questions, as well as the model receptor species/population. The measurement endpoints are discussed in the following section.

Measurement Endpoints

Risk questions are assessed using measurement endpoints. Types of measurement endpoints used in the risk assessment process fall generally into four categories: 1) comparison of estimated or measured exposure levels of COPCs to levels known to cause adverse effects, 2) bioassay testing of site and reference media, 3) *in-situ* toxicity testing of site and reference media, and 4) comparison of observed effects on-site with those observed at a reference site. Measurement endpoints selected for assessment endpoint evaluation in this risk assessment consistently fell in to the first category of measurement endpoints. Only existing data were evaluated as part of this assessment.

The following measurement endpoints were identified for each of the assessment endpoints and their respective risk questions.

- **Functioning Water Column Invertebrate Communities—***Are levels of site contaminants in surface water sufficient to cause adverse alterations to the functioning of water column invertebrate communities?*

  To address this risk question, surface water chemistry data will be evaluated and compared to water ecological benchmarks. This evaluation will determine whether water column invertebrate communities are impacted by site contaminants.

- **Functioning Benthic Invertebrate Communities—***Are levels of site contaminants in sediment sufficient to cause adverse alterations to the functioning of benthic invertebrate communities?*

  To address this question, sediment chemistry data will be evaluated and compared to sediment ecological benchmarks. These evaluations will determine whether benthic invertebrate communities are impacted by site contaminants.

- **Benthic Fish Survival and Reproduction—***Are levels of site contaminants sufficient to cause survival or reproductive impairment in benthic fish?*

  This risk question will be answered through evaluation of whole benthic fish tissue concentrations as compared to Toxicity Reference Values (TRVs).

- **Pelagial Fish Survival and Reproduction—***Are levels of site contaminants sufficient to cause survival or reproductive impairment in pelagial fish?*

  This risk question will be answered through evaluation of whole pelagial fish tissue concentrations as compared to TRVs.

- **Insectivorous Bird Survival, Physiology, and Reproduction—***Are levels of site contaminants sufficient to cause survival or reproductive impairment or deformity in insectivorous birds?*

  Risk to insectivorous birds will be evaluated through examination of measured whole body and egg COPC concentrations as compared to TRVs.

- **Piscivorous Bird Survival, Physiology, and Reproduction—***Are levels of site contaminants sufficient to cause survival or reproductive impairment or deformity in piscivorous birds?*

Risk to piscivorous birds will be evaluated through examination of measured whole body, egg, or brain COPC concentrations, or estimated dietary COPC concentrations as compared to TRVs.

- **Carnivorous Bird Survival, Physiology, and Reproduction—***Are levels of site contaminants sufficient to cause survival or reproductive impairment or deformity in carnivorous birds?*

  Risk to carnivorous birds will be evaluated through examination of measured egg or liver COPC concentrations, or estimated dietary COPC concentrations as compared to TRVs.

- **Piscivorous Mammal Survival and Reproduction—***Are levels of site contaminants sufficient to cause survival or reproductive impairment in piscivorous mammals?*

  Risk to piscivorous mammals will be evaluated through comparing estimated dietary COPC concentrations to dietary TRVs because tissue concentrations from piscivorous mammals are not available.

Where field population or other observational data are available, these will be used for comparison to HQs for each reach or zone in the system.

6.2.4 Conceptual Site Model

For toxicity to occur, both a contaminant and a receptor must be present and there must be a complete exposure pathway by which the receptor is exposed to the contaminant. The physical contaminant fate and transport model was presented in Section 2. The biological conceptual model identifies where contaminant interactions with biota can occur, describes the uptake of site contaminants into the biological system (in this case, the water and sediments of the Lower Fox River and Green Bay), and diagrams key receptor contaminant exposure pathways. Due to the large area being assessed for risk, more than one conceptual model was necessary. The Lower Fox River, from the mouth of Lake Winnebago to the De Pere dam, was evaluated using the same conceptual model (Figure 6-1). The De Pere dam restricts movement of alewife and rainbow smelt further up the Lower Fox River. Therefore, the Lower Fox River below the De Pere dam and Zone 2 of Green Bay were evaluated for risk based on a second conceptual model (Figure 6-2). Finally, the remaining zones of Green Bay (zones 3 and 4) were evaluated using a third conceptual model (Figure 6-3). Zones 3 and 4 of Green Bay are distinct from the southern zones of Green Bay because the water is deeper and colder in the northern bay and different receptor species

reside there. The only differences in conceptual model receptor species between these three models are the fish.

The principal components of the conceptual site models are described below and are intended to be inclusive of the trophic structure that is present at the site and potentially most at risk. Primary and secondary exposure pathways are discussed, as applicable.

Exposure Media

Exposure media are those abiotic media that contain COPCs, which can be potentially transferred to aquatic organisms. These media include: surface water, sediment, and sediment pore water.

Primary Producers

Phytoplankton are primary producers whose principal exposure route is through active or passive uptake of contaminants from surface water. Phytoplankton, in turn, are a food base for both fish and some benthic invertebrates.

Detritus

Detritus is primarily composed of dead organic matter and is rich with detrivores and microbial decomposers. The detrital food chain is important because it is the main base of carbon and energy in river and bay sediments.

Primary Consumers

Primary consumers are those organisms (zooplankton and benthic infauna) that feed directly on the phytoplankton or detritus/organic carbon within the sediment. Principal uptake routes of contaminants include respiration (uptake of dissolved contaminants across respiratory organs such as gills or filaments, and across body walls) from surface water or pore water, ingestion of contaminated food items, and direct ingestion of sediment for benthic infauna. Secondary routes can include ingestion of surface water or pore water, and direct contact with contaminants in sediment.

Secondary Consumers

Secondary consumers are those insects, fish (e.g., carp, perch, alewife) or birds (e.g., tree swallows) that may feed on either the primary producers and/or primary consumers. Some benthic organisms may also be secondary consumers. Principal exposure routes for this group of organisms include respiration and ingestion of contaminated sediment or prey. Secondary exposure may occur through water ingestion and direct contact with contaminants in sediments or surface water.

Higher Consumers

Higher consumers in the Lower Fox River/Green Bay system include carnivorous fish (walleye), piscivorous birds (cormorants, terns), and piscivorous mammals (mink). For carnivorous fish, principal uptake routes are through respiration and ingestion of contaminated prey. Of secondary importance would be ingestion of, and/or direct contact with water or sediment. For piscivorous mammals and birds, the principal exposure route is ingestion of contaminated prey. Of secondary importance for piscivorous mammals would be ingestion of, and/or direct contact with water or sediment.

Top Predators

The top predators in the Lower Fox River/Green Bay system include carnivorous birds (bald eagle). For carnivorous birds, the principal exposure route is ingestion of contaminated prey. Of secondary importance is ingestion of, and/or direct contact with water.

Ecological Receptors

The receptor species selected for the Lower Fox River site are representative species for each assessment endpoint. Receptor species of the Lower Fox River system are based upon the review of habitats and ecological systems defined in Section 2. Effects to receptor species are models for providing an answer to the risk questions posed.

Model receptor species were also selected as representatives of the various identified trophic levels in the conceptual site model. As representatives, the selected species were intended to have the same potential risk as species present in the Lower Fox River and Green Bay system. Receptor species were primarily selected based on an elevated potential for exposure and sensitivity to contaminants of concern. For each species, use and range of habitat types, trophic level, use as prey by predators, species-specific site data, and toxicological data from the literature were all evaluated.

Because of the changes in food web structure in the transition from the Lower Fox River to Green Bay Zone 4, three different conceptual food web models were used. The only differences in these food web models were the fish receptor species selected. These conceptual models assumed that invertebrates, tree swallow, Forster's tern, common tern, double-crested cormorant, bald eagle, and mink are common receptors to all areas. Selected fish receptor species for evaluating potential exposure between Lake Winnebago and the De Pere dam (conceptual model Figure 6-1) include gizzard shad, shiner species, yellow perch, carp, and walleye. These same species were also used as receptors in Green Bay Zone 1 (De Pere to Green Bay) and Green Bay Zone 2 (conceptual model Figure 6-2).

Additionally, this second food web model included rainbow smelt and alewife as receptors. Finally, a third food web model (conceptual model Figure 6-3) was created for the northern area of Green Bay—zones 3 and 4. Here, receptor species included gizzard shad, rainbow smelt, alewife, brown trout, and walleye. Gizzard shad and walleye were fish receptors evaluated in all three models, and rainbow smelt and alewife were evaluated in all zones of Green Bay. The life stages and habitat use for each selected receptor species are described below including, when possible, site-specific information.

Receptor species and abundance of these species found within the Lower Fox River and Green Bay have changed over time as discussed in Section 2 (the RI Summary), and it can be expected that components of the ecosystem will continue to change. Fluctuations in population levels of many of the receptor species are discussed below, but evaluation of the receptor's current population, will not be considered solely as a measure of risk. While population level endpoints can be an appropriate tool to assess risk, the population data discussed below were not collected specifically for risk assessment and must be viewed within the context of other confounding environmental factors. These can include such things as immigration, emigration, food availability, habitat suitability and availability, species competition, predation, and weather. Confounding environmental factors are not evaluated as part of this risk assessment.

Identified receptor species in the conceptual model were profiled in detail in Section 2.4. The importance of the selected receptor species and their diets are briefly resummarized here.

Invertebrates

Both aquatic and benthic invertebrates accumulate and transfer contaminants up the food chain. As indicated, phytoplankton are exposed to chemicals through the water column. Consumers of phytoplankton in the Lower Fox River/Green Bay system include zooplankton, gizzard shad, and shiner species.

Benthic invertebrates are in direct contact with sediments and pore waters, are generally stationary, are known to accumulate contaminants, and are an important food base for higher-level trophic organisms, particularly fish. Benthic invertebrates, through bioturbation, release nutrients contained in the interstitial water into the water column (Krieger, 1992). Oligochaetes and chironomids are the predominant invertebrates found in all reaches of the Lower Fox River. These kinds of invertebrates are particularly important as decomposers (Krieger, 1992).

Overall, investigations of the Lower Fox River and southern Green Bay benthic communities determined that there was low taxa richness (number of taxa at each

sampling location), and low community diversity (the number of species relative to the abundance of species) (IPS, 1993a, 1993b; WDNR, 1996a, 1996b).

Species of oligochaetes (worms) generally feed on dead organic matter including fine detritus, algae, and other microorganisms. The primary food for chironomids is planktonic algae and detritus. Chironomids and oligochaetes are normally found in greatest abundance in soft sediment deposits in pools, runs of streams, profundal areas and littoral areas of lakes with soft bottoms, and harbor or bay areas where stream-transported sediments have been deposited. River rock and riffle areas are not preferred habitat. Invertebrate sensitivity to contaminants varies widely with each species and can range from very sensitive (e.g., mayflies) to relatively insensitive (e.g., chironomids and worms).

Fish

Through the mid-1970s the population levels of fish species, such as walleye and perch, were low within the Lower Fox River and southern Green Bay ecosystems. Contaminants along with low dissolved oxygen conditions brought about by uncontrolled and untreated wastewater dumped into the river were believed to be a contributing factor causing low population levels. Principal species found within the system were those that could tolerate these conditions, especially bullhead and carp.

With the institution of water quality controls in the mid-1970s, contaminants and dissolved oxygen conditions improved and the WDNR undertook a program to reintroduce walleye into the last reach of the river (De Pere to Green Bay) and in Green Bay through a stocking program beginning in 1973. That program was wholly successful; self-sustaining populations of walleye now exist within the river below the De Pere dam and in the bay.

In addition to walleye, a number of other species became reestablished in the Lower Fox River and Green Bay, including yellow perch, alewife, shad, bass, and other species. Historical anecdotal data from the Oneida tribe and more recent creel survey data from the WDNR indicate that Duck Creek and Suamico tributaries to southern Green Bay were used by numerous fish species (Nelson, 1998).

As previously discussed, Green Bay zones 2 and 4 are known to be quite different in terms of their physical characteristics, which affects species distribution and trophic complexity. Green Bay Zone 2 is hypereutrophic (warm and highly productive), while Zone 4 is meso-oligotrophic (cooler and less productive). Related distinguishing characteristics of Zone 4 are that there are lower population densities of fish, less trophic complexity, clearer water, and less human

development as compared to Zone 2 (Brazner and Beals, 1997; Sager and Richman, 1991).

A detailed discussion of fish species, life histories, and migration patterns was presented in Section 2.4. The remainder of this section summarizes receptor species feeding and prey status of the food web.

Rainbow Smelt. Rainbow smelt (*Osmerus mordax*) are widespread and abundant non-indigenous pelagic planktivores in the Great Lakes (Jones *et al.*, 1995). Investigations have indicated that rainbow smelt dominantly feed on zooplankton as juveniles and adults (Mills *et al.*, 1995; Urban and Brandt, 1993). However, as adults, rainbow smelt will also feed on other fish, particularly when they are in inshore waters in the spring or during the fall when they share the same water depth as young-of-the-year alewife (Becker, 1983; O'Gorman, 1974). In addition to consuming alewife, rainbow smelt have been shown to consume their own species as well as other fish including emerald shiner, yellow perch, sculpin, burbot, and rock bass (Becker, 1983; Brandt and Madon, 1986; O'Gorman, 1974; Selgeby *et al.*, 1978; Stedman and Argyle, 1985). In fact, fish can constitute up to 98 percent of their diet, as was measured in Crystal Lake, Michigan.

Rainbow smelt in turn can be prey for larger fish (walleye, perch, trout) as well as piscivorous birds such as cormorants and terns.

Alewife. Alewife (*Alosa pseudoharengus*) are non-indigenous small anadromous pelagic planktivores that prefer open water and sandy habitats. Their distribution is limited to Green Bay zones. The De Pere dam represents a physical barrier to upstream migration. Both juvenile and adult alewife dominantly consume zooplankton; however, while young-of-the-year alewife only consume zooplankton (Urban and Brandt, 1993), adult alewife may also consume amphipods, chironomids, fish eggs, and larvae (Hewett and Stewart, 1989). Predation by alewives on native fish eggs and larvae may be contributing substantially to the decline of native fish species (Hewett and Stewart, 1989). In Lake Michigan, yellow perch year-class strength has been inversely related to abundance of alewife (Brandt *et al.*, 1987; Mason and Brandt, 1996) and alewife have also been implicated as a principal factor in the failure of lake trout populations (Holey *et al.*, 1995; Jones *et al.*, 1995). Becker (1983) indicated that prey selection may be dependent on depth; filamentous algae represent 50 percent of the stomach contents of fish taken in the shore zone of Green Bay and zooplankton represent the other 50 percent, while deepwater amphipods dominate the diet when alewife feed in water depths of 9 to more than 30 meters.

Alewife are in turn consumed by walleye, perch, and trout, as well as cormorants and terns.

Gizzard Shad. Gizzard shad (*Dorosoma cepedianum*) is an abundant omnivore in many central and southern United States lakes (Shepherd and Mills, 1996). Juveniles, up to about 30 mm, are visual particulate feeders that dominantly consume zooplankton and can influence zooplankton populations both directly and indirectly through consuming phytoplankton (Roseman, 1996). Juvenile gizzard shad are important forage fish for predators such as walleye in Green Bay (Wolfert and Bur, 1992; Becker, 1983). Peak consumption of gizzard shad occurs when they are approximately 25 mm; by the fall, gizzard shad may measure 90 mm and are too big for some predators to consume (Michaletz, 1997). The rapid growth of these fish, more than the population density, limits predator consumption, at least for small predators (Michaletz, 1997).

Shiner Species. Shiner species considered as receptors in the Lower Fox River and Green Bay include golden shiner (*Notemigonus crysoleucas*), emerald shiner (*Notropis atherinoides*), and common shiner (*Notropis cornutus*). While zooplankton are the dominant prey of juvenile and adult golden and emerald shiners, common shiners ingest approximately equal amounts of animal and plant material and generally less zooplankton (Becker, 1983). Other prey consumed by shiners include algae, terrestrial insects, small fish, fish eggs, aquatic insects, oligochaetes, amphipods, molluscs, plants, and detritus (Becker, 1983).

Yellow Perch. Yellow perch (*Perca flavescens*) are native to Green Bay and have been a popular commercial and recreational catch species. Perch are a schooling species that feed during the day and rest on the bottom at night. Prey consumed by yellow perch varies seasonally. Juveniles mostly consume zooplankton and chironomids. Feeding preferences of perch change with age and season; after the first year, yellow perch become increasingly piscivorous. As young fish, perch primarily feed on zooplankton and switch to a diet of benthic invertebrates, eggs, and young fish as they age (Scott and Crossman, 1973). However, zooplankton are an important food source for all sizes of yellow perch (Becker, 1983). Although yellow perch will eat fish, chironomid larvae are the predominant food until 180 mm, and it is only after this size (around age 3 or 4) that fish become a major food item (Becker, 1983). In turn, perch are prey for several fish and bird species (Scott and Crossman, 1973).

Carp. Carp (*Cyprinus carpio*) is an abundant bottom-dwelling species found in southern Green Bay. Carp are uniquely equipped with the ability to sense food taste and can distinguish salty, sweet, bitter, and acid stimuli (Becker, 1983). Young-of-the-year carp eat copepods, chironomids, and cladocerans. On a weight basis,

chironomids are the dominant food source (Weber and Otis, 1984). The adult diet is generally half plant and half animal material that is taken from sediments and occasionally from the surface. Animal prey can include aquatic insects, crustaceans, annelids, and molluscs. Fish or fish eggs may be taken, but they are not a significant item in the carp diet (Becker, 1983). Plant material consumed can include green tree leaves, grasses, twigs, roots, aquatic plants, and algae (Becker, 1983). Animal material is particularly important in the winter when plant material may not be available (Becker, 1983).

Brown Trout. Brown trout (*Salmo trutta*) is a popular, seasonally-caught game fish in Green Bay. Brown trout tend to be nocturnal feeders, and food items can include aquatic and terrestrial insects, crustaceans, molluscs, frogs, shrimp, salamanders, and other fish. Zooplankton are an important food source for small brown trout (Becker, 1983). When up to about 229 mm in length, they are insect feeders, but beyond this length they dominantly consume (70 percent of the diet) fish such as young trout, sculpins, minnows, darters, and lampreys (Becker, 1983). Magnuson and Smith (1987) found that brown trout collected in the spring from Green Bay Zone 3 dominantly consumed alewife (73 percent of the diet); rainbow smelt were the other 27 percent of the identified forage fish consumed. Half of the brown trout collected in the fall in this region of the bay had empty stomachs and, therefore, prey consumption was not evaluated (Magnuson and Smith, 1987). Presumably, this was about the same time as their spawning. It is suspected that over the summer, brown trout, like walleye, increase their consumption of rainbow smelt (Magnuson and Smith, 1987).

Walleye. Walleye (*Stizostedion vitreum*) is a popular, year-round game and commercial fish found in Lake Michigan, generally in areas less than 7 meters deep (Magnuson and Smith, 1987). Walleye diet is seasonally dependent. Feeding is generally greatest in the summer and early fall when forage fish are abundant; however, feeding is infrequent when water temperatures are below 15 ˚C. Young-of-the-year walleye are believed to eat mainly phytoplankton, including diatoms and blue-green algae. At approximately 30 mm in length, young walleye begin to feed on fish, including alewife and yellow perch. For older walleye, fish dominate the diet except during times when prey fish are less abundant, in which case walleye will feed on benthic invertebrates. For both young-of-the-year walleye and older walleye, prey is selected that is less that 90 mm total length (Wolfert and Bur 1992; Knight *et al.*, 1984).

Walleye diets were investigated in spring and fall in three areas of the Lower Fox River and southern Green Bay system: just below the De Pere dam, at the mouth of the Lower Fox River (fall only), and in Green Bay Zone 3 (Magnuson and Smith, 1987). Walleye collected at the mouth of the Lower Fox River were the

only fish to contain all three major forage fish—alewives, rainbow smelt, and gizzard shad. Invertebrates were only prey for fish collected just below De Pere dam and were consumed in both the spring and fall. Like the diet of the fish collected at the river mouth, walleye collected from Green Bay Zone 3 consumed only fish (alewife and rainbow smelt).

Birds

Tree Swallow. Tree swallows (*Tachycineta bicolor*) are migratory songbirds that breed in and migrate through the Lower Fox River region. Tree swallows nest in semi-colonial groups in natural cavities (trees, posts, streambanks) near water. Tree swallows feed exclusively on insects, predominately aquatic insects.

Tree swallow population data are not available from the Lower Fox River and Green Bay because studies of these birds in this region have used artificial nest boxes rather than relying on naturally nesting populations (Ankley *et al.*, 1993; Custer *et al.*, 1998).

Forster's and Common Tern. Forster's tern (*Sterna forsteri*) and common tern (*Sterna hirundo*) are migratory species of colonial waterbirds that breed in the Great Lakes and generally winter in more southern coastal areas. These species are listed by the WDNR as endangered, as is the Caspian tern (*Sterna caspia*). Around the Green Bay area, nesting Forster's terns have been reported since the late 1930s, although they were likely nesting without record prior to this period. The Forster's tern preferred habitat is around wetlands where they feed mainly on small fish (alewife, emerald shiner, and rainbow smelt) and on some aquatic invertebrates.

Double-crested Cormorants. Double-crested cormorants (*Phalacrocorax auritus*) are a migratory species of colonial waterbird that breed in the Great Lakes and generally winter in coastal areas, including Alaska. These birds nest in large communities in a variety of habitats including cliffs, grassy slopes, low bushes, or dead trees. Historically, the double-crested cormorant population in the Great Lakes region experienced large population declines, beginning in the 1950s and continuing through the 1970s, largely from the presence of contaminants. More recently, populations of double-crested cormorants in the Great Lakes region have greatly increased (Weseloh *et al.*, 1994).

Cormorants consume approximately 25 percent of their body weight each day and on average weigh 1.9 kg. The primary food consumed is small fish such as rainbow smelt and alewife and, as available, perch.

In 1972, the double-crested cormorant was listed as a Wisconsin state endangered species due to the lack of nesting pairs of birds in the state. Prior to 1979, inland breeding populations exceeded the number of nesting birds on the Great Lakes. By 1986, populations in the state increased such that the double-crested cormorant was removed from the Wisconsin state endangered species list. Since 1990, the Great Lakes population of double-crested cormorants has exceeded the inland population levels by approximately five times (Matteson, 1998). The nesting population in the Green Bay and Lake Michigan region, as of 1997, accounted for 81 percent of the total breeding population. The largest colonies were found in the following four locations: Spider Island, Cat Island, Hat Island, and Jack Island as indicated. Of these islands, Cat Island is located closest to the mouth of the Fox River and contains the second highest density of double-crested cormorants.

Bald Eagles. Bald eagle (*Haliaeetus leucocephalus*) preferred habitat has a large water-to-land edge area and large areas with an unimpeded view (Palmer, 1988). Eagles are not generally found in areas of high human use (EPA, 1993a). Within the Great Lakes area, some eagles are present on a year-round basis, while others are transient and winter in more southern locations (Palmer, 1988). The Green Bay region contains the largest number of nesting eagles in the U.S., excluding Alaska (Palmer, 1988). Nesting locations within the Fox River and Green Bay are shown on Figures 2-21 and 2-24.

For feeding, eagles often follow each other or other species in search of prey (Palmer, 1988). The estimated prey weight limit is 5 pounds (Palmer, 1988). The majority of the bald eagle diet is fish, but can also include waterfowl or other birds, or mammals. Diet selection depends on the abundance of prey because bald eagles are opportunistic (EPA, 1993a). Prey can either be taken live or as carrion (Palmer, 1988).

Mammals

Mink. Mink (*Mustela vison*), high trophic level, opportunistic carnivores, are found throughout the United States. They are found near all types of aquatic habitats, preferring protected wetlands with ample vegetative cover, as opposed to open water sites. The presence of mink around the Lower Fox River system has not been confirmed, although adequate habitat is present (Patnode, 1998). Limited presence may be due to the mink's sensitivity to PCBs (Tillitt *et al.*, 1996). Their prey can include birds, small mammals, crustaceans, amphibians, and fish. In Michigan, fish is a dominant portion of the mink diet particularly in size classes of 5 to 18 cm (Alexander, 1977).

The size of home range areas is dependent on both the availability of vegetative cover and prey. Seasonal changes may affect availability of prey and, therefore, habitat use (Linscombe *et al.*, 1982). In the winter, habitat use is restricted. Home ranges for adult males have been measured at 1,800 to 5,000 square meters with an average of 2,630 square meters, while home range size for juvenile males is half that size (Linscombe *et al.*, 1982). Female home ranges have been measured at 1,000 to 2,800 square meters with an average of 1,850 square meters (Linscombe *et al.*, 1982).

Mink dens have been found between 5 and 100 meters of the water and mink have not been witnessed more than 200 meters from the water (EPA, 1993a). Dens are cavities supported by rocks or tree roots that are above the waterline. The availability of suitable dens can limit the number of mink in an area. Mink reach sexual maturity at 10 months of age and they can reproduce for 7 years (EPA, 1993a).

Exposure Routes

Exposure routes define how ecological receptors are potentially exposed to COPCs. This section describes the potential exposure routes for the ecological receptors identified previously. Exposure routes involve either water, sediment, or trophic transfer through food consumption. These exposure routes are described below and summarized in Table 6-3.

Water. Phytoplankton and zooplankton are exposed to dissolved contaminants from the water column through respiration, ingestion, and direct contact. Zooplankton may ingest particulate-based contaminants during feeding. Benthic invertebrates also can experience contaminant transfer through direct contact with pore waters. MacDonald (1993) examined the distribution of PCB congeners in seven lake systems, and the influences of sediment/biota partitioning and food web transport. Results of this study indicated that food web transport has a greater impact than sediment/biota partitioning in higher trophic levels. Exposure to contaminants through gill respiration is a primary exposure route for fish. Direct contact with water can also result in chemical exposure in fish, although this is a secondary exposure route. Wildlife can also be exposed to contaminants through ingestion of water.

Sediment. Primary exposure routes for benthic invertebrates are through direct contact with sediments and ingestion of sediments. For benthic fish and carnivorous birds and mammals, incidental ingestion of sediments can occur during feeding on fish or benthic invertebrates.

Food. Transfer of contaminants through food consumption is a primary exposure route for chemicals that bioconcentrate in tissues and particularly those that biomagnify. All of the COPCs have the potential to biomagnify up the food chain except for lead and arsenic which can, however, bioconcentrate. Therefore, lower trophic level organisms accumulate less contaminants in their tissues from food consumption than top predators.

6.2.5 Food Chain Model

Food chain modeling will be used for the assessment of risks to piscivorous birds and mammals because of the limited tissue data available. While there are tissue concentrations available for some of the identified bird receptors, not all river reaches and Green Bay zones have corresponding tissue data. For the purposes of assessing risk, exposure was modeled for all piscivores in all river reaches and Green Bay zones, even for those zones where tissue data were available.

Dietary exposure for insectivorous birds were not evaluated for any river reach or bay zone because this exposure pathway was presumed to result in minimal risk to insectivores. This decision was largely based on findings of risk to tree swallows presented in the *Draft Baseline Human Health and Ecological Risk Assessment* (ThermoRetec, 1999). The principal study that guided this decision was that of Custer *et al.* (1998). These data, collected in 1994 and 1995, are the most recent that have been collected for insectivorous birds from the Lower Fox River and Green Bay. Eggs were collected and analyzed from nest boxes located in Little Lake Butte des Morts and Green Bay Zone 2 (Kidney Island). Contaminants measured in eggs from these two stations were compared to contaminant levels measured in eggs from two reference stations. Only two organochlorines were detected in the birds, p,p'-DDE and total PCBs. Pipers (newly-hatched young of 1 day or less in age) contained mean p,p'-DDE concentrations of 0.19 mg/kg (Little Lake Butte des Morts) and 0.20 mg/kg (Green Bay Zone 2) which were not significantly different from the mean concentration at one of the reference stations (0.15 mg/kg), but they were significantly different from the mean concentration at the second reference station (0.11 mg/kg). Mean total PCB concentrations in pipers were 3.10 mg/kg (Little Lake Butte des Morts), 3.24 mg/kg (Green Bay Zone 2), 0.77 mg/kg (reference station 1), and 0.29 mg/kg (reference station 2). Both Little Lake Butte des Morts and Green Bay Zone 2 piper concentrations of total PCBs were not significantly different from each other, but were significantly different from the concentrations measured in the reference stations. The only significant differences found for the endpoints of clutch size, clutch success, and egg success were between Green Bay Zone 2 and Little Lake Butte des Morts where clutch success was higher in Green Bay Zone 2 (Custer *et al.*, 1998).

For comparison of exposures between reaches and zones, both piscivorous bird and piscivorous mammal concentrations were modeled for all areas; measured COPC exposures are reported as mg/kg in the tissues, however, modeled exposures are reported in units of mg/kg-BW/day.

Only those contaminants that potentially bioaccumulate in the food chain (PCBs, DDT and metabolites, dieldrin, and mercury) will be modeled to piscivorous bird and mammal receptors.

Modeled exposure is based on examining the daily consumption of COPCs from all dietary sources—the ***oral dose*** approach (EPA, 1993a). This oral dose approach principally considers COPC intake through prey items, water, and sediment. Exposures via inhalation and dermal contact are assumed to be minimal and are not evaluated.

The structure of the food web model is as follows:

$$ED_T = \sum \frac{\left(C_x \times I_x\right)}{bw}$$

where:

ED_T = estimated daily dose (mg/kg-BW/day ww)
C_x = concentration of the COPC in medium x (mg/kg ww)
I_x = rate of ingestion of medium (mg or kg/day ww)
bw = body weight (kg)

Two general assumptions of this model are that the COPC in the medium is 100 percent available and that the area from which the COPC concentrations were obtained are completely inside the relevant forage area of the receptor.

Water and food ingestion were considered as possible sources of contaminant exposure for both bird and mammal receptors. Sediment ingestion was only considered a potential exposure pathway for mammals because birds are assumed to not come into contact with sediment. Birds were assumed to only take fish from near the water surface. Where COPCs were not detected in media (e.g., prey, sediment, and water) it was assumed that COPCs concentrations were at half the mean detection limit.

Exposure modeling input values (body weight, and ingestion rates for food, water, and sediment) were selected based on a review of values presented in:

- *Great Lakes Water Quality Initiative (GLWQI) Technical Support Document for Wildlife Criteria* (EPA, 1995d);

- *Trophic Level and Exposure Analyses for Selected Piscivores Birds and Mammals, Volume I: Analyses of Species in the Great Lakes Basin* (EPA, 1995e); and

- EPA *Wildlife Exposure Factors Handbook* (EPA, 1993a).

Selected input values for piscivorous bird and mammals are separately described below, and are presented in Table 6-4.

Piscivorous Bird Model

Exposure model input values were selected for the four piscivorous birds for which exposure was modeled: common tern, Forster's tern, double-crested cormorant, and bald eagle. A common assumption for all four species was that prey were caught live from close to the water surface and, therefore, these birds were not incidentally ingesting any sediment. The exposure modeling for piscivorous birds in the ecological risk assessment of the Sheboygan River also assumed that ingestion of sediment was not a route of exposure (EVS, 1998). Therefore, input parameters were limited to body weight, food ingestion rate, and water ingestion rate. Derivation of these input parameters is described below.

Body Weight. All body weights selected for birds came from the *Trophic Level and Exposure Analyses for Selected Piscivores Birds and Mammals, Volume I: Analyses of Species in the Great Lakes Basin* (EPA, 1995e) which were summarized in the *Great Lakes Water Quality Initiative (GLWQI) Technical Support Document for Wildlife Criteria* (EPA, 1995d). These body weights were the calculated average of adult male and female birds. As presented in Table 6-4, the body weights selected for each species are 120 grams (common tern), 158 grams (Forster's tern), 1,680 grams (double-crested cormorant), and 4,650 grams (bald eagle).

Water Ingestion Rate. Water ingestion rates for all piscivorous bird species were calculated using the allometric equation developed by Calder and Braun (1983), for all birds, which is presented in the EPA *Wildlife Exposure Factors Handbook* (EPA, 1993a). This equation is presented below, and only depends on knowledge of the birds' body weight.

$$Water\ Intake\ (L/day) = 0.059 * body\ weight\ (kg)^{0.67}$$

As presented in Table 6-4, the calculated water ingestion rates for each species are 0.014 L/day (common tern), 0.017 L/day (Forster's tern), 0.084 L/day (double-crested cormorant), and 0.165 L/day (bald eagle).

Food Ingestion Rate. ***Bald Eagles.*** As reported in the *Great Lakes Water Quality Initiative (GLWQI) Technical Support Document for Wildlife Criteria* (EPA, 1995d), Stalmaster and Gessaman (1984) observed bald eagles feeding in Washington state on pre-weighted salmon that were provided to them. Their study indicated that adult bald eagles consume 552 grams of fish per day, and that juveniles and subadults consume 410 to 549 grams of fish per day.

Feeding studies, such as Stalmaster and Gessaman (1984), can be used to estimate the overall food ingestion rate through a series of metabolic equations, as described in the EPA *Wildlife Exposure Factors Handbook* (EPA, 1993a). The EPA Handbook lists the gross energy (*GE-kcal/g*) of numerous aquatic and terrestrial prey, and also lists the assimilation efficiency (*AE*-%) of prey consumed by birds and mammals. Metabolizable energy (*ME-kcal/g prey*), which is the energy that can be used for growth and reproduction, is the product of the gross energy and the assimilation efficiency as indicated by the equation below.

$$ME\left(kcal\ /\ g\ \ prey\right) = GE\left(kcal\ /\ g\right) * AE\left(\%\right)$$

The *ME* is specific for both the type of prey (in this case, fish) consumed and the type of predator (in this case, birds). Therefore, the *ME* of birds consuming fish is 0.948 kcal/g fish, based on a *GE* of 1.2 kcal/g for bony fish, and a *AE* of 79 percent for birds that consume fish. The total *ME* requirement, also known as the free-living metabolic rate (*FMR*), of the predator (kcal/day) is calculated by multiplying the *ME* of the prey (*kcal/g prey*) by the daily food ingestion rate (*g prey/day*).

$$FMR\ \left(kcal/day\right) = ME\ \left(kcal/g\right)\ prey * Food\ Ingestion\ Rate\ \left(g/day\right)$$

Based on the data collected by Stalmaster and Gessaman (1984) on adult bald eagles, the *FMR* required was 523 kcal/day: 552 g fish/day multiplied by 0.948 kcal/g fish. Craig *et al.* (1988), using the model of Stalmaster and Gessaman (1984), estimated the *FMR* required for bald eagles in Connecticut. Their results indicated that adult bald eagles require 448 kcal/day. For the calculation of wildlife criteria for the Great Lakes, based on these two studies (Stalmaster and Gessaman, 1984 and Craig *et al.*, 1988), the EPA assumed that bald eagles required an *FMR* of 500 kcal/day (EPA, 1995a).

Assuming an *FMR* of 500 kcal/day, if bald eagles consumed only fish, each individual would need to consume 527 grams of fish per day (500 kcal/day divided by 0.948 kcal/g fish). Although the calculation of wildlife criteria for the Great Lakes included the consumption of birds (8% of the diet) and fish (92% of the diet), the Lower Fox River and Green Bay exposure model for bald eagles assumed that they consumed only fish (100% of the diet). Bowerman (1993) in his research of bald eagles from the Great Lakes found that 93 percent of the diet of bald eagles is comprised of fish. Therefore, it was assumed that bald eagles in the Lower Fox River and Green Bay consumed 527 grams of fish per day: 80 percent being trophic level 3 fish (422 grams) and 20 percent being trophic level 4 fish (105 grams). These percentages consumed of trophic level 3 and 4 fish came from the *Great Lakes Water Quality Initiative (GLWQI) Technical Support Document for Wildlife Criteria* (EPA, 1995a).

Double-crested Cormorant. Feeding studies, like those conducted for the bald eagle, have not been conducted for the double-crested cormorant. Therefore, the *FMR* for this species was estimated using the equation for all birds derived by Nagy (1987) and presented in the EPA *Wildlife Exposure Factors Handbook* (EPA, 1993a). This equation is presented below.

$$FMR\left(kcal/day\right) = 2.601 * \ body\ weight\left(g\right)^{0.640}$$

Assuming a body weight of 1,680 grams, the *FMR* equals 301.54 kcal/day. Further assuming that these birds consume only fish, the mass of fish required per bird per day can be calculated by dividing the *FMR* by the metabolizable energy (*ME*) of fish consumed by birds. This *ME* of 0.948 kcal/g fish was calculated in the previous section describing the food ingestion rate of bald eagles. Dividing the *FMR* by the *ME* indicates that double-crested cormorants need to consume 318 grams of fish per day.

Common Tern. Feeding studies, like those conducted for the bald eagle, have not been conducted for the common tern. Therefore, the *FMR* for this species was calculated using the equation for seabirds derived by Nagy (1987) and presented in the EPA *Wildlife Exposure Factors Handbook* (EPA, 1993a). This equation is presented below.

$$FMR\left(kcal/day\right) = 1.916 * \ body\ weight\left(g\right)^{0.704}$$

Assuming a body weight of 120 grams, the *FMR* equals 55.74 kcal/day. Further assuming that these birds consume only fish, the mass of fish required per bird per day can be calculated by dividing the *FMR* by the metabolizable energy of fish

consumed by birds. This *ME* of 0.948 kcal/g fish was calculated in the previous section describing the food ingestion rate of bald eagles. Dividing the *FMR* by the *ME* indicates that common terns need to consume 58.8 grams of fish per day.

Forster's Tern. Feeding studies, like those conducted for the bald eagle, have not been conducted for the Forster's tern. Therefore, the *FMR* for this species was calculated using the equation for seabirds derived by Nagy (1987) and presented in the EPA *Wildlife Exposure Factors Handbook* (EPA, 1993a). This equation is presented below.

$$FMR\left(kcal/day\right) = 1.916 * body\ weight\left(g\right)^{0.704}$$

Assuming a body weight of 158 grams, the *FMR* equals 67.65 kcal/day. Further assuming that these birds consume only fish, the mass of fish required per bird per day can be calculated by dividing the *FMR* by the metabolizable energy of fish consumed by birds. This *ME* of 0.948 kcal/g fish was calculated in the previous section describing the food ingestion rate of bald eagles. Dividing the *FMR* by the *ME* indicates that Forster's terns need to consume 71.4 grams of fish per day.

Piscivorous Mammal Model

The only piscivorous mammal selected for exposure modeling is mink. It was assumed that mink consume water, sediment, and fish. Input parameters selected included body weight, food ingestion rate, water ingestion rate, and sediment ingestion rate. Derivation of these input parameters is described below and final values are presented in Table 6-4. It should be noted that the input parameters selected and described below were not the same as the input parameters selected for modeling exposure to mink in the EPA Upper Green Bay Risk Assessment (Appendix C).

Body Weight. The body weight selected for mink came from the *Trophic Level and Exposure Analyses for Selected Piscivorous Birds and Mammals, Volume I: Analyses of Species in the Great Lakes Basin* (EPA, 1995b) and this body weight was also used for the derivation of wildlife criteria for the Great Lakes (EPA, 1995a). This body weight of 800 grams represents an average weight of adult males and females.

Water Ingestion Rate. The water ingestion rate for mink was calculated using the allometric equation developed by Calder and Braun (1983) for all mammals, which is presented in the EPA *Wildlife Exposure Factors Handbook* (EPA, 1993a). This equation is presented below, and only depends on knowledge of the mammal's body weight.

$$Water\ Intake\ (L/day) = 0.099 * body\ weight\ (kg)^{0.90}$$

As presented in Table 6-4, the calculated water ingestion rate for mink is 0.081 L/day.

Sediment Ingestion Rate. The EPA *Wildlife Exposure Factors Handbook* does not contain sediment ingestion rates for mink, but it does present estimated soil ingestion rates for other mammals. Expressed as a percent of the total diet, these soil ingestion rates are: 2.8 percent (red fox), 9.4 percent (racoon), and 2.4 percent (meadow vole) (EPA, 1993a). Alexander (1977) examined the diet of mink, and although they also did not publish an estimate of sediment ingestion, they did report that vegetation represented approximately 2 percent of the total diet. For the purposes of exposure modeling for piscivorous mammals in the Lower Fox River, it was assumed that sediment represents 2 percent of the mink diet. This rate of sediment intake was also used in the exposure modeling for mink on the Sheboygan River (EVS, 1998). Assuming that mink ingest food at a rate of 179.9 g/day (described below), estimated sediment ingestion is 4 g/day.

Food Ingestion Rate. The food ingestion rate for mink was calculated in the same way that the food ingestion rates for birds were calculated. As previously indicated, the EPA Handbook lists the gross energy (*GE-kcal/g*) of numerous aquatic and terrestrial prey, and also lists the assimilation efficiency (*AE*-%) of prey consumed by birds and mammals. The *GE* of fish (1.2 kcal/g) is the same whether it is birds or mink that is consuming them. The *AE* for mammals consuming fish is 91 percent. Therefore, the metabolizable energy (*ME-kcal/g prey*), which is the energy that can be used for growth and reproduction, is 1.09 kcal/g prey for piscivorous mammals.

The *FMR* for mink was calculated based on the equation developed by Nagy (1987) for non-herbivorous mammals, which is presented in the EPA *Wildlife Exposure Factors Handbook* (EPA, 1993a). This equation is presented below.

$$FMR\ (kcal/day) = 0.6167 * body\ weight\ (g)^{0.862}$$

Assuming a body weight of 800 grams, the *FMR* equals 196.13 kcal/day. The food ingestion rate was calculated based on the equation below.

$$Food\ Ingestion\ Rate\ (g/day) = FMR\ (kcal/day) \div ME\ (kcal/g\ prey)$$

Assuming that mink consume only fish, the food ingestion rate is 179.9 g/day (196.13 kcal/day divided by 1.09 kcal/g). However, the mink food consumption study by Alexander (1977), indicated that during the spring, winter, and fall, fish represent approximately 85 percent of the mink diet. During the summer months, fish consumption may decrease as the fish move towards deeper and cooler waters. For the purposes of exposure modeling for mink in the Lower Fox River and Green Bay, it was assumed that the mink consumed 85 percent fish in their diet. Therefore, of the total food intake rate of 179.9 g/day, 153 g/day (85 percent) represents the fish intake rate. Other dietary items can include crustacea (e.g., crayfish), small mammals (e.g., muskrat) or bird eggs, but these are not included in the exposure model both because of the limited data and the small proportion these items represent in the mink diet.

Fish Species Selected for Modeling

Fish species selected for modeling of piscivorous bird concentrations were determined from: preferred prey noted in the literature, fish species used in dietary toxicity testing, and available data within the FRDB. While the specific species selected for food web modeling are noted in the tables of estimated exposure concentrations for piscivorous birds and mammals, the general rationale for the species selected is given below.

Terns and cormorants rely upon small forage fish as a primary food source; principally alewife and rainbow smelt (Matteson, 1988; Mossman, 1988; Environment Canada, 1998). Alewife, therefore, were selected as the preferred trophic level 3 prey for these species, but alewife are excluded from the Lower Fox River above the De Pere dam. When alewife data were not available, then another small forage fish was selected (e.g., gizzard shad or rainbow smelt), and if small forage fish data were also not available, then yellow perch data were used. While yellow perch are not a preferred prey item, they do represent a trophic level 3 pelagial fish.

Bald eagles consume both trophic 3 level and trophic 4 level fish. In a study of bald eagle fish consumption in the Great Lakes, it was found that almost 50 percent of their diet is suckers, a species similar to carp (Bowerman, 1993). Of the whole fish data available, carp (trophic level 3) and walleye (trophic level 4) best represent potential prey for bald eagles. Generally, data for these fish was widely available for all Lower Fox River reaches and Green Bay zones.

Most of the dietary laboratory studies of contaminant toxicity to mink involve feeding mink contaminated carp. Therefore, since this species was widely represented in the FRDB it was selected a the sole prey of mink.

6.3 Characterization of Ecological Effects

Described in this section are known biological effects of each COPC to identified receptors, if available, or a similar species. As discussed in Section 6.2.4, effects of arsenic and lead to birds and mammals will not be evaluated because modeling was not possible given the lack of data and, importantly, the lack of detection of these compounds where data were available.

As part of this section, toxicity reference values (TRVs) are selected that will be used as part of the risk analysis. TRVs may be based on data from laboratory toxicological evaluations or field toxicological studies. The studies which were used to derive TRVs for this risk assessment are described in each individual COPC section below and a summary of the TRVs selected is presented in Table 6-5. Unless otherwise indicated, all concentrations are reported on a wet weight (ww) basis.

6.3.1 Data Sources Reviewed

In order to derive TRVs, a comprehensive literature search was performed. A variety of databases were searched for literature references containing toxicological information. Some of these literature sources included Biological Abstracts, Applied Ecology Abstracts, Chemical Abstract Services, Medline, Toxline, BIOSIS, ENVIROLINE, Current Contents, Integrated Risk Information System (IRIS), the Aquatic Information Retrieval Database (AQUIRE) maintained by the EPA, and the Environmental Residue Effects Database (ERED) maintained by the EPA and U.S. Army Corps of Engineers.

In addition, a number of secondary literature sources provided summaries or reviews of the toxicological literature related to a variety of contaminants. These documents were not used directly to derive TRVs because they do not capture the details of the toxicological methods which are important to the selection of technically defensible TRVs. However, these summary documents provided an excellent means of locating original studies that may have been overlooked in the database searches. Examples of such summary documents include Agency for Toxic Substances Disease Registry (ATSDR) documents, U.S. Fish and Wildlife Service Contaminant Hazard Reviews, EPA Great Lakes Water Quality Initiative documents, and EPA Ambient Water Quality Criteria documents.

The TRVs selected for this assessment were discussed with and agreed upon by BTAG members. Importantly, the consensus on the TRVs are for site-specific use only and are not intended to be used at other sites.

Evaluation Criteria

All studies were evaluated for the appropriateness of their use in deriving TRVs. A number of criteria were considered when evaluating the appropriateness of using a particular study for deriving a TRV. The two most important considerations were the suitability of the test result for evaluating the assessment endpoint, and the likelihood that a similar result would be obtained if the test were repeated. A number of additional criteria were also considered. For example, studies were selected in which the test organism was similar to the receptor species. Doses had to be quantified, and effects measured and reported. The exposure duration was preferably either chronic, subchronic, or involved a sensitive life stage; multigenerational studies were also deemed appropriate. Sample sizes had to be adequate, preferably including at least three treatment groups in addition to any control groups. At the very least, a negative control should have been included in the study design. In addition, appropriate statistical analyses must have been performed and the statistical significance reported. Finally, toxicity results were examined to determine if they were ecologically and regionally relevant.

For the purposes of deriving a TRV for an ecological risk assessment, an ecologically relevant endpoint is one closely tied to the functioning, reproduction, and survival of a population. Usually, the endpoints that are measured for this purpose are survival, growth, and reproduction. Wherever possible, higher priority was placed on TRVs derived for selected receptors within the areas of Green Bay, Lake Michigan, or the Great Lakes. However, despite the numerous toxicity tests conducted on fish and birds from the Great Lakes, including Lake Michigan, tests using field-collected species are confounded by the presence of multiple contaminants and potentially non-optimal health. Therefore, where available, TRVs were based on laboratory toxicity tests that used uncontaminated sources of species rather than field tests or laboratory tests on field-collected organisms which may be affected by both contaminant and non-contaminant stressors. Selected TRVs are shown on Table 6-5.

TRV Effect Levels or Dietary Thresholds

For each COPC and receptor, two effect levels were selected as TRVs: a NOAEC and a LOAEC. The NOAEC is defined as the highest concentration known to ***not*** cause unacceptable adverse effects. The LOAEC is the lowest concentration measured associated with an adverse impact that is statistically different from responses observed in control organisms in controlled experiments. The selected TRVs are based on studies that most closely satisfied the described requirements and best professional judgement. Ideally, both the LOAEC and NOAEC are obtained from the same study so that the same field or experimental test conditions are taken into account. In instances where only one of these effect levels was reported, the other effect level was estimated rather than obtaining it

from an alternate source. Estimated LOAECs and NOAECs were calculated based on an extrapolation factor of 10 which was used to convert from one to the other (EPA, 1989c; Sample *et al.*, 1996).

In instances where receptor tissue residue data is not available for a COPC, receptor risk was evaluated from estimated COPC consumption and established dietary thresholds.

TRVs for Aquatic and Benthic Invertebrates

The TRVs for both water column invertebrates and benthic invertebrates are based upon promulgated federal or state criteria, or regionally accepted criteria or threshold values. Wisconsin aquatic state criteria are less than or equivalent to federal aquatic criteria.

Water column invertebrate TRVs were selected through the following hierarchy:

- Wisconsin warm water *Surface Water Quality Criteria and Secondary Values for Toxic Substances* (Wisconsin Administrative Code Chapter NR 105), if there are no state criteria for that COPC, then

- Federal freshwater chronic National Ambient Water Quality Criteria (NAWQC) if there are no federal criteria for that COPC, if there are no state criteria for that COPC, then

- A threshold from reviewed literature.

Wisconsin state criteria were selected as the first criteria because they are regionally specific and applicable to the Lower Fox River and Green Bay area being investigated for risk. Secondly, National Ambient Water Quality Criteria were selected as appropriate standards to evaluate toxicity given wide applicability of these criteria and the broad review that was conducted to derive these criteria. Finally, if no state or federal criteria have been established for a COPC, it was necessary to review primary literature.

Sediment TRVs were selected through the following hierarchy:

- *ARCS Program Sediment Effects Concentration* (SEC) (EPA, 1996a); if there are no SEC criteria for that COPC, then

- *Draft Federal Sediment Quality Criteria* (SQC) (Federal Register, Vol. 59, No. 11, January 18, 1994); if there are no federal criteria for that COPC, then

- Environment Canada Threshold Effects Level (TEL) (Smith *et al.*, 1996), and

- Estimated Sediment Chemistry Screening Values (EPA, 1997b).

SECs from the Assessment and Remediation of Contaminated Sediments (ARCS) program were selected as the primary source for sediment thresholds because these thresholds were derived for the Great Lakes based on survival, growth, and reproduction of the amphipod *Hyalella azteca* and/or the midge *Chironomus riparius* in sediment toxicity tests. SECs are defined as the concentrations of individual contaminants in sediment below which toxicity is rarely observed and above which toxicity is frequently observed. These sediment toxicity evaluations are the most comprehensive, site-specific, and current source of TRVs for benthic invertebrates, and these toxicity data have been used as threshold criteria in previous assessments. In the absence of an ARCS SEC, the remaining criteria will be used in the order shown (as discussed in the SLRA) (RETEC, 1998b).

6.3.2 Polychlorinated Biphenyls (PCBs)

Polychlorinated Biphenyl General Effects and Mode of Action

PCBs can produce a wide variety of responses in organisms, and have been documented as neurotoxicants, hepatotoxicants, immunotoxicants, and carcinogens (Safe, 1991; Shain *et al.*, 1991). PCBs potentially exert broad toxic effects on virtually all organisms of concern in the Fox River watershed. While sensitivity and responses tend to be species-specific (Eisler, 1986), general responses include lethality, reproductive and/or developmental toxicity, hepatic lesions, tumor promotion, suppression of the immune system, and induction of drug-metabolizing enzymes (McFarland and Clarke, 1989; Safe, 1990). Recently, PCBs have been implicated as potentially causing endocrine (hormonal) disruption in certain fish and wildlife species (EPA, 1997c). However, this has yet to be fully documented and, as such, risks of PCBs due to endocrine disruption will not be evaluated as part of this BLRA.

Effects to wildlife can range from reproductive failure, birth defects, liver damage, and tumors to wasting syndrome and death (Eisler and Belisle, 1996). In vertebrates, PCBs induce metabolic breakdown in the liver through enzyme induction within the cytochrome P450 system (Eisler and Belisle, 1996). The degree of metabolic breakdown is primarily dependent on the degree of chlorination and their spatial arrangement. As the number of chlorine atoms in the PCB molecule increase, and the number of unsubstituted adjacent carbon atoms decrease, metabolic transformation decreases. PCB elimination is limited

due to the highly lipophilic nature of these compounds. This causes PCBs to bioaccumulate in organisms and biomagnify up the food chain.

PCB risks to wildlife may not be adequately described by measuring concentrations of Aroclors or total PCBs in water, sediments, or tissues (McFarland and Clarke, 1989). As noted in the Remedial Investigation, expressions of Aroclors or total PCBs are summations of a complex mixture of 209 possible PCB congeners. A limited subset of these congeners are most frequently found concentrated in tissues. McFarland and Clarke (1989) reported that as much as 75 percent of tissue burdens of PCBs in invertebrates, fish, birds, and mammals could be attributed to only 25 specific congeners. Within the ecotoxicological literature, there is a growing body of scientific evidence which supports the theory that the most toxic of these congeners are the planar non-, ortho-, or mono-ortho substituted PCBs, which chemically resemble, and toxicologically behave similarly to, the 2,3,7,8- substituted polychlorinated dibenzofurans (PCDFs) and dibenzo-p-dioxins (PCDDs) (Walker and Peterson, 1991). Collectively, these compounds are referred to as planar chlorinated hydrocarbons (PCH) and of the 209 PCB congeners, 20 can have a planar configuration (Eisler and Belisle, 1996). However, their potencies vary by many orders of magnitude (Safe, 1991). Specifically, several lines of testing have implicated the planar PCB congeners 77, 81, 126, and 169 as major contributors to the toxicity of PCB mixtures (Ankley *et al.*, 1991).

Examination of field and laboratory data suggest that many of the toxic effects caused by PCHs are mediated subcellularly by the aryl hydrocarbon receptor (Ah-R); the same receptor responsible for mediating dioxin toxicity.[6] This receptor is involved in the translocation of PCHs into the nucleus and their subsequent binding to the PCH-Ah receptor complex on the DNA (Safe, 1991). The signs of PCB 126 toxicity in lake trout early life stages are similar to those shown by TCDD, and include yolk-sac edema, multifocal hemorrhages, craniofacial malformation, in addition to mortality (Zabel *et al.*, 1995). However, recent work has suggested that while the TCDD-like congeners act by a common mechanism (i.e., the Ah receptor), the combined effects of TCDD with the coplanar PCB congeners may not be strictly additive (Walker *et al.*, 1996). Despite this uncertainty, the additive model continues to be acceptable for assessing risk because deviation from additivity has been estimated to be within an accepted tenfold range (Walker *et al.*, 1996).

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[6] PCB toxicity is not limited to either its planar constituents nor the Ah-R receptor-mediated pathway (e.g., estrogenic mimicry and disruption of Ca^{2+} homeostasis). For a discussion of the most recent research on these PCB toxicity issues, refer to Fischer *et al.*, 1998, Symposium overview: Toxicity of non-coplanar PCBs. *Toxicological Sciences.* 41:49–61.

Polychlorinated Biphenyl Toxic Equivalency Factors (TEFs)

As a means of normalizing toxicity amongst dioxin-like compounds, toxicity is expressed relative to the most toxic PCH (2,3,7,8-TCDD) by the use of toxic equivalency factors (TEFs) (Safe, 1990, 1991). The term TEF is generally defined as the relative potency of a compound compared to the ability of 2,3,7,8-TCDD to cause a particular toxic or biological effect. TEFs are calculated by setting the toxic potency of 2,3,7,8-TCDD equal to 1.0, and determining the relative potencies of other PCHs as the ratio of the concentration of PCH to the concentration of 2,3,7,8-TCDD producing an equivalent response.

Multiplication of each congener concentration by its TEF generates a toxic equivalency (TEQ) for that congener. The sum of the TEQ for each congener yields the total TEQ for the mixture. This model assumes that congeners act additively through a similar mode of action to produce toxicity. The additivity of dioxin-like compounds is based on an Ah receptor-mediated response. For this reason, TEFs cannot be applied to non-Ah receptor-mediated effects. Total TEQ levels, rather than individual congener levels, have been shown to better correlate with biological endpoints, such as lethality and deformities in fish, mammals, and birds (Ankley *et al.*, 1989; Giesy *et al.*, 1995, 1994a, 1994b; Tillitt *et al.*, 1992; Walker and Peterson, 1991; Zabel *et al.*, 1995).

TEFs are frequently based on biochemical endpoints such as the induction of cytochrome P4501A or binding affinity to the Ah receptor, although other endpoints such as lethality and deformity have been used for TEF development. The mechanisms by which changes at the biochemical level can result in changes at the population level (i.e., survival and reproduction endpoints) have not been well defined. The toxic responses in fish, birds, and mammals are affected by: age, size, reproductive state, sex, nutrition, and other environmental conditions (Van den Berg *et al.*, 1998). To estimate PCB and dioxin risks to fish, birds, and mammals, it is necessary to identify a tissue or dietary TEQ concentration for the receptor organism that is associated with a specific adverse effect. By comparing the total TEQ in an exposed organism to the 2,3,7,8-TCDD TRV for that species, the risk associated with any mixture of PCDDs, PCDFs, and PCBs can be estimated.

Because PCB mixtures are composed of diverse congeners with differing toxicities, there is still a large degree of uncertainty surrounding the ability of PCB TEQ calculations to adequately predict adverse effects. Variability in TEQ calculations inherently result from the use of different TEF values; however, even when the same TEF values are used TEQ calculations do not appear be consistently correlated to adverse effects. Part of this correlation variation may be due to the fact that only a few congeners, those assumed to be the most toxic, are evaluated

and considered to represent the toxicity of the entire PCB mixture. Also, differential response to COPCs may not only be species-specific, but also population specific as in the case of PCBs tested on lake trout (Zabel *et al.*, 1995) and double-crested cormorants (Haffner *et al.*, 1997).

Uncertainties regarding TEFs generally include potential nonadditive interactions, differences in the shape of the dose-response curve, and differences in species responsiveness. Despite these uncertainties, it was concluded that the TEF methodology remained the most plausible and feasible approach for risk assessment of halogenated aromatic hydrocarbons with dioxin-like properties (Van den Berg *et al.*, 1998).

In 1997, an expert meeting was organized by the World Health Organization (WHO) to derive consensus TEFs for PCDDs, PCDFs, and PCB congeners. This effort represents the most recent and extensive review of TEFs for fish, birds, and mammals. Recommended WHO TEFs for fish, birds, and mammals were published by Van den Berg *et al.* in 1998.

The following criteria were used to determine whether a study should be included in the derivation of TEF values: 1) at least one PCDD, PCDF, or PCB congener and a reference compound were studied; 2) either TCDD or PCB 126 was included as a reference compound in the experiment or studied with the same experimental design by the same authors; and 3) the relevant endpoint would affect both the congener studied and the control. It was also decided that for a compound to be assigned a TEF value, it must: 1) show a structural relationship to PCDDs and PCDFs; 2) must bind to the Ah receptor; 3) must elicit Ah receptor-mediated biochemical and toxic responses; and 4) must be persistent and accumulate in the food chain (Van den Berg *et al.*, 1998).

Subsequently, the WHO TEFs (Van den Berg *et al.*, 1998) have been used internationally, and are currently considered the most widely accepted TEFs. These TEFs for birds and fish are presented in Table 6-6 along with other developed TEFs for comparison. Justification of TEFs selected for this assessment are provided below for fish, birds, and mammals. While TEFs are available for more PCB congeners than those that are listed in Table 6-6 (congeners 77, 81, 105, 118, 126, and 169), these are the congeners that are most likely bioaccumulate and pose dioxin-like toxicity, and are the only congeners that are evaluated in this assessment.

Fish TEFs. Numerous TEFs have been derived for salmonid fish (Walker and Peterson, 1991; Newsted *et al.*, 1995; Zabel, 1995), because salmonids are known to be sensitive species. Not only did Walker and Peterson (1991) and Zabel (1995) use

the same endpoint, they also used the same test species—rainbow trout. All of the TEFs proposed by Walker and Peterson (1991) were confirmed by Zabel (1995), but Zabel additionally presented TEFs for PCB and dioxin/furan congeners that were not reported by Walker and Peterson (1991). The WHO has recently reviewed each of these TEF sources as well as additional data, and has proposed unique TEFs (Van den Berg *et al.*, 1998).

Fish TEFs have either been based on CYP1A induction (Newsted *et al.*, 1995) or early life-stage mortality (Walker and Peterson, 1991; Zabel, 1995). Those for mortality are generally lower than those for CYP1A induction. Overall, however, fish are considered less responsive, in terms of CYP1A1 enzyme induction, to mono-ortho PCBs than birds and mammals, despite having a similar hepatic cytochrome P450 enzyme system (Van den Berg *et al.*, 1998).

The most common method of deriving TEFs in fish is through egg injection studies. Studies have shown that mortality of trout sac fry to waterborne exposures are nearly identical to those seen in egg injection studies (Van den Berg *et al.*, 1998). Also, early life-stage mortality, as a response to exposure to congener 126, has been shown to be similar for both rainbow trout and lake trout (Zabel *et al.*, 1995).

All of the above studies were included in the selection of WHO fish TEFs. For this reason the 1998 WHO fish TEFs will be used in this BLRA to assess risk to fish receptors in the Lower Fox River and Green Bay.

Bird TEFs. Many studies have investigated Ah receptor-mediated toxic responses in whole birds or bird eggs following exposure to TCDD and PCBs. However, these studies were not designed to determine comparative relative potencies of TCDD and PCB congeners and, therefore, cannot be used for the derivation of TEFs. Currently, TEFs for birds are based on egg injection, avian hepatocyte, or cultured thymus cell studies (Van den Berg *et al.*, 1998). These test methods have been used in experiments on numerous species including domestic chicken, domestic duck, domestic turkey, pheasant, gull, common tern, double-crested cormorant, and American kestrel (Van den Berg *et al.*, 1998). CYP1A1 induction in avian hepatocytes has been shown to be well correlated with embryo mortality (Van den Berg *et al.*, 1998).

In general, the relative potencies of PCBs, PCDDs, and PCDFs in birds is similar to the relative potencies of these compounds in mammals, except for TCDF, which, based on bird hepatocyte cultures, is more toxic than TCDD (Van den Berg *et al.*, 1998). Unlike fish, birds and mammals experience greater relative toxic potency from exposure to mono-ortho PCBs. Also, based on studies from

chicken embryos, it has been concluded that birds can be highly responsive, in terms of CYP1A1 induction, toward exposure to non-ortho PCBs (Van den Berg *et al.*, 1998).

Specific TEFs used for the evaluation of bird exposure to PCB congeners include Tillitt *et al.* (1991a) and, more recently Kennedy *et al.* (1996). TEFs, proposed by Tillitt *et al.* (1991a,) were developed by using the H4IIE rat hepatoma assay. This assay was developed by Tillitt *et al.* to measure the relative TCDD equivalent (TCDD-Eq) response of any sample extract on which the assay was run. The specific response being measured in this assay is induction of the cytochrome P4501A1 monoxygenase enzyme. The H4IIE assay has been shown to be very sensitive to TCDD and PCHs; the lower limit of detection of TCDD is 10 pg and the coefficient of variation for environmental samples is between 10 and 20 percent (Tillitt *et al.*, 1991a). TEFs were developed by assaying pure PCB congeners (77, 126, 105, 156) and 2,3,7,8-TCDF, and comparing the response levels to the response observed for 2,3,7,8-TCDD alone. For comparison, chicken eggs were spiked with congener 77, then extracted, followed by extract assay. Results indicated that TCDD-Eq levels measured on the pure congener 77 and the congener 77 spiked chicken egg extract were essentially equivalent. Based on these results, it was concluded that this assay could be used to determine the level of TCDD-Eqs in field-collected bird eggs. From Green Bay specifically, extracts from field-collected double-crested cormorant eggs and Caspian tern eggs have been analyzed using the H4IIE assay (Tillitt *et al.*, 1991b).

Subsequently, the PCH toxicity investigations by Tillitt and others (Tillitt *et al.*, 1992; Jones *et al.*, 1993; Williams *et al.*, 1995a, 1995b; Ludwig *et al.*, 1996; Froese *et al.*, 1998) on field-collected birds and bird eggs from the around the Great Lakes has followed either of two analytical methods using tissue/egg extraction followed by analysis using the H4IIE assay to derive a TCDD-Eq concentration, or analysis for congener concentrations followed by multiplication by respective TEFs (from Tillitt *et al.*, 1991a) and summation to derive a total TEQ. Some researchers explicitly used both of these methods for comparative purposes.

TEFs proposed by Kennedy *et al.* (1996) were based on ethoxyresorufin-O-deethylase (EROD) induction in chicken embryo hepatocyte cultures and these TEFs were much greater than the TEFs derived by Tillitt *et al.* (1991a). This is expected given that chickens are known to be one of the most sensitive avian species to PCH exposure (Eisler and Belisle, 1996), and are specifically known to be more sensitive than double-crested cormorants and Caspian terns (Ludwig *et al.*, 1996). Because double-crested cormorants and Caspian terns are selected receptors for this BLRA, use of the Kennedy *et al.*

(1996) TEFs was considered to potentially overestimate risk to these piscivorous species.

When the WHO reviewed and derived TEF values, their review included the work by Tillitt *et al.* (1991a) and also relied on chicken data (both egg injection and hepatocyte) as investigated by Kennedy *et al.* (1996) (Van den Berg *et al.*, 1998). For all TEFs that both Kennedy *et al.* (1996) and Tillitt *et al.* (1991a) derived, the corresponding WHO TEFs were in between these two; the only exception to this is for congener 77 where the WHO TEF exceeded both the Kennedy and Tillitt *et al.* TEFs. Additionally, the WHO derived bird TEFs for dioxin and furan congeners that had not previously been determined.

For risk evaluation of the identified avian receptors in the Lower Fox River and Green Bay, both the TEFs of Tillitt *et al.* (1991a) and the WHO TEFs (Van den Berg *et al.*, 1998) will be used.

Mammalian TEFs. The majority of TEF values derived for mammals come from rat studies. The WHO determined that the relative toxic potencies of PCDDs and PCDFs are not different between mink and rat; however, because the FRDB contains no piscivorous mammal tissue data, mammalian TEFs were not used as part of assessing congener risk for piscivorous mammals.

Ecotoxicity of Polychlorinated Biphenyls (PCBs)

The effects of PCBs on Great Lakes fish and wildlife has been extensively documented. PCB-induced reproductive impairment has been demonstrated for several fish species (Mac, 1988; Ankley *et al.*, 1991; Walker and Peterson, 1991; Walker *et al.*, 1991a, 1991b; Williams and Giesy, 1992), a number of insectivorous and piscivorous birds (Kubiak *et al.*, 1989; Gilbertson *et al.*, 1991; Tillitt *et al.*, 1992) and mink (Aulerich *et al.*, 1973, Aulerich and Ringer, 1977; Bleavins *et al.*, 1980; Wren, 1991; Giesy *et al.*, 1994c; Heaton *et al.*, 1995a, 1995b; Tillitt *et al.*, 1996). A more detailed discussion for each of the receptor groups is given below. A discussion of PCB toxicity is limited to total PCBs. PCB congener toxicity is discussed in the following section on dioxin.

Polychlorinated Biphenyl (PCB) Toxicity to Aquatic and Benthic Invertebrates. For determination of risks to pelagial aquatic invertebrates, a LOAEC TRV of 0.5 μg/L total PCBs will be used with a tenfold lower estimated NOAEC TRV of 50 ng/L based on the review by Niimi (1996). For determination of risks to benthic infauna, the ARCS SEC (EPA, 1996a) sediment value of 31.6 μg/kg dry weight total PCBs will be used.

Invertebrates do not have an Ah receptor and are, therefore, not impacted by this receptor-mediated toxicity. Also, invertebrates have a limited cytochrome P450 detoxification system, so there is limited metabolic breakdown of these compounds. As a result, PCB toxicity to invertebrates is potentially less than that experienced by vertebrate species, and PCBs are retained in invertebrate tissues. The ability of invertebrates to accumulate PCBs from sediment or the water column makes them good indicators of both sediment and water quality, and makes these contaminants available for trophic transfer to fish and other wildlife species.

Aroclor mixtures with elevated sediment hazard quotients within the Lower Fox River and Green Bay system include Aroclors 1242, 1248, 1254, and 1260 (RETEC, 1998b). Therefore, these Aroclors are the most likely contributors to water column PCB concentrations. Data from toxicity studies of PCB Aroclor mixtures have been compiled by the EPA on the AQUIRE database (http://www.epa.gov/medecotx/data_download/aquire/aquire_ascii_download.htm). Invertebrate toxicity data using the water flea *Daphnia magna* as a test species for several Aroclors indicated toxicity in the range of 24 to 206 μg/L for reproductive EC_{50}s, and 25 to 253 μg/L for mortality LC_{50}s.

The review of PCB toxicity by Niimi (1996) has suggested that PCB concentrations of greater than 10 μg/L cause zooplankton death within a few days, and concentrations of 1 to 10 μg/L cause death over longer periods of exposure. Shrimp and oyster invertebrates, however, are more tolerant, where concentrations of greater than 10 μg/L or greater than 25 μg/L may cause death. Sublethal effects, such as zooplankton reproduction, are affected by PCB concentrations greater than 1 μg/L and macroinvertebrate developmental effects can occur at concentrations of 0.5 to 5 μg/L. Based on this review by Niimi (1996) a LOAEC TRV of 0.5 μg/L total PCBs will be used with a tenfold lower estimated NOAEC TRV of 50 ng/L for determination of risks to pelagial aquatic invertebrates.

For determination of risks to benthic infauna, the ARCS SEC (EPA, 1996a) sediment value of 31.6 μg/kg dry weight total PCBs will be used. This specific value is the Threshold Effect Level (TEL) derived from a 28-day chronic test using *Hyalella azteca*.

Polychlorinated Biphenyl (PCB) Toxicity to Fish. The total PCB TRVs selected for whole body fish are a NOAEC of 0.76 mg/kg and an estimated LOAEC of 7.6 mg/kg, where the endpoint is fish growth. These TRVs were primarily based on two toxicity tests: one using rainbow trout (Hendricks *et al.*, 1981) and one using

lake trout (Mac and Seelye, 1981). These TRVs will be used to assess risk to all selected fish receptors based on the whole body data contained within the FRDB.

PCBs have been measured in many important fish species of the Great Lakes, as well as the most abundant species of the Lower Fox River and Green Bay (e.g., alewife, carp, perch, pike, walleye) (Ankley *et al.*, 1992; Brazner and DeVita, 1998). Unfortunately, for the selected receptor species in this BLRA, there are no available PCB toxicity reference values. Much of the available toxicity data are for salmonids, which have been found to be very sensitive to PCBs.

During the 1970s, salmonid hatchery managers in the Great Lakes region noticed increased mortalities in chinook salmon fry (*Oncorhynchus tschawytscha*) and persistent organic contaminants were suspected as a possible cause of continued reproductive failure (Mac, 1988). PCB exposures in laboratory-rearing water have produced similar symptoms and mortalities as those observed in hatchery fry (Hogan and Brauhn, 1975; Berlin *et al.*, 1981; Mac, 1988; Williams and Giesy, 1992). PCB concentrations in eggs have also been the suspected cause of recruitment failure in lake trout (*Salvelinus namacyacush*), which have experienced significant early life-stage mortality in contaminated regions of the Great Lakes (Mac *et al.*, 1985; Mac, 1988). Other reported effects of PCBs on fish have included: changes in growth, increased activity of cytochrome P450 enzymes, and sublethal effects such as sluggishness or weight loss (Eisler and Belisle, 1996).

Generally, the most sensitive endpoints for effects of PCBs in fish are early life-stage survival and recruitment where exposure has resulted from transfer of PCBs from maternal tissue to eggs (Eisler and Belisle, 1996; Walker *et al.*, 1996). Whole body environmental concentrations of PCBs in adult fish do not generally result in death (Eisler and Belisle, 1996). Numerous field studies evaluating PCB fish tissue concentrations and adverse effects, as summarized by Niimi (1996), also supports this. Based on several field studies, lethal body burden concentrations have been estimated at greater than 100 mg/kg for young fish and greater than 250 mg/kg for older fish (Niimi, 1996). Although it is difficult to separate the effects of PCBs alone in these field studies, since the tissues also contain varying amounts of other organochlorine residues (e.g., DDT), the Niimi review does conclude that greater than 50 mg/kg of PCBs may cause chronic effects in fish. Field studies during the 1970s and 1980s on the effects of PCBs in fish populations in the Great Lakes were also compromised by the fact the source of control fish for the tests already had body burdens of organochlorines. This confounding factor, as further described below, resulted in the selection of growth as the TRV toxicity endpoint rather than early life-stage mortality.

The experiment of Mac and Seelye (1981), on which the selected TRVs are based, involved the examination of effects (growth and mortality) of Aroclor 1254 when exposed to hatchery lake trout fry through food (1 mg/kg) and water (50 ng/L). Fish were exposed to treatments with and without the solvent acetone, which potentially increases uptake of PCBs, for up to 52 days. At the end of 52 days of exposure, mortality was not significantly different between treatment groups, but growth, as evaluated through both length and weight, was significantly different. Non-solvent control fry weight and length were significantly less than the non-solvent PCB-treated fish. PCB residues (wet weight) were 0.24 mg/kg in the non-solvent control fish and 1.8 mg/kg in the non-solvent PCB exposure group. Therefore, the concentration of 1.8 mg/kg represents a growth LOAEC. The endpoints (growth and mortality) were also measured in the treatment groups on days 17 and 41 of the exposure, and neither endpoint on these days was significantly different when the non-solvent control and the non-solvent PCB treatment were compared. The PCB residues in the non-solvent treatment were 0.76 mg/kg and 2.1 mg/kg on days 17 and 41, respectively. Therefore, assuming that the concentration of 1.8 mg/kg was a LOAEC, the concentration of 0.76 mg/kg was considered to be a NOAEC.

The experiment of Hendricks *et al.* (1981) was also used to support the selection of PCB TRVs. This experiment involved the exposure of two female laboratory-reared rainbow trout, both 22 months old, to a diet containing 200 mg/kg Aroclor 1254 for 2 months prior to spawning. Eggs from these PCB-exposed females were fertilized with sperm from non-exposed males and, at the same time, a set of control eggs were also fertilized with sperm from non-exposed males. Fertilized eggs were allowed to develop without further exposure to PCBs and, beginning at swim-up, they were fed a control diet ad libitum for a period of 1 year. PCB concentrations were determined in the tissues on the day of spawning, day 21 of egg incubation, and 1 week following hatching. Fish were sacrificed at 9 and 12 months and examined for weight and liver tumor incidence. Measured PCB concentrations in the PCB-exposed eggs (1.64 mg/kg ww on spawning day) were significantly greater than measured concentrations in the control eggs (0.47 mg/kg ww on spawning day) and the concentrations remained significantly different on day 21 following spawning and on day 7 after hatching. Mortality and incidence of liver tumors were not significantly different because of PCB exposure, but growth was significantly reduced at both 9 and 12 months as compared to the control. Therefore, based on these results, the egg concentration of 1.6 mg/kg was considered to be a growth LOAEC.

The Hendricks *et al.* (1981) study did not report estimated rates of maternal transfer for PCBs. Therefore, other sources of maternal transfer rates were consulted. Maternal-egg transfer of PCHs has been demonstrated for salmon

(Ankley *et al.*, 1989), trout (Newsted *et al.*, 1995; Mac *et al.*, 1993), pike (Larsson *et al.*, 1993), brook trout (Johnson *et al.*, 1998b), and walleye (Fisk and Johnston, 1998). The research by Johnson *et al.* (1998b) indicated that 39 percent of maternal female concentrations of TCDD were contained in eggs. Mac *et al.* (1993) reported the maternal transfer rate of PCBs to eggs in nine lake trout; transfer rates ranged from 10.6 percent to 24.8 percent with an average of 20.9 percent. Applying the transfer rate of 20.9 percent to the Hendricks *et al.* (1981) data, a PCB LOAEC egg TRV of 1.6 mg/kg is equivalent to an estimated whole body PCB LOAEC TRV of 7.6 mg/kg. Therefore, an estimated whole body PCB NOAEC TRV is 0.76 mg/kg, which is equivalent to the NOAEC TRV reported by Mac and Seelye (1981) as discussed above.

The maternal transfer rates presented above each suggest that maternal tissues can transfer between 0 and 100 percent of the PCB concentration in maternal tissues to eggs. A study by Russell *et al.* (1998) on maternal-to-egg transfer rates of organochlorine contaminants in fish and other wildlife, however, has indicated that concentrations transferred to fish eggs may exceed the concentrations in the maternal tissues. The results in this study were reported in terms of the ratio of the concentration in eggs to the concentration in maternal tissues, rather than the relative percentage of contaminant transferred to eggs. This study had a modeling component and a field study component, and the results of each were compared. The model assumed that organochlorine contaminants are transferred from maternal to egg tissues through a passive process of chemical equilibrium; that equilibrium is rapidly achieved; that organochlorine contaminants are transferred in lipoproteins and once transferred to eggs, the organochlorine contaminants are not metabolized; and that lipids are the sole component of the total fugacity of the egg and maternal tissues. This model predicted that the lipid-normalized egg-to-maternal tissue ratio of organochlorine contaminant concentrations is 1.0. Field testing of these model results was undertaken between May and November for the following species: black crappie, quillback carpsucker, carp, gizzard shad, freshwater drum, and whitefish. Ninety-five percent of egg-to-muscle ratios fell within the range of 0.46 to 54, and the mean ratio was 11.8 (i.e., eggs contained a higher concentration of the organochlorine than maternal tissues). When these ratios were lipid normalized, 95 percent of the ratios were within a factor of two of the mean ratio of 1.22, and essentially in agreement with the modeled results. Lipid normalization reduced the variability between species of the egg-to-maternal organochlorine concentrations by 20 times. Part of this variability is believed to be caused by insufficient time for lipids to reach an equilibrium between maternal and egg tissues. How this variability and uncertainty may have affected the results of this assessment are discussed in Section 6.6.

Other studies considered for the selection of PCB TRVs for whole fish were studies using hatchery- or laboratory-reared fish (Freeman and Idler, 1975; Mauck *et al.*, 1978; Mayer *et al.*, 1977; Mayer *et al.*, 1985; Hogan and Brauhn, 1975; Lieb *et al.*, 1974) and laboratory studies using fish collected from Lake Michigan (Berlin *et al.*, 1981; Mac and Edsall, 1991; Stauffer, 1979; Ankley *et al.*, 1991; Mac and Schwartz, 1992; Mac *et al.*, 1993). Endpoints evaluated in the studies using hatchery- or laboratory-reared fish included growth, survival, and hatching success, and effect levels were found to occur only at much higher levels than field studies have reported. Growth NOAECs ranged from 8.5 to 70 mg/kg ww, growth LOAECs ranged from 125 to 645 mg/kg ww, survival NOAECs ranged from 8.5 to 120 mg/kg ww, survival LOAECs ranged from 125 to 645 mg/kg ww, and the only reported threshold for hatching success was a LOAEC of 77.9 mg/kg. Endpoints evaluated in the studies using fish collected from Lake Michigan included growth, survival, and reproductive success. The only reported growth threshold was a LOAEC of 1.53 mg/kg (Berlin *et al.*, 1981). Survival NOAECs ranged from 0.17 to 6.33 mg/kg ww, survival LOAECs ranged from 0.31 to 1.53 mg/kg ww, reproductive success NOAECs ranged from 2.8 to 3.7 mg/kg ww, and reproductive success LOAECs ranged from 3.3 to 4.2 mg/kg ww.

In support of the Natural Resource Damage Assessments (NRDA) within the Lower Fox River and Green Bay, the USFWS examined sublethal health effects in walleye using the following endpoints: liver histopathology, immune response, incidence of infection, hepatic EROD activity, and plasma vitellogenin (Stratus Consulting, 1999a). The only statistically significant differences between assessment area fish and reference area fish were the increased incidence of liver tumors and pre-tumors in assessment area fish, particularly in females. Although PCB concentrations in fish livers from the assessment area were also significantly higher than reference concentrations, the causal relationship between tumor incidence and PCB concentrations was not established. Therefore, TRVs for these sublethal effects could not be established.

The NRDA investigation for fishery injuries also focused on reproductive effects in lake trout using the following endpoints: egg mortality, fry mortality, unhatched eggs, and abnormal fry hatched. These results were then compared to egg concentrations of thiamine, total PCBs, and total TEQ, and the only significant correlation found was between thiamine concentration and fry mortality; with thiamine deficiency correlating to increased fry mortality (swim-up syndrome). Also, thiamine-deficient eggs were more susceptible to adverse effects from PCBs. Further investigation of why lake trout eggs are thiamine deficient has implicated a primary prey of these fish—alewife, because alewife contain thiaminase, an enzyme that destroys thiamine. This is a relatively new hypothesis, however, with few supporting data (Stratus Consulting, 1999a).

Therefore, until the contribution of thiamine deficiency towards toxicity is more understood, it will not be considered as influencing toxicity thresholds of PCBs in fish.

Polychlorinated Biphenyl (PCB) Toxicity to Birds. There is a great degree of variability among different bird species in response to PCBs. In sensitive species, normal patterns of growth, behavior, reproduction, and metabolism may be altered. Liver concentrations of PCBs are generally highest in piscivorous birds, followed by birds that feed on other small birds and mammals, birds that feed on worms and insects, and herbivorous or seed-eating birds, respectively. However, bird embryos are the most sensitive life stage for assessing the effects of contaminants (Elliott *et al.*, 1996; Kubiak and Best, 1991). The role of PCBs in reproductive impairment and development of morphological abnormalities in emergent insect- and fish-eating birds has been studied more than any other effect on wildlife species within the Great Lakes (Fox *et al.*, 1991a, 1991b; Kubiak *et al.*, 1989; Giesy *et al.*, 1995; Williams *et al.*, 1995a). Also, PCBs have been linked to lethality and deformities in embryos and chicks of double-crested cormorants (*Phalacrocorax auritus*) (Tillitt *et al.*, 1992; Yamashita *et al.*, 1993), Caspian terns (*Hydroprogne caspia*) (Yamashita *et al.*, 1993), and bald eagles (*Haliaeetus leucocephalus*) (Giesy *et al.*, 1995).

TRVs were developed for three separate endpoints and used to assess risk to the selected bird receptors. These TRVs are:

1. A LOAEC TRV of 7.6 mg/kg-egg and a NOAEC TRV of 4.7 mg/kg-egg were selected as TRVs primarily based on the work by Hoffman *et al.* (1993) in Green Bay and other Bays around the Great Lakes. These TRVs will be used to assess hatching success risk to all bird receptors from total PCBs by comparison to egg and whole body concentrations.

2. An estimated LOAEC of 8 mg/kg-egg and a NOAEC of 0.8 mg/kg-egg were selected as TRVs based on the work by Ludwig *et al.* (1996). These TRVs will be used to assess deformity risk to all bird receptors from total PCBs by comparison to egg and whole body concentrations.

3. A LOAEC TRV of 1.12 mg/kg-BW/day and an estimated NOAEC TRV of 0.11 mg/kg-BW/day were selected as TRVs based on the work by Peakall and Peakall (1973) and Tori and Peterle (1983). These TRVs will be used to assess dietary risk to piscivorous bird receptors from total PCBs by comparison to estimated dietary exposure concentrations.

These TRVs will be used for all bird receptor species, except for the dietary TRV which will not be used for insectivorous birds because exposure is not being modeled for these receptors. However, as indicated from the research by Custer *et al.* (1998) on the Lower Fox River and Green Bay, tree swallows had no adverse effects to hatching success even though they accumulated organochlorines. This observation suggests that they are less sensitive than other birds identified as receptors of concern in this BLRA, and dietary risks to bird insectivores are adequately represented by these other bird receptors.

Selection of bird TRVs are discussed in detail below for each endpoint.

Egg TRV - Hatching Success. PCB toxicity and its effects have been extensively studied within the Lower Fox River and Green Bay. Kubiak *et al.* (1989) conducted a study in 1983 and reported elevated PCB concentrations (mean = 22.2 mg/kg) in Forster's tern (*Sterna fosteri*) eggs in Green Bay, that were associated with significantly reduced hatching success when compared to control eggs collected at Lake Poygan containing 4.5 mg/kg total PCBs. Harris *et al.* (1993) sampled the same colonies 5 years later in 1988 and noted that measures of reproductive performance (hatching success, number of young fledged, and length of incubation) were improved from 1983.[7] Total PCB concentrations in the eggs were 7.3 mg/kg and when compared to the Lake Poygan control data collected by Kubiak *et al.* (1989), hatching success was not significantly different. Based on data presented in Kubiak *et al.* (1989), a NOAEC and a LOAEC of 4.5 mg/kg-egg and 22.2 mg/kg-egg, respectively, can be derived for hatching success in the Forster's tern.

Hoffman *et al.* (1993) did not observe any apparent adverse effects in a field population of common terns with corresponding egg PCB concentrations of 4.7 mg/kg ww, but a decrease in hatching success and increase in embryo deformities was observed at a corresponding egg PCB concentration of 7.6 mg/kg ww.

Bosveld and Van Den Berg (1994) reported adverse effects on hatching success in the Forster's tern and common tern at egg PCB concentrations of 19 and 8 mg/kg, respectively, with a corresponding NOAEC for both bird species of 7 mg/kg (Bosveld and Van Den Berg, 1994). However, Struger and Weseloh (1985) did not observe any adverse effects on eggshell thickness or reproductive success in Caspian terns from the Great Lakes with egg PCB concentrations as high as approximately 39 mg/kg PCBs.

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[7] Note that while the Harris *et al.* (1993) study states that the same Green Bay colonies were sampled, the two studies (Kubiak *et al.*, 1989 and Harris *et al.*, 1993) collected samples at different locations in southern Green Bay: Longtail Point/Oconto Marsh and Renard Island, respectively.

In nine colonies of great blue herons, no apparent adverse reproductive effects were observed at the highest mean egg PCB concentration of 7.8 mg/kg (Boily *et al.*, 1994). Similarly, no adverse reproductive effects were observed in a field population of black-crowned night herons with mean egg PCB concentrations of up to 10.9 mg/kg ww (Tremblay and Ellison, 1980).

Tillitt *et al.* (1992) monitored 11 double-crested cormorant colonies around the Great Lakes as well as a reference site outside of the Great Lakes for hatching success in 1986, 1987, and 1988. A significant correlation was found between total egg PCB concentrations and egg mortality. A NOAEC and LOAEC could not be derived from this study because 21 percent egg mortality was observed in a colony whose mean egg PCB concentration was 0.1 mg/kg, whereas the reference area exhibited 8 percent egg mortality with a corresponding mean egg PCB concentration of 0.8 mg/kg.

Larson *et al.* (1996) investigated double-crested cormorant hatching success on Spider Island in Green Bay and at a reference station. Clutch size and hatching success on Spider Island were significantly less than measures of these endpoints at the reference site. Total PCB concentrations in eggs were also significantly different between these two sites, where Spider Island PCB concentrations averaged 7.7 mg/kg and reference site PCB concentrations averaged 1.03 mg/kg. While these data suggest that PCB concentrations are causing adverse effects, closer examination of these data this hypothesis could not be accepted or rejected. This is due to the lack of correlation between PCB egg concentration and adverse effects on Spider Island. In Spider Island nests where all eggs hatched, the average egg total PCB concentration was 7.6 mg/kg; however, in nests where no eggs hatched, the average egg total PCB concentration was 8.22 mg/kg. These mean concentrations were not significantly different.

Total concentrations of PCBs in the eggs of bald eagles have been correlated with reproductive impairment (Wiemeyer *et al.*, 1984; Wiemeyer, 1990; Giesy *et al.*, 1995; Dykstra and Meyer, 1996) and productivity (Bowerman *et al.*, 1995; Wiemeyer *et al.*, 1993; Best *et al.*, 1993). Fish-eating birds like the bald eagle are potentially more exposed to PCBs through bioaccumulation than are other identified receptor species.

In bald eagles, the mean egg PCB concentration in successful nests (defined as having one or more young produced in the year of sample egg collection) was 7.2 mg/kg, and in unsuccessful nests the mean egg PCB concentration was 13 mg/kg (Wiemeyer *et al.*, 1984). Similar results were obtained for bald eagles by Wiemeyer *et al.* (1993), in which a significant reduction in the number of young raised were noted at a corresponding mean egg PCB concentration of 13 mg/kg,

although the authors indicate that DDE may have contributed more to the decreased production than PCBs. Wiemeyer (1990) reported that eagle egg PCB concentrations of 4.0 mg/kg should be adequate to ensure normal reproduction.

Dykstra and Meyer (1996) compared bald eagle reproductive rates, food availability, and organochlorine contaminant loads in Green Bay and inland Wisconsin. Reproductive rates in Green Bay and Lake Michigan averaged 0.39 young per occupied territory from 1990 to 1994 and reproductive rates measured in inland Wisconsin averaged 1.09 young per occupied territory from 1990 to 1993. Mean organochlorine contaminant levels in addled eggs from Green Bay were 35 mg/kg PCBs and 10.3 mg/kg DDE. Food availability, based on analyses of time spent feeding, mean rate of food delivered, and adult nest attendance was found to be normal. Therefore, these researchers concluded that PCB concentrations were likely causing reproductive impairment.

The studies of Wiemeyer *et al.* (1993, 1984), Wiemeyer (1990), and Dykstra and Meyer (1996) are confounded by the presence of DDE in the eggs, and controversy exists over the contribution of DDE versus PCBs causing the observed effects (Bosveld and Van Den Berg, 1994).

The TRVs selected for total PCB levels in bird eggs were derived from experiments conducted by Hoffman *et al.* (1993) where contaminant residues in common tern eggs were measured and hatching success was monitored. The reported LOAEC of 7.6 mg/kg-egg and a NOAEC of 4.7 mg/kg will be used as TRVs for all bird receptors.

Deformity TRV. Deformities, while potentially not a limiting factor to population levels of birds, are an important endpoint effect that will be evaluated in this BLRA. Developmental abnormalities in birds (e.g., crossed bills in double-crested cormorants) are known to result from exposure to PCBs and have been extensively reviewed as part of the Great Lakes Embryo Mortality, Edema, and Deformities Syndrome (GLEMEDS) by Gilbertson *et al.* (1991).

Hoffman *et al.* (1993) examined deformities in common tern eggs from Green Bay as compared to deformities in common tern eggs from two reference sites in the north of Lake Michigan (Cut River and Point aux Chenes). These researchers found that abnormalities in 11 Green Bay individuals included edema and incomplete skeletal ossification, while there were no abnormalities in individuals from either of the reference sites. PCB residues in eggs averaged 10.0 mg/kg for those collected from Green Bay, and 4.0 and 4.7 mg/kg for those collected from the reference sites. These residue concentrations in Green Bay eggs were statistically different. DDE and mercury residues in eggs from these stations were

also compared. Residues of DDE were not significantly different between the stations, but mercury residues were; average mercury residues from the reference sites were 0.33 and 0.37 mg/kg, and average residue from Green Bay was 0.76 mg/kg. These data suggest that a LOAEC for deformity in common terns is a total PCB residue of 10.0 mg/kg in eggs.

Larson *et al.* (1996) investigated double-crested cormorant embryo deformity on Spider Island in Green Bay and at a reference station in addition to their investigation of hatching success. There were significantly more bill defects in double-crested cormorant chicks on Spider Island as compared to the reference site. Total PCB concentrations in eggs were also significantly different between these two sites, where Spider Island PCB concentrations averaged 7.7 mg/kg and reference site PCB concentrations averaged 1.03 mg/kg. As previously indicated, while these data suggest that PCB concentrations are causing adverse effects, closer examination of these data indicate that this hypothesis could not be accepted or rejected. This is due to the lack of correlation between PCB egg concentration and adverse effects on Spider Island. In Spider Island nests where all eggs hatched, the average egg total PCB concentration was 7.6 mg/kg, in nests where no eggs hatched, the average egg total PCB concentration was 8.22 mg/kg, and in nests where a chick was deformed, the average egg total PCB concentration was 7.3 mg/kg. None of these mean concentrations were significantly different.

Ludwig *et al.* (1996) reviewed available data on concentrations of contaminants in eggs and observed deformities in embryos and chicks of double-crested cormorants and Caspian terns. Between 1986 and 1991, hatched chicks and live and dead eggs from 37 colonies in the upper Great Lakes were evaluated annually for gross anatomical deformities. Deformity rates were higher in all Great Lakes areas evaluated (including Green Bay) than at a reference colony. Hatching and deformity rates were correlated with concentrations of planar PCBs and TCDD-Eqs. PCB concentrations ranged from 3.6 mg/kg in eggs collected from Lake Superior to 7.3 mg/kg in eggs collected from Green Bay; PCB concentration in eggs from the reference colony was 0.8 mg/kg. The authors concluded that there is a causal relationship between the incidence of deformities in cormorants and terns and exposure to planar halogenated compounds measured as TCDD-Eqs or total PCBs in the Great Lakes. The results from this investigation indicated that in double-crested cormorant eggs, the reference area NOAEC for deformity was 0.8 mg/kg PCBs. An estimated LOAEC, based on these NOAEC, is 8 mg/kg PCBs or 380 ng/kg TEQ. These NOAEC and LOAEC values will be used to evaluate the risk of deformity to birds.

Dietary TRV. No studies were found in which the dietary toxicity of PCBs to the Lower Fox River and Green Bay piscivorous bird receptor species was examined.

Therefore, literature pertaining to the toxicity of PCBs to other bird species was reviewed and is summarized below.

Robins (*Erithacus rubecula*) fed a diet containing 5 mg Clophen A50 per day for a period of 11 to 13 days displayed abnormal nocturnal behavior and activity patterns compared to control birds (Ulfstrand and Södergren, 1971). Given a reported average body weight of 18.2 grams (Dunning, 1993), the daily dose would equal 275 mg/kg-BW/day.

In a study on mallard ducks, a dietary concentration of 150 mg/kg Aroclor 1242 resulted in egg shell thinning of 8.9 percent (Haseltine and Prouty, 1980). Based on a food ingestion rate of 0.25 kg/day (Newell *et al.*, 1987), and a body weight of 1.043 kg (EPA, 1993a) the estimated daily dietary dose is 36 mg/kg-BW/day.

Another study of effects on mallards involved feeding Aroclor 1254 to 9 month-old mallard hens at a concentration of 25 mg/kg dry weight for at least 1 month prior to egg laying. No detrimental effects on reproduction or nest attentiveness were observed (Custer and Heinz, 1980). Assuming that the diet was one-third solids, this equates to a wet-weight concentration of approximately 8.3 mg/kg or approximately 2.0 mg/kg-BW/day.

When screech owls were fed Aroclor 1248 in their diet at a concentration of 3 mg/kg for two breeding seasons, the number of eggs per clutch, hatchability, chick malformations, survival, and eggshell thickness were not affected (McLane and Hughes, 1980). Assuming a mean body weight of 0.185 kg (Dunning, 1993) and a food ingestion rate of 0.019 kg/day, calculated using an allometric equation (Nagy, 1987), the resulting dietary dosage is 0.3 mg/kg-BW/day.

Nesting white pelicans captured from the wild were fed 100 mg of Aroclor 1254 daily for 10 weeks in addition to a controlled diet. Following the 10-week exposure period, the birds were stressed for an additional 2 weeks by reducing their food consumption by half. The initial mean body weight of the birds prior to the treatment was 6.2 kg. The mean body weight at the end of the 12-week experimental period was 4.8 kg. Micrograph examination of the livers from the birds in the treatment group indicated a 22 percent increase in hepatocyte size, a significant 25 percent increase in the number of mitochondria, a significant 20 percent fewer cristae per mitochondria, and a 22 percent increase in the number of lysosomes, microbodies, and other membrane-bounded vacuoles (Stotz and Greichus, 1978). Assuming a body weight of 4.8 kg, the exposure concentration is equivalent to 20.8 mg/kg-BW/day.

Yearling male American kestrels were fed prey items (day-old cockerels) containing approximately 33 mg/kg ww of Aroclor 1254 for 62 to 69 days. This dose was converted by the investigators to a daily exposure concentration of 9 to 10 mg/kg-BW/day. Kestrels receiving the treated diet exhibited a significant 22 to 27 percent reduction in sperm concentrations (Bird *et al.*, 1983).

In another study of American kestrels, male and female pairs were fed diets containing 3 mg/kg ww of Aroclor 1248 incorporated into a commercial diet for approximately 20 weeks. Eggs were collected from the pairs 2 to 4 days after egg laying was complete. The eggs collected from the treated pairs of birds exhibited a significant 5 percent reduction in eggshell thickness (Lowe and Stendell, 1991). Assuming a kestrel body weight of 0.200 kg and a food ingestion rate of 0.0154 kg/day (Nice, 1938), this exposure concentration is equivalent to 0.231 mg/kg-BW/day. However, a more recent summary paper by Peakall and Lincer (1996) indicates that PCBs do not cause eggshell thinning except at very high doses that are likely to cause other reproductive toxicological effects as well. Therefore, the LOAEC based on the Lowe and Stendell (1991) study was not used in this risk assessment to evaluate the dietary toxicity of PCBs in birds.

Peakall and Peakall (1973) maintained ring doves on a diet that contained 10 mg/kg Aroclor 1254. They found that reproductive success was dependent on exposure of the female to the PCB compound. Females fed PCB-spiked food were less attentive to their nest and had erratic nesting behaviors which interfered with egg development. Artificial incubation greatly increased the breeding success for these birds. The food concentration of 10 mg/kg is equivalent to 1.12 mg-Aroclor 1254/kg-BW/day using 11.2 g/day as the ingestion rate, and 100 grams as a body mass estimate based on mourning doves (Kenaga, 1973). Similar values were obtained by Peakall *et al.* (1972) for the ringed turtle dove, in which a dietary Aroclor 1254 concentration of 10 mg/kg adversely affected hatching success due to heavy embryonic mortality.

Another study investigated the behavioral component of reproduction in mourning doves given dietary supplements of 0, 10, or 40 mg/kg Aroclor 1254 (Tori and Peterle, 1983). Using the ingestion rate and body weight specified previously (Kenaga, 1973), these doses correspond to 0, 1.12, and 4.48 mg/kg-BW/day. Control doves displayed normal courtship behaviors and patterns. Doves that were fed at the 10 ppm (1.12 mg/kg-BW/day) level spent twice as much time in the courtship phase as the control birds, with only 50 percent completing courtship and nesting. Of the 50 percent that did nest and incubate eggs, nest initiation was significantly delayed, resulting in a delay in egg laying as well. None of the doves on the 40 mg/kg dietary supplement completed the nesting process (Tori and Peterle, 1983).

The TRVs selected for estimating risk of total PCBs to piscivorous birds are a LOAEC of 1.12 mg/kg-BW/day and an estimated NOAEC of 0.11 mg/kg-BW/day. These TRVs were based on studies by Peakall and Peakall (1973) and Tori and Peterle (1983). These TRVs will be compared to calculated daily dose concentrations to assess risk to piscivores from total PCBs.

Polychlorinated Biphenyl (PCB) Toxicity to Mammals. For this BLRA, a LOAEC of 0.13 mg/kg-BW/day (0.72 mg/kg in carp) and a NOAEC of 0.004 mg/kg-BW/day (0.015 mg/kg in carp) (Heaton *et al.*, 1995a) will be used to assess risk to piscivorous mammals in the Lower Fox River and Green Bay.

Mink have been shown to be highly sensitive to the effects of PCBs in their diet, and so they have been identified as an indicator species for water quality and ecosystem health in the Great Lakes (EPA, 1993a). Effects of PCBs on domestic mink in the Great Lakes were noted in the early 1970s (Aulerich *et al.*, 1971, 1973; Platonow and Karstad, 1973), and subsequent studies have demonstrated that domestic mink survival and reproduction are greatly affected by PCBs (Aulerich and Ringer, 1977; Bleavins *et al.*, 1980). Studies have evaluated both the acute toxicity of specific Aroclors (Aulerich and Ringer, 1977, 1980; Bleavins *et al.*, 1980), as well as reproduction and kit survival of mink fed with PCB-contaminated fish (Restum *et al.*, 1998; Aulerich *et al.*, 1973, 1986; Heaton *et al.*, 1995a, 1995b; Hornshaw *et al.*, 1983; Jensen *et al.*, 1977).

PCB toxicity in mammals is highly variable. While some PCBs are extremely toxic, and can cause reproductive failure and produce death in very low levels, others appear to produce few, if any, toxic responses (Eisler, 1986). Toxic responses to PCBs are also highly species-specific. Mink are highly susceptible to PCB toxicity, while closely related mammals, such as the European ferret, are more resistant (Eisler, 1986). Younger mammals appear to be more susceptible to PCB poisoning than adults (Eisler, 1986). Mutagenic, carcinogenic, and teratogenic effects of PCB exposure have been observed, with mutagenic activity appearing to increase with increasing chlorination of the PCB molecule (Eisler, 1986).

Several studies were found pertaining to the dietary toxicity of PCBs to mink, most of which examined effects on reproduction, growth, and survival. Since the mink is the measurement endpoint receptor to be evaluated in this risk assessment, these mink studies were the only studies that were reviewed to derive a TRV for piscivorous mammals.

In a preliminary study to determine the cause of reproductive complications in mink fed Great Lakes fish, adult breeder mink were fed a basal diet supplemented

with 30 mg/kg of PCBs for 6 months (181 days). However, all of the mink became emaciated and died by the end of the experimental period (Aulerich and Ringer, 1977). As a result of the preliminary study, a long-term study was conducted to ascertain the effects of long-term, low-level consumption of PCBs on growth. Mink were fed a basal diet supplemented with 5 and 10 mg/kg of PCBs for a period of approximately 8.5 months. The basal diet plus 10 mg/kg of PCBs resulted in a significant 56 percent decrease in body weight gain after a period of 4 months. Body weight gain was reduced by 39 percent in the 5 mg/kg treatment group, but this reduction was not significant. Both the 5 and 10 mg/kg treatment groups failed to produce offspring; the control group produced 17 live and 8 dead kits. Various degrees of embryotoxicity were observed during necropsy of the treated animals (Aulerich and Ringer, 1977). The 5 and 10 mg/kg doses were converted to daily exposure concentrations based on a food ingestion rate of 0.153 kg/day and a body weight of 0.8 kg, resulting in exposure concentrations of 0.96 and 1.9 mg/kg-BW/day, respectively.

Based on the results of this experiment, another experiment was conducted to determine the effects of long-term consumption of low-level PCBs on reproduction. A concentration of 15 mg/kg of Aroclor 1254 in the diet resulted in a complete inhibition of reproduction and 31 percent adult mortality, compared to 6 percent mortality in the controls. A concentration of 5 mg/kg of Aroclor 1254 resulted in a 95 percent reduction in the number of kits born live; the ratio of live kits to female adults was reduced by 87 percent. However, in an effort to determine the persistence of the impaired reproductive condition, 11 adult females that received 5 mg/kg of Aroclor 1254 for a period of 6 months were placed on a control diet for 1 year. The results indicate that the impaired reproductive performance of these females was not a permanent condition (Aulerich and Ringer, 1977). As indicated above, a concentration of 5 mg/kg in the diet is equivalent to a daily dose of 1.9 mg/kg-BW/day, and the concentration of 15 mg/kg is equivalent to a daily dose of 2.9 mg/kg-BW/day, respectively.

Eight month-old mink fed a basal diet containing 1.0 mg/kg of Aroclor 1254 for a period of approximately 6 months exhibited no mortality or any significant changes in the thyroid, pituitary, adrenal glands, or serum T3 and T4 levels (Wren *et al.*, 1987a). Reproduction and kit development was evaluated under the same test conditions in a separate study (Wren *et al.*, 1987b) by the same investigators. Male fertility and female offspring production were not affected by the 1.0 mg/kg Aroclor 1254 diet. However, growth rate of kits nursed by exposed mothers was significantly reduced. The investigators estimated the daily exposure concentrations to be 0.10 mg/kg-BW/day for males and 0.18 mg/kg-BW/day for females.

When Kubiak and Best (1991) fed mink a liver diet contaminated with PCBs, a concentration of 1.0 mg/kg PCBs (0.19 mg/kg-BW/day) resulted in reproductive impairment and a concentration of 5 mg/kg (0.96 mg/kg-BW/day) resulted in mortality.

In another study, one year-old mink were fed a diet of beef and cereal prepared from cows which had been given 10 consecutive daily oral doses of 1 and 10 mg/kg of Aroclor 1254 dissolved in an olive oil and dairy concentrate (Platonow and Karstad, 1973). The cows did not exhibit any clinical, gross, or histopathological signs of PCB toxicity. The cows were killed 24 hours following the last dose, and the musculature, liver, and kidneys ground and mixed with commercial mink food cereal at a level of 24 percent cereal. The resulting rations containing 0.64 mg/kg (0.12 mg/kg-BW/day) and 3.57 mg/kg (0.68 mg/kg-BW/day) of total PCBs were fed to mink for a period of 160 days. The mink were fed this diet *ad libitum* (as much as the mink wanted to consume) for 2 months prior to the breeding season and continued until 160 days exposure was reached. All 16 mink that were fed 3.57 mg/kg of PCBs died by day 105. Two of the 16 mink that were fed 0.64 mg/kg died by days 122 and 129. The mink exhibited poor appetites, lethargy, and weakness before dying. Some passed tarry feces, indicating gastrointestinal hemorrhaging. At both treatment levels, males survived longer than females.

More recent studies have focused on assessing the risks to feral populations of minks eating PCB-contaminated fish. This has included documenting the widespread presence of PCBs in wild mink tissues (Proulx *et al.*, 1987; Foley *et al.*, 1988), and the potential exposure of wild mink populations to PCBs through a fish diet (Giesy *et al.*, 1994c; Heaton *et al.*, 1995a, 1995b; Tillitt *et al.*, 1996). In a risk assessment conducted on three rivers in Michigan, Giesy and his colleagues found significant risk posed to wild mink populations based upon concentrations of PCBs found in fish prey (sucker, walleye, pike, and salmonids) (Giesy *et al.*, 1994c).

In a recent study, male and female ranch-bred mink were acclimated to a diet consisting of ocean fish scraps, commercial mink cereal, and meat byproducts. Ocean fish scraps made up 40 percent of this diet. Dietary treatment levels were prepared by substituting 10, 20, and 40 percent of the ocean fish scraps with PCB-contaminated carp from Saginaw Bay, Lake Huron. The mean dietary PCB concentrations were 0.015 mg/kg (control), 0.72 mg/kg (10 percent carp), 1.53 mg/kg (20 percent carp), and 2.56 mg/kg (40 percent carp). Groups of 15 mink (3 males, 12 females) were assigned to one of the four treatment groups for a period of 12 weeks. Mink receiving the highest PCB-containing diet (40 percent carp or 0.32 mg/kg-BW/day, as reported by the investigators) exhibited a 42

percent reduction in mean litter size, 86 percent fewer live kits at birth, and no kits surviving beyond 24 hours post-partum. Even mink receiving the 10 percent carp diet (or 0.13 mg/kg-BW/day, as reported by the investigators) exhibited a 67 percent reduction in kits surviving 3 to 6 weeks relative to the control (Heaton *et al.*, 1995a).

In a study of multigenerational effects on mink fed the same Saginaw Bay PCB-contaminated carp, Restum *et al.* (1998) determined that after 6 months of exposure to PCBs, mink on the 1.0 mg/kg PCB diet had significantly decreased kit survival as compared to the controls, and after 18 months of exposure to PCBs, mink in the 0.5 mg/kg and the 1.0 mg/kg PCB diet had significantly decreased kit survival as compared to the controls. Noteworthy in their study was that adverse effects on kit survival were observed even several months after the parents had been placed on the control diet. Further, Restum *et al.* (1998) observed that parental exposure for only 3 months prior to breeding resulted in reduced kit body birth weights in the 0.5 mg/kg diet group (0.1 mg/kg-BW/day), and at 3 to 6 weeks, reduced kit body weight in the 0.25 mg/kg diet group (0.05 mg/kg-BW/day). However, this reduced kit weight was not shown to effect kit survival.

It is interesting to note that for this same endpoint (kit survival), Restum *et al.* (1998) observed a similar LOAEC as Tillitt's and Heaton's (i.e., 0.1 vs. 0.13 mg/kg-BW/day) (Heaton *et al.*, 1995a, 1995b; Tillitt *et al.*, 1996). The NOAEC and LOAEC derived by Heaton *et al.* (1995a) were selected as TRVs and these TRVs are the same as those that were used for the assessment of risk to piscivorous mammals in the Upper Green Bay Risk Assessment (EPA, 2000a).

6.3.3 Dioxins and PCB Congeners

Dioxins Mode of Action

Dioxins, like PCBs, are polychlorinated hydrocarbons, and toxicity is believed to be mediated intracellularly by binding with the aryl hydrocarbon receptor (Ah-R). The resulting PCH-Ah-R complex moves into the cell nucleus, where it will bind to the DNA, and may alter the expression of a number of gene sequences. Many of the observed toxic effects of dioxins (and the coplanar PCBs) are attributable to specific alterations in gene expression (Giesy *et al.*, 1994c).

Ecotoxicity of Dioxins

The effects of tetra-chloro dibenzodioxins (TCDDs) have been thoroughly reviewed by Safe (1990) and by Giesy *et al.* (1994b). Dioxins are not generally acutely toxic to adult organisms, but their long-term accumulation is thought to be expressed chronically, and may ultimately result in death. Key effects important to this BLRA are those causing reproductive dysfunction. The

polychlorinated dibenzodioxins (PCDDs) and polychlorinated dibenzofurans (PCDFs) are thought to cause alterations to developmental endocrine functions (thyroid and steroid hormones), as well as interference in vitamin production, which results in disruption of patterns of embryonic development at critical stages (Giesy *et al.*, 1994b). General population level manifestations of dioxin exposure include adversely affected patterns of survival, reproduction, growth, and resistance to diseases (Eisler and Belisle, 1996). Poor reproductive efficiencies and opportunistic diseases are characteristic of wild animals in these exposed populations of the Great Lakes region (Giesy *et al.*, 1994b).

Ecotoxicity of dioxins and coplanar and PCB congeners to the specific receptor groups for the Lower Fox River and Green Bay are further discussed below.

Dioxin and Furan Toxicity to Aquatic and Benthic Invertebrates. As discussed previously, aquatic invertebrates are presumed to lack the Ah receptor, and, as such, are thought to be relatively insensitive to dioxins. *Daphnia magna* exposed to nominal concentrations of TCDD in water were not affected at concentrations as high as 1,030 ng/L. There are no established dioxin threshold criteria for aquatic invertebrates based on the sources reviewed.

Dioxins have been reported to bioaccumulate in benthic invertebrates to significant concentrations. West *et al.* (1997) exposed *Chironomus tentans* and *Lumbriculus variegatus* to dietary concentrations of TCDD and no toxic effects were observed in full life-cycle tests with either species at tissue residue concentrations up to 9,533 μg-TCDD/kg-lipid. The only evaluated source for sediment criteria that reported a value for dioxin was the EPA (1997d) where the criteria was 0.0039 μg/kg 2,3,7,8-TCDD. This criteria will be used is the assessment of dioxin risk to benthic invertebrates in the Lower Fox River and Green Bay.

Dioxin, Furan, and PCB Congener Toxicity to Fish. TEQ thresholds selected as TRVs include a LOAEC of 84 ng/kg-egg and a NOAEC of 41 ng/kg-egg based on the research of Johnson *et al.* (1998b). These TEQ TRVs will be used to evaluate risk to all fish receptors based on comparison to whole body total TEQ concentrations. Total TEQ concentrations are calculated from individual PCB and dioxin congener concentrations by using TEFs as described earlier in this section. Comparing fish egg TRVs to whole body concentrations is a source of uncertainty that will be further discussed in Section 6.6.

TCDD toxicity to fish embryos is highly species-specific. Elonen *et al.* (1998) recently compared the toxicity of 2,3,7,8-TCDD to egg viability for seven freshwater species, several of which are found in, and are important to, the ecology of the Lower Fox River and Green Bay. Within their study, they found

that fathead minnow and channel catfish had the greatest sensitivity (LC_{50}s of 539 and 644 ng/kg-egg, respectively), while northern pike and zebrafish were the least sensitive (LC_{50}s of 2,460 and 2,610 ng/kg-egg, respectively). When compared to the LC_{50}s reported for lake trout eggs (Walker *et al.*, 1996), the LC_{50} for northern pike is at least two orders of magnitude higher (74:2,460 ng/kg-TCDD/egg).

Studies of toxicity of individual PCB congeners to fish embryos have demonstrated that the most toxic moieties within the PCB mixtures are PCB congener numbers 77, 81, and 126 (Harris *et al.*, 1994; Walker and Peterson, 1991; Zabel *et al.*, 1995). However, each is less toxic than 2,3,7,8-TCDD.

In chinook salmon eggs collected from Lake Michigan, no significant impact on hatching to the fry stage was found where TEQ concentrations were as high as 514 ng/kg ww (Williams and Giesy, 1992). By contrast, Ankley *et al.* (1991) reported that hatching success in chinook salmon also collected from Lake Michigan was consistently reduced at egg TCDD-Eq concentrations greater than 100 ng/kg. More recent controlled laboratory experiments using brook trout have indicated that adult whole body 2,3,7,8-TCDD concentrations of 1,200 ng/kg, the highest dose tested, caused only a slight adverse response (Tietge *et al.*, 1998). This variability can be the result of several factors that influence toxicity; differential toxicity between congeners and 2,3,7,8-TCDD alone, antagonistic or synergistic congener interactions, and species, population, and age-specific responses.

Lake trout eggs have been shown to be particularly sensitive to PCHs (Mac *et al.*, 1985; Mac, 1988; Zabel *et al.*, 1995). For example, for PCB 126, the reported lethal dose to 50 percent of the test population (LD_{50}) for lake trout is 29,000 ng/kg-egg (Zabel *et al.*, 1995) and the LD_{50} for rainbow trout of 74,000 ng/kg-egg (Walker *et al.*, 1991b). Comparatively, testing has indicated that the LD_{50} for 2,3,7,8-TCDD exposed lake trout is 85 ng/kg-egg (Zabel *et al.*, 1995). Further, research by Walker *et al.* (1991b) has indicated that the mortality LOAEC and NOAEC for lake trout exposed to 2,3,7,8-TCDD are 55 ng/kg-egg and 34 ng/kg-egg, respectively. Indicative that lake trout are a sensitive species, a reported LC_{50} for rainbow trout exposed to 2,3,7,8-TCDD is 230 ng/kg-egg (Walker and Peterson, 1991).

Reported 2,3,7,8-TCDD toxicity endpoints for brook trout derived by Walker and Peterson (1994) include an LD_{50} of 200 ng/kg-egg, a LOAEC of 185 ng/kg-egg, and a NOAEC of 135 ng/kg-egg. These results suggest that rainbow trout and brook trout have similar sensitivity to 2,3,7,8-TCDD. However, more recent studies of 2,3,7,8-TCDD toxicity to brook trout by Johnson *et al.* (1998b) suggest

that brook trout may be more sensitive than rainbow trout. Reported 2,3,7,8-TCDD mortality endpoints for brook trout derived by Johnson *et al.* (1998b) include an LD_{50} of 127 ng/kg-egg, a LOAEC of 84 ng/kg-egg, and a NOAEC of 41 ng/kg-egg. As indicated from these data, and additional reported endpoints of 88 ng/kg-egg (LC_{10}) and 184 ng/kg-egg (LC_{90}) the dose response curve was very steep. Steep dose response curves result in no effect and significant effect levels that are close to each other in terms of the toxicant concentrations.

The lowest LOAEC and NOAEC reported were those by Walker *et al.* (1991b), at 55 ng/kg-egg and 34 ng/kg-egg, respectively. The LOAEC and NOAEC values reported by Johnson *et al.* (1998b) were selected as TRVs because the study was multigenerational and the experimental fish were not collected from the field as they were for the study by Walker *et al.* (1991b). These selected TRVs may be overly protective; brook trout have been demonstrated to be more sensitive to PCHs than other Great Lake species found within the Lower Fox River and Green Bay[8] (Elonen *et al.*, 1998). Consequently, those values should be sufficiently protective of most of the fish species in the system.

Dioxin and Furan Toxicity to Birds. Bird TRVs for dioxins and furans were developed for two separate endpoints: reproductive impairment (including egg lethality) and deformity. The NOAEC value of 7 ng/kg ww in eggs will be used as a risk threshold in this BLRA. The effect concentration values for 20 percent (191 ng/kg-egg), 30 percent (308 ng/kg-egg) response, and the NOAEC (7 ng/kg-egg) will be applied in this BLRA to estimate reproductive risks to avifauna. A NOAEC of 38 ng/kg and an estimated LOAEC of 380 ng/kg will be used to assess the risk of deformity to avifauna (Ludwig *et al.*, 1996).

In addition to the two common measures of toxicity, the NOAEC and LOAEC, where available, TEQ concentrations that are associated with a 20 or 30 percent response (i.e., EC_{20} or EC_{30}) may also define a hazard threshold for the receptors of concern. EPA (1989b) TEFs, as adopted for international use by Ahlborg *et al.* (1994) were used to develop the toxicity threshold of 7 ng/kg-egg (NOAEC), and the TEF values developed by Tillitt *et al.* (1991a) were used by Giesy *et al.* (1994b) and Tillitt *et al.* (1992) for the toxicity thresholds of 191 ng/kg (LD_{20}) and 308 ng/kg (LD_{30}) (Table 6-7).

Support for the use of these TEQ avian threshold values also comes from two studies that focused on feral raptor populations exposed to dioxins near bleach

[8] Species evaluated in that study included lake herring, fathead minnow, catfish, white sucker, and northern pike. These species are all reported from the Fox River and are trophically similar to most of the selected receptor species.

kraft pulp and paper mills. Woodford *et al.* (1998) reported on the effects of exposure to polychlorinated hydrocarbons in osprey (*Pandion halieaus*) breeding in north-central Wisconsin. They reported that the ospreys breeding and foraging within the contaminated area ate principally fish, with no apparent effect in hatching or fledging of chicks, relative to reference areas. Measured egg concentrations of TEQ were as high as 171 ng/kg-ww/egg, with most of that contribution coming from TCDD. Since no reproductive effects were observed, the authors suggested a NOAEC equal to, or greater than 136 ng/kg-TEQ/egg.

Similarly, Elliott *et al.* (1996) studied hatchout success for bald eagle nesting pairs on Vancouver Island, British Columbia that were within a 25-km radius of a kraft pulp and paper mill. In that study, the data suggested a NOAEC of 100 ng/kg-TEQ/egg, and a LOAEC of 210 ng/kg-TEQ/egg.[9]

When compared to the avian threshold values for this BLRA, the osprey and eagle NOAECs (136 and 100 ng/kg-egg, respectively) and LOAEC (210 ng/kg-ww/eagle) TEQ concentrations fall near or below the LD_{20} (191 ng/kg TEQ) predicted for birds. Given the relatively small sample size in both studies (osprey n = 4 to 5/year over 4 years; eagles n = 25), it is likely that a 20 percent mortality would not be considered statistically significant. It is interesting to note, however, that in the bald eagle study, the difference in hatching success between nests near pulp mills and the reference areas were 19 percent.

The U.S. Fish and Wildlife Service developed a TEQ NOAEC of 1 ng-TEQ/kg-ww/egg based upon a literature review of TCDD effects on developing chicken embryos (USFWS, 1995). The 1 ng/kg value was derived using an LC_{50} for chicken embryos of 100 ng/kg (Henshel, 1995), and applying an uncertainty factor of 100. Kubiak and Best (1991) developed a TEQ NOAEC for bald eagles of 20 ng/kg ww in the egg, based on TEQ calculation using the H4IIE bioassay. Giesy *et al.* (1995) reviewed the literature for TEQ NOAECs/LOAECs, reporting a range of values from 1 to 114 ng/kg ww in eggs, and derived a NOAEC of 7 ng/kg ww in eggs. This value has been adopted as a "consensus value" and has subsequently been applied for assessing risks to avian reproduction in the Great Lakes (Giesy *et al.*, 1995; Froese *et al.*, 1998). This NOAEC value, 7 ng/kg ww in eggs, will be used as a risk threshold in this BLRA.

Hoffman *et al.* (1996), citing multiple studies in the Great Lakes region, suggest that three PCB congeners (congeners 126, 77, and 105) account for over 90

[9] Both Elliot *et al.* (1996) and Woodford *et al.* (1998) expressed their TEQs using the World Health Organization TEFs in determining their TEQ numbers. However, since most of the toxicity expressed in those two studies was due to 2,3,7,8-TCDD, the numbers are generally comparable.

percent of the PCB congener toxicity in bird eggs. Concentrations of TEQ that have been correlated with effects in feral piscivorous birds have been reported by Giesy *et al.* (1994b) and by Hoffman *et al.* (1996). Table 6-7 presents TEQ concentrations reported in the literature that are specifically associated with lethality (expressed as lethal doses [LD]) to eggs of double-crested cormorants and Caspian terns in feral populations (Giesy *et al.*, 1994b). A plot of those data points, along with the "consensus value" NOAEC (7 ng/kg-egg), is shown on Figure 6-4. A regression on those values shows a very tight correlation (r^2 = 0.923). In addition, Table 6-7 and Figure 6-4 also show points generated from a regression analysis done by Tillitt *et al.* (1992) on egg mortality of double-crested cormorants in the Great Lakes as a function of TEQ. The line equation has been used to generate TEQ concentrations in eggs that would be associated with embryo death. The values for 20 percent (191 ng/kg-egg), 30 percent (308 ng/kg-egg), and the NOAEC (7 ng/kg-egg) will be applied in this BLRA to estimate risks to avifauna.

The research by Ludwig *et al.* (1996) was the source of the TRVs selected for avian deformity. This study was previously cited in this BLRA for the sources of avian deformity TRVs resulting from exposure to PCBs. These PCB NOAEC and LOAEC TRVs were also reported in terms of TCDD-Eq concentrations. These TCDD-Eq concentrations were derived from direct measurement using the H411E rat hepatoma cell bioassay rather than indirect derivation through TEF application. The measured NOAEC TRV is 38 ng/kg and th estimated LOAEC is 380 ng/kg.

Dioxin, Furan, and PCB Congener Toxicity to Mammals. TEQ TRVs for mink have been calculated by Tillitt *et al.* (1996) based on the application of TEFs for carp, on which the mink fed. These TEFs were calculated by using the H4IIE rat hepatoma assay, the same assay that was used to develop TEF applied to birds (Tillitt *et al.*, 1991a). However, while the Tillitt *et al.* TEFs applied to birds have been used in numerous studies around the Great Lakes, those for mink have not been widely applied. For this assessment of PCB risk to piscivorous mammals, it was decided that PCB risk would be determined through the estimation of total PCB risk only, and not congener risk.

6.3.4 Dichlorodiphenyl Trichloroethane (DDT)

Dichlorodiphenyl trichloroethane (DDT) and its principal metabolites, DDD and DDE, are organochlorine compounds that were used as an insecticide for agricultural purposes in the Fox River valley. Commercially sold DDT is a mixture of two isomers: p,p'-DDT where chlorine is in the para position, and o,p'-DDT. The p,p'-DDT isomer represents 80 percent of the mixture with the other 20 percent being the other isomer, o,p'-DDT.

Concerns arose over DDT's use, mostly due to its acute effects in non-target organisms and chronic effects, such as reproductive impairment in birds. In addition to toxic effects, DDT and its metabolites can bioaccumulate in aquatic and avian species. One well-documented response is eggshell thinning in birds, in which the activity of Ca^{2+} ATP-ase systems in the shell gland are affected, thereby interfering with the deposition of calcium in the shell (Lundholm, 1987). Because of the tendency of DDT to magnify in food chains, higher trophic level birds appear to be at greater risk for egg loss due to shell thinning. Eggshell thinning of greater than 20 percent has been associated with decreased nesting success due to eggshell breakage (Anderson and Hickey, 1972; Dilworth *et al.*, 1972).

Another well-defined effect of DDT exposure is inhibition of acetylcholinesterase (AChE) activity. Inhibition of this enzyme results in the accumulation of acetylcholine in the nerve synapses, resulting in disrupted nerve function. Chronic inhibition of 50 percent of brain AChE has been associated with mortality in birds (Ludke *et al.*, 1975).

The effects of DDT on other receptor groups are not as clearly defined. Recent studies indicate that DDT may be an estrogenic mimic, resulting in adverse reproductive effects. Observed effects include feminization and increased female-to-male population ratios for some receptors. Other responses include histopathological changes, alterations in thyroid function and changes in the activity of various enzyme groups (Peakall, 1993).

The TRVs selected for each assessment endpoint are discussed below.

DDT Toxicity to Aquatic and Benthic Invertebrates

A reported 48-hour LC_{50} for daphnids is 4.7 μg/L (Johnson and Finley, 1980). The federal freshwater chronic National Ambient Water Quality Criteria (NAWQC) limit of 1 ng/L will be used to evaluate the effects of DDT and its derivatives to water column invertebrates, because this criteria has been nationally established and is at a concentration lower than concentrations with observed effects to aquatic invertebrates.

Sediment criteria levels have been established for the protection of benthic invertebrates, including TELs established by Environment Canada (Smith *et al.*, 1996). Sediment TELs include: total DDT (7.0 μg/kg dwt), 4,4'-DDE (1.42 μg/kg dwt), and 4,4'-DDD (3.54 μg/kg dwt). These TELs will be used as TRVs to evaluate the effects of DDT and its derivatives to benthic invertebrates in the Lower Fox River and Green Bay.

DDT Toxicity to Fish

Based upon the study by Burdick *et al.* (1964), a LOAEC TRV of 2.95 mg/kg DDT and an estimated NOAEC TRV of 0.3 mg/kg DDT in whole egg tissues will be used to assess risks to whole receptor fish through tissue residue analysis. Selection of these TRVs is discussed below.

DDT is toxic to several fish species, with the greatest mortalities occurring in the younger age groups. The organochlorines accumulate in eggs and can lead to the death of fry as the yolk sac is absorbed (Connell and Miller, 1984). DDT-contaminated feed has caused massive mortalities of sac-fry of brook, rainbow, and cutthroat trout in hatcheries (Connell and Miller, 1984). Rainbow trout and coho salmon have been similarly affected in DDT-contaminated lakes (Connell and Miller, 1984).

Within the ERED database, lake trout were found to be an order of magnitude more sensitive to DDE than the fathead minnow or the mosquito fish (Burdick *et al.*, 1964; Jarvinen *et al.*, 1977). This study with lake trout involved collecting adult female fish from 12 different lakes in New York that had known contamination of DDT (Burdick *et al.*, 1964). Eggs were stripped from these fish and fertilized. A subset of these eggs were incubated and remaining eggs were analyzed for DDT content. The LOAEC for mortality caused by DDT in lake trout was 2.95 mg/kg in whole egg tissues (Burdick *et al.*, 1964). Based on this LOAEC, an estimated NOAEC of 0.3 mg/kg is assumed to be protective of most fish species. These NOAEC and LOAEC values will be used as TRVs for residues of DDE in receptor fish species.

DDT Toxicity to Birds

Three separate endpoints were used as TRVs to assess risk for selected bird receptors from DDT:

1. A LOAEC TRV of 18 mg/kg of DDT equivalents in brain tissue and an estimated NOAEC TRV of 1.8 mg/kg ww were selected as TRVs based primarily on the study by Blus (1996). These TRVs will be used to determine risks to bird receptors from DDT and its metabolites where brain residues are available.

2. A LOAEC TRV of 5.1 mg/kg-egg of DDE and a NOAEC TRV of 3 mg/kg-egg were selected as TRVs based primarily on the study by Wiemeyer *et al.* (1984). These TRVs will be used to assess risks to bird receptors by comparison to egg and whole body residues.

3. A LOAEC TRV of 0.18 mg/kg-BW/day of DDE, and an estimated NOAEC TRV of 0.018 mg/kg-BW/day were selected as TRVs based primarily on the study by Longcore and Samson (1973). These TRVs will be used to assess risks to piscivorous and carnivorous bird receptors by comparison estimated daily doses.

Each of these TRVs is discussed in detail below.

Brain TRV. Toxicity tests on the effects of DDT on birds have measured DDT and its metabolites in bird brains and have found that brain residues are a good predictor of lethality (Blus, 1996). In this review by Blus, numerous studies were examined interpreting the lethal residue levels of DDT and its metabolites in bird brain tissue. When DDT was exposed to birds, lethal brain residues of DDE and DDD generally ranged from less than 1 to 28 mg/kg. However, when birds were exposed to DDE, the DDE brain residue levels were higher, generally greater than 300 mg/kg, and when birds were exposed to DDD, the DDD brain residue levels averaged 172 mg/kg (Blus, 1996).

Based upon these data, a system of DDT equivalents has been developed similar to the TEF system for PCBs and dioxins. Under the DDT toxicity equivalent system, one toxic equivalent is equal to 1 mg/kg DDT, 5 mg/kg DDD, or 15 mg/kg DDE (Blus, 1996). The lethal threshold of DDT equivalents has been established as 18 mg/kg in the brain; this level will be used as a LOAEC TRV. The NOAEC TRV for DDT in bird brains has been estimated as 1.8 mg/kg ww.

Egg TRV. DDT causes both eggshell thinning and reproductive failure in many bird species (Connell and Miller, 1984). Organochlorine pesticide use has caused serious population declines for bald eagles (*Haliacetus leucophalus*), osprey (*Pondion haliaetus*), peregrine falcon (*Falco peregrinus*), European sparrow hawk (*Accipiternisus* sp.), and brown pelican (*Pelicanus occidentalis*) (Connell and Miller, 1984). For bald eagles, one study showed that 10 percent eggshell thinning occurred from 10 mg/kg-DDT ww (Blus, 1996). Research on eggshell thinning has suggested that generally, thinning above 15 or 20 percent for a period of years will result in population level effects (Anderson and Hickey, 1972). Generally, 18 percent thinning is an accepted LOAEC for egg mortality (Longcore and Samson, 1973; Anderson and Hickey, 1972; Dilworth *et al.*, 1972).

There is a tremendous amount of species variation in sensitivity to DDT and its metabolites. The brown pelican appears to be one of the most sensitive bird species with eggshell thinning and decreased reproductive success at concentrations of 3 mg/kg DDE in the egg and total reproductive failure at 3.7 mg/kg. The peregrine falcon, on the other hand, does not experience adverse

effects until DDE concentrations are 30 mg/kg-egg. Black-crowned night herons have a more gradual toxic response curve, with surviving young even at concentrations of greater than 25 mg/kg-egg (Blus, 1996).

Due to DDT's capacity to biomagnify in the food chain, bald eagles, as high trophic level predators, represent the most sensitive species of the Lower Fox River and Green Bay receptors evaluated and, as discussed, embryos represent the most sensitive life stage. Therefore, TRVs based upon bald eagle data were assumed to be protective of all other bird receptors. Research by Wiemeyer *et al.* (1984) on DDE and other organochlorine pesticides in bald eagle eggs indicated that mean 5-year production was near normal when eggs contained DDE residues of 3 mg/kg or less, and production decreased when eggs contained DDE residues of 5.1 mg/kg or greater. These values were selected for use as TRVs where the NOAEC is 3 mg/kg-DDE in eggs and the LOAEC is 5.1 mg/kg-DDE in eggs.

Dietary TRV. Dietary TRV thresholds for birds fed DDE were developed based on the research of Longcore and Samson (1973). These researchers fed DDE at a concentration of 10 mg/kg dwt (equivalent to 3 mg/kg ww) to black ducks for 7 months. As compared to the control group, the ducks fed DDE had significantly thinner eggshells: 22 percent at the equator, 30 percent at the cap and 33 percent at the apex. Although the dosed hens laid more eggs than the controls, hatching success of the dosed hens' eggs were significantly less that the control eggs. To express this dose in units of mg/kg-BW/day, this value was multiplied by the food ingestion rate for the black duck (0.05 kg/day) (EPA, 1993a) and divided by the lowest reported body weight of a black duck (0.9 kg) to yield a LOAEC of 0.18 mg/kg-BW/day. This concentration will be used as a dietary LOAEC, and yields an estimated dietary NOAEC TRV of 0.018 mg/kg-BW/day.

The selected dietary TRVs for birds, when compared to calculated dietary TRVs for another upper trophic level bird species, the American kestrel (*Falco sparverius*), further indicates that the selected TRVs are reasonably protective. A study by Wiemeyer *et al.* (1986), reported mortality in American kestrels receiving 2.8 mg/kg DDE for 14 to 16 months. To express this dose in units of mg/kg-BW/day, this value was multiplied by the food ingestion rate of an American kestrel (0.03 kg/day) and divided by the lowest known body weight of an American kestrel (0.103 kg) to yield a dietary LOAEC exposure concentration of 0.81 mg/kg-BW/day. Further, Lincer (1975) determined that female American kestrels receiving a diet containing 0.55 mg/kg-BW/day DDE produced eggs with shells that were 15.1 percent thinner than experimental controls.

DDT Toxicity to Mammals

Dietary exposure TRVs for assessing risks to piscivorous mammals from exposure to DDT were based upon Giesy *et al.* (1994d), Duby *et al.* (1971), and Aulerich and Ringer (1970). These TRVs are a NOAEC of 19 mg/kg-BW/day and an estimated LOAEC of 191 mg/kg-BW/day. These TRVs will be compared to estimated daily exposure concentrations for piscivorous mammals.

Research has shown that mink are less sensitive to DDT and its metabolites than PCBs (Giesy *et al.*, 1994d; Jensen *et al.*, 1977). Metabolism of DDT is necessary to elicit a toxic response, and are the same as those noted for PCBs, including sluggishness, lack of coordination, vomiting, tremors, and convulsions (Aulerich and Ringer, 1970; Giesy *et al.*, 1994d; Jonsson, 1993).

Several studies have been conducted in which the effects of organochlorine toxicants on mink have been investigated (Giesy *et al.*, 1994d; Jensen *et al.*, 1977; Aulerich and Ringer, 1970; Duby *et al.*, 1971; Frank *et al.*, 1979; Proulx *et al.*, 1987). Generally, population-sensitive endpoints of survival, growth, and reproduction have been evaluated, with reproduction being the most sensitive endpoint.

Aulerich and Ringer (1970) determined 48-hour lethal doses for mink based on intraperineal injections. They determined that a lethal dose of DDD was between 450 and 500 mg/kg BW, a lethal dose of DDT was between 350 and 400 mg/kg BW, and DDE was not lethal up to concentrations of at least 1,000 mg/kg BW. These researchers also determined that no effects to mink were observed when the mink were fed diets containing both 100 mg/kg ww p,p'-DDT and 50 mg/kg ww p,p'-DDD (Aulerich and Ringer, 1970). Similarly, Duby *et al.* (1971) found that mink fed DDT up to 100 mg/kg over two generations had no adverse effects to fertility. Giesy *et al.* (1994d) concluded that 100 mg/kg ww total DDT in fish was an acceptable NOAEC for mink reproduction.

In a review of the literature, only one study reported a toxic DDE dietary value for mink much lower than those previously reported. Gilbert (1969), exposed ranch mink to commercial feed containing field-collected fish, with DDE levels between 0.42 and 0.58. Results showed statistically significant embryonic loss in the test group. However, given that these fish were caught from a contaminated region, other contaminants present may have been responsible for the observed toxicity. This study was not further considered for TRV development.

Another study conducted by the National Cancer Institute and considered as a source of TRVs involved the exposure of rats and mice to high and low concentrations of DDE and DDT in order to assess potential carcinogenicity

(NCI, 1978). Results indicated that the group of female mice dosed with DDE experienced increased mortality. The low dose is equivalent to a daily dose of 12.1 mg/kg-BW/day, but this was not used as a LOAEC TRV because of several drawbacks of the study. Most importantly, the high and low dose concentrations were not consistent throughout the study and control survival was poor for male mice.

TRVs for the dietary risk of consuming DDT and metabolites were based on the research of Giesy *et al.* (1994d), Duby *et al.* (1971), and Aulerich and Ringer (1970). From these studies, 100 mg/kg DDT in food was considered to be NOAEC. Assuming a food ingestion rate of 0.153 kg/day and a body weight of 0.8 kg, 100 mg/kg in food is equivalent to 19.1 mg/kg-BW/day. Based on this NOAEC, the LOAEC was estimated to be 191 mg/kg-BW/day.

6.3.5 Dieldrin

Dieldrin, a chlorinated insecticide, was widely used from the 1950s to the 1970s primarily for soil and seed treatment, mosquito and tsetse fly control, sheep dip, wood treatment, and mothproofing of woolen products. Most uses of dieldrin were banned in 1975, and it is no longer produced in, or imported to, the United States (ASTDR, 1998b). Dieldrin's toxic effects include carcinogenicity, mutagenicity, neurotoxicity, teratogenicity, and reproductive impairment (EPA, 1992b).

Dieldrin toxicity, like the toxicity of numerous other xenobiotics, is believed to be mediated through the creation of reactive oxygen species (Pedrajas *et al.*, 1998). The reactive oxygen species cause oxidative stress in organisms and specific vertebrate effects include lipid peroxidation, DNA damage, and peroxisomal changes (Pedrajas *et al.*, 1998).

The TRVs selected for risk to dieldrin toxicity, for each receptor, are discussed below.

Dieldrin Toxicity to Aquatic and Benthic Invertebrates

The surface water criteria for dieldrin established by the State of Wisconsin in Chapter NR 105 of the Administrative Code is 0.077 μg/L, and this value will be used as a TRV for effects for water column invertebrates. Federal Sediment Quality Criteria for the protection of benthic invertebrates from dieldrin has been established at 11 mg/kg organic carbon, and this criteria will be used as a TRV.

Dieldrin Toxicity to Fish

The whole body bluegill concentration of 3.7 mg/kg, measured in the study by Gakstatter and Weiss (1967), was selected as the LOAEC TRV and from this the

NOAEC TRV was estimated as 0.37 mg/kg. These TRVs will be used to evaluate dieldrin toxicity to the selected fish receptors based on the whole body fish data available in the FRDB.

Summaries of dieldrin toxicity to fish were found within the ERED database and the review article by Peakall (1996); additionally, primary literature sources were reviewed. All sources indicated that few studies have investigated body burdens of dieldrin as related to toxicity; as summarized by Peakall (1996), no tissue residues for fish correlated to toxicity. The range of mortality NOAECs available from the ERED database for measured whole body concentrations in fish was 1 mg/kg for the spiny dogfish to 151 mg/kg for the golden ide. In selecting TRVs, bounded effect levels were given more consideration than unbounded no effect levels. Only three LOAECs were reported in the ERED database: 34 mg/kg for mortality and behavioral endpoints in sheepshead minnow; and with behavioral endpoints 3.8 mg/kg for goldfish, and 3.7 mg/kg for bluegill (Gakstatter and Weiss, 1967).

In the Gakstatter and Weiss study, 60 to 70 small bluegills (*Lepomis macochirus*) and goldfish (*Carassius auratus*), were exposed to 0.03 ppm radioactively-labeled dieldrin for periods ranging from 5 to 19 hours. The fish were then placed in a recovery tank. Over the next 32 days, the fish were analyzed for whole body insecticide content. When the exposures were terminated, the bluegills had severe symptoms of poisoning in the form of equilibrium loss and convulsions. The whole body concentrations of dieldrin at test termination were 3.7 mg/kg.

Dieldrin Toxicity to Birds

Three separate endpoints were used as TRVs to assess risk for birds from dieldrin:

1. A LOAEC TRV of 4.9 mg/kg in brain tissue with an estimated NOAEC of 0.49 mg/kg, based primarily on the study by Stickel *et al.* (1969), will be used for the evaluation of available brain residue data in the FRDB.

2. A LOAEC of 1 mg/kg-egg and an estimated NOAEC of 0.1 mg/kg-egg, based primarily on studies by Wiemeyer *et al.* (1984) and Lockie *et al.* (1969), will be used for the evaluation of egg and whole body residue data in the FRDB.

3. A LOAEC of 1.06 mg/kg-BW/day and an estimated NOAEC of 0.11 mg/kg-BW/day, based primarily on the study by Dahlgren and Linder (1974), will be used for the risk evaluation of estimated dietary exposures of piscivorous and carnivorous birds.

Each of these TRVs is discussed in detail below.

Brain TRVs. Researchers have found that brain residues, weight, and lipid content are remarkably stable, even after death, which makes these residues advantageous to measure (Peakall, 1996). Peakall (1996) reviewed field studies that reported lethal brain tissue residue concentrations for numerous bird species exposed to dieldrin and, potentially, other organochlorines. In this review, some reported mean lethal dieldrin residue levels in brains included 5.4 mg/kg in peregrine falcon, 22.2 mg/kg for the red-winged blackbird, and 11.9 mg/kg for the lesser scaup (Peakall, 1996). For male birds, the 95 percent lower confidence limit for death was 4.9 mg/kg dieldrin in brains. For female birds, the 95 percent lower confidence limit for death was 10.4 mg/kg dieldrin in brains. The Peakall (1996) review concluded that brain residues of 4 to 5 mg/kg indicate that birds are in danger from dieldrin poisoning.

The two major papers that helped establish the brain residue threshold range are both studies on quail brains: Robinson *et al.* (1967) and Stickel *et al.* (1969). Robinson *et al.* exposed quails to diets with 10 to 40 mg/kg dieldrin (as well as single doses of 50, 75, and 100 mg/kg), while Stickel *et al.* exposed quails to 2 to 250 mg/kg dietary doses of dieldrin. Predictably, quail mortality in both studies occurred more rapidly the higher the dose. However, brain tissue residue levels were independent of dose. The critical level of dieldrin in the brain was 10 mg/kg for the Robinson *et al.* study and 4.9 mg/kg for the Stickel *et al.* study. Although the critical brain threshold level was lower in the Stickel *et al.* study, this study also had more variable results and the lowest dose at which quail death occurred was 6.23 mg/kg. Despite its flaws, the 4.9 mg/kg value established by the Stickel *et al.* study is the most widely accepted threshold for lethal brain residues of dieldrin (Peakall, 1996).

Egg TRV. Birds exposed to dieldrin have been shown to suffer reproductive impairment including eggshell thinning, although not as profound as eggshell thinning from DDE exposure. In one study, Winn (1973), related degree of thinning to residue levels. Another study conducted by Wiemeyer *et al.* (1986), showed eggshell thinning, but from a combination of dieldrin, DDT, DDE, and DDD (Peakall, 1996).

Wiemeyer *et al.* (1984) investigated the reproductive success in bald eagles as related to organochlorine pesticide, PCB, and mercury residues in eggs. While the investigators stated that their results indicated a higher correlation with DDE than with the other contaminants, they nonetheless concluded that bald eagle eggs containing more than 1 ppm dieldrin may be at reproductive risk. This LOAEC was based upon numerous other studies that cited adverse reproductive

effects on several species of birds. In particular, a study with golden eagles reported egg breakage that was correlated with amounts of dieldrin in the eggs exceeding 1 ppm (Lockie *et al.*, 1969).

Wiemeyer reported other reproduction NOAECs of dieldrin in eggs, including: 0.94 mg/kg for the brown pelican, 17.5 mg/kg for gallinules, and 11.8 mg/kg for homing pigeons (Wiemeyer *et al.*, 1984). In the Peakall review (1996) (discussed above), deleterious reproductive effects were observed with whole egg residue levels of dieldrin concentrations of 45.2 to 92.5 mg/kg for mallards (egg production and fertility). Based on the reported results, dieldrin TRVs selected for use in assessing risk to birds in the Lower Fox River and Green Bay were a LOAEC of 1 mg/kg in eggs and an estimated NOAEC of 0.1 mg/kg.

Dietary TRV. The dietary threshold TRVs were obtained from reviewing research by Dahlgren and Linder (1974) in which effects of dieldrin to three generations of pheasants were monitored. These researchers investigated several endpoints including mortality to both adults and offspring, reproduction (egg production, fertility, and hatchability), weight, and behavioral changes (hand catching and depth perception). Adult hens and cocks were fed weekly doses of 0, 6, or 10 and 0, 4, or 6 mg, respectively, for a period of 16 and 17 weeks.

The lowest common dose of dieldrin administered to hens and cocks (6 mg) caused increased mortality in the adults, and decreased survival for chicks at 4 weeks post-hatch. Aberrant behavior was noted in offspring from dieldrin administered to either parent. However, the results for the reproductive endpoints (i.e., egg production, fertility, hatchability, and viability of the embryo at the time of hatching) were too erratic to be conclusive. Additionally, there were no statistically significant differences observed in either the adult's or offspring's weight. This paper further stated that the concentration of 6 mg per week corresponds to 20 mg/kg in the diet. Assuming a pheasant body weight of 1.3 kg and using the allometric equations provided by the EPA *Wildlife Exposures Factors Handbook* (EPA, 1993a), a LOAEC TRV of 1.06 mg/kg-BW/day was calculated and will be used to assess dieldrin risks to piscivorous birds. The NOAEC of 0.11 mg/kg-BW/day is an estimated TRV based on the calculated LOAEC and will also be used to assess risk.

This LOAEC is supported by other research examining dietary dose and reproductive endpoints. Adverse reproductive effects were observed in pheasants exposed to concentrations of 25 and 50 mg/kg dieldrin (4.3 and 8.75 mg/kg-BW/day) in their diet (Genelly and Rudd, 1956). Hungarian partridges exposed to 3 mg/kg dieldrin (0.5 mg/kg-BW/day) in their diet for 90 days during the breeding season resulted in decreased egg production and hatchability. Heath *et*

al. (1972) reported an acute LC_{50} of 6 mg/kg-BW/day for the bobwhite quail. Chickens exposed to 5 mg/kg dieldrin (0.9 mg/kg-BW/day) in their diet showed no effects on egg production or hatchability.

Dieldrin Toxicity to Mammals

Selected TRVs for piscivorous mammals, based upon the study by Harr *et al.* (1970a, 1970b), are a LOAEC of 0.018 mg/kg-BW/day and a NOAEC of 0.009 mg/kg-BW/day. These TRVs will be used to evaluate the risk of the estimated dietary dieldrin exposure to piscivorous mammals.

In general, mammals are somewhat more sensitive to dieldrin poisoning than are birds (Peakall, 1996). In mammals, dieldrin is rapidly absorbed from the GI tract upon ingestion. Dieldrin is metabolized by the mixed function oxidase (MFO) enzyme system in the liver. Metabolites are then transported to various tissues in the body, including the brain, blood, liver, and adipose tissue. Dieldrin has been shown to cross the placental barrier, and for that reason has been studied for its teratogenic properties and reproductive effects.

In a review of dieldrin toxicity by Peakall (1996), lethal doses observed in mammals do not vary appreciably; however, the toxicity tests reviewed did not include exposures to mink specifically. Most chronic toxicity data has been collected on mammals other than mink; therefore, this section also reviews toxicity to other mammals.

In the Harr *et al.* (1970a) study, rats of varying ages (28 to 750 days old) were exposed to dietary concentrations of dieldrin ranging from 0.08 to 40 mg/kg. Adult rats and pups were examined for clinical signs of toxicosis and then killed. Various tissue and stomach milk curd analyses (on the pups) were conducted. Exposures to 2.5 mg/kg (0.11 mg/kg-BW/day) and greater resulted in nonspecific neural and vascular lesions, cranial edema, and convulsions; no effects were noted at dietary concentrations of 1.25 mg/kg (0.058 mg/kg-BW/day) or less. Reproductive effects on rats from this study were published separately (Harr *et al.*, 1970b). These results indicated that a concentration of 0.31 mg/kg (0.018 mg/kg-BW/day) was the lowest concentration that resulted in adverse reproductive effects, including reduced pup survival and conception rate; no adverse effects to reproduction were found at an exposure concentration of 0.16 mg/kg (0.009 mg/kg-BW/day) (Harr *et al.*, 1970b). The researchers calculated the highest dietary dieldrin level consistent with normal reproductive values to be 0.24 mg/kg ppm (0.014 mg/kg-BW/day).

Acute toxicity to mink as measured by intraperineal injections of dieldrin showed that concentrations of 75 to 100 mg/kg were lethal within a few hours (Aulerich

and Ringer, 1970). These researchers also observed the survival and reproductive effects of dieldrin fed to mink at concentrations of 2.5 mg/kg and 5 mg/kg. Results indicated that effects of dieldrin ranged from some mortality occurring from exposure to 2.5 mg/kg, to no effects to survival and reproduction occurring from exposure to 5 mg/kg. From these results, the authors reported that the long-term effects of dieldrin on reproduction were inconclusive (Aulerich and Ringer, 1970).

In a 128-week study, no adverse effects were noted in mice exposed to 0.1 and 1 mg/kg dieldrin (0.013 and 0.13 mg/kg-BW/day) in their diet (Walker *et al.*, 1972). In a similar study, no effect on mortality or longevity was observed over three generations of rats exposed to 2.5, 12.5, or 25.0 mg/kg dieldrin in their diet, but an increase in the liver/body weight ratio was observed at all concentrations (Treon and Cleveland, 1955). Another chronic study resulted in no significant pup mortality when mice were fed a dose of 0.33 mg/kg-BW/day of dieldrin (Virgo and Bellward, 1975).

Because the results reported for mink were inconclusive (Aulerich and Ringer, 1970), selected TRVs were based on the rat reproduction studies by Harr *et al.* (1970b); selected TRVs for assessing risk to piscivorous mammals were a LOAEC of 0.018 mg/kg-BW/day and a NOAEC of 0.009 mg/kg-BW/day.

6.3.6 Arsenic

Eisler (1988a) presents the following points agreed upon by most investigators: 1) arsenic may be absorbed by ingestion, inhalation, or permeation of the skin or mucous membranes; 2) cells accumulate arsenic by using an active transport system normally used in phosphate transport; 3) arsenicals are readily absorbed after ingestion, most being rapidly excreted in the urine during the first few days; 4) the toxicity of arsenicals conforms to the following order from greatest to least toxicity: arsines > inorganic arsenites > organic trivalent compounds (arsenoxides) > inorganic arsenates > organic pentavalent compounds > arsonium compounds > elemental arsenic; 5) solubility in water and body fluids appear to be directly related to toxicity; and 6) the mechanisms of arsenical toxicity differ considerably among arsenic species, although signs of poisoning appear similar for all arsenicals.

The primary mechanisms of inorganic trivalent arsenic toxicity begins with its initial metabolism to the trivalent arsenoxide form, followed by its subsequent reaction with sulfhydryl groups of tissue proteins and enzymes, which inhibits oxidative degradation of carbohydrates and decreases cellular ATP (Eisler, 1988a). Inorganic pentavalent arsenic does not react as readily with sulfhydryl groups. Inorganic trivalent arsenic interrupts oxidative metabolic pathways and sometimes

causes morphological changes in liver mitochondria. Methylation greatly reduces the toxicity of inorganic arsenic (both trivalent and pentavalent) and is usually the major detoxification mechanism (Eisler, 1988a).

In studies on vertebrate species, arsenic has been shown to have teratogenic effects such as developmental malformations (Eisler, 1988a). While arsenic has been found to accumulate in aquatic organisms, it has not been found to biomagnify up the aquatic food chain (Eisler, 1988a). Inorganic arsenic is toxic to fish and can concentrate in tissues, whereas organic arsenic is generally rapidly eliminated by fish (Eisler, 1988a).

Arsenic Toxicity to Aquatic and Benthic Invertebrates

The surface water criteria for arsenic established by the State of Wisconsin in Chapter NR 105 of the Administrative Code is 152.2 μg/L, and this value will be used as a TRV for effects for water column invertebrates. For determination of risks to benthic infauna, the ARCS SEC (EPA, 1996a) sediment value of 12.1 mg/kg dwt arsenic will be used as a TRV. This specific value is the Effect Range Low (ERL) derived from a 14-day chronic test using *Hyalella azteca*.

Arsenic Toxicity to Fish

The TRVs selected to assess whole body arsenic concentrations in fish were a mortality NOAEC of 0.5 mg/kg arsenic and an estimated LOAEC of 5 mg/kg based on the work of Barrows *et al.* (1980). Derivation of these TRVs is described below.

Diminished growth and survival were reported in immature bluegills (*Lepomis macrochirus*) when total arsenic residues in muscle were greater than 1.3 mg/kg fresh weight in juveniles or greater than 5 mg/kg in adults (Eisler, 1988a). Barrows *et al.* (1980) reported a NOAEC for bluegill mortality of 0.52 mg/kg arsenic. Endpoints for rainbow trout include a mortality LOAEC of 4.7 mg/kg arsenic (Dixon and Sprague, 1981), a growth LOAEC of 3.1 mg/kg arsenic oxide (Cockell and Hilton, 1988), and a growth LC_{12} of 17.9 mg/kg arsenic oxide (Cockell and Hilton, 1988). The mortality LOAEC was derived based on a study of acclimation of rainbow trout exposed to arsenic for up to 21 days (Dixon and Sprague, 1981). Results indicated that rainbow trout acclimated to arsenic for as little as 7 days, representing a body burden of 4.7 mg/kg ww, and had significantly lower LC_{50}s than control fish that were not pre-exposed to arsenic. Therefore, although a LOAEC was derived from this study, based on the experimental design, this endpoint is not useful for deriving a TRV. Cockell and Hilton (1988), exposed juvenile rainbow trout to arsenic for 8 weeks and recorded growth and mortality, but only growth was evaluated for significant differences.

The NOAEC derived for bluegill, 0.52 mg/kg, was based on a study examining bioconcentration (Barrows *et al.*, 1980). This investigation did not determine a LOAEC (i.e., the NOAEC was unbounded); however, an estimated LOAEC would be 5.2 mg/kg. This estimated LOAEC is within the range of reported low effect levels for rainbow trout: 3.1 to 17.9 mg/kg (Cockell and Hilton, 1988).

6.3.7 Lead

The toxic effects of lead on aquatic and terrestrial organisms are extremely varied and include mortality, reduced growth and reproductive output, blood chemistry alterations, lesions, and behavioral changes. However, many effects exhibit similar trends in their toxic mechanism. Generally, lead inhibits the formation of heme which adversely affects blood chemistry, and it can accumulate in tissues (Eisler, 1988b). At high concentrations, near levels causing mortality, marked changes to the central nervous system occur prior to death (Eisler, 1988b).

Lead Toxicity to Aquatic and Benthic Invertebrates

The surface water criteria for lead established by the State of Wisconsin in Chapter NR 105 of the Administrative Code is 49.42 μg/L, and this value will be used as a TRV for effects for water column invertebrates. For determination of risks to benthic infauna, the ARCS SEC (EPA, 1996a) sediment value of 34.2 mg/kg dwt lead will be used as a TRV. This specific value is the Threshold Effect Level (TEL) derived from a 14-day chronic test using *Hyalella azteca*.

6.3.8 Mercury

Ecotoxicity of Mercury

Mercury is a toxicant which potentially exerts acute, chronic, and subchronic effects on all organisms within the Fox River system. Mercury adversely affects reproduction, growth, behavior, osmoregulation, and oxygen exchange in aquatic organisms. It impairs developmental processes and reproduction in fish, amphibians, birds, and mammals (Birge *et al.*, 1981). Organomercury compounds, especially methylmercury (MeHg), are more toxic than inorganic forms. Methylmercury is among the strongest known inhibitors of mitotic cell division, and may also produce chromosomal aberrations, polyploidy, somatic mutations, and teratogenesis (Birge *et al.*, 1981).

Elemental mercury and mercury-containing compounds have no known normal metabolic function in living organisms. Elevated levels of mercury in living organisms represent contamination from natural and anthropogenic sources. Mercury in the environment occurs in both inorganic (e.g., HgOH) and organic (e.g., MeHg) forms. Both forms bioaccumulate, but methylmercury most often

biomagnifies into upper trophic levels. Up to 90 percent of mercury found in biological tissues is methylmercury.

Mercury compounds bind with sulfhydryl groups and interfere with thiol metabolism in organisms, causing inhibition or inactivation of proteins containing thiol ligands and, ultimately, leading to mitotic disturbances (Das *et al.*, 1982).

Eisler (1987) reports that juvenile life stages are the most susceptible to acute effects of mercury exposure. In fish, acute exposure results in impaired respiration, sluggishness, and loss of equilibrium (Armstrong, 1979). At comparatively low concentrations in birds and mammals, mercury adversely affects growth and development, behavior, motor coordination, vision, hearing, histology, and metabolism. In mammals, methylmercury irreversibly destroys the neurons of the central nervous system. In mammals, the fetus is the most sensitive life stage to mercury (Eisler, 1987).

Like PCBs, mercury has been a persistent problem in Great Lakes wildlife, and often co-occurs in wildlife tissue with PCBs. This has included invertebrates (Ciborowski and Corkum, 1988), fish (McMurty *et al.*, 1989; Wiener *et al.*, 1990; Grieb *et al.*, 1990; Giesy *et al.*, 1995), birds (Frank and Holdrinet, 1975; Bowerman, 1993; Giesy *et al.*, 1995; Wiemeyer *et al.*, 1984; Bishop *et al.*, 1995) and mammals (Aulerich *et al.*, 1974; Wobeser *et al.*, 1976a, 1976b).

It is beyond the scope of this risk assessment to discuss in detail the physical and biological processes in sediment and water that are involved in methylmercury formation in freshwater environments. Excellent reviews on the topic may be found in reports by Winfrey and Rudd (1990), Wiener *et al.* (1990), Meili (1991), Meili *et al.* (1991), and Matilainen *et al.* (1991). The sections below will discuss uptake, biomagnification, and toxicity of mercury to fish, birds, and mammals, with special attention to the targeted receptor organisms for the Fox River.

Mode of Bioaccumulation and Biomagnification

The uptake and trophic transfer of inorganic and organic mercury is dependent upon the chemical speciation of the mercurous compounds. Predicting speciation in aquatic environments is difficult. Speciation is regulated by a variety of environmental variables including mercury loadings, microbial activity, nutrient content, pH and redox, suspended sediment load, sedimentation rates, alkalinity, humic content of water, and presence of complexing agents (e.g., sulfides) (Björnberg, 1988). The most toxic form of mercury, methylmercury, can form under natural conditions through both abiotic and biotic-mediated reactions, although abiotic processes are more rare. Bacterial synthesis of methylmercury

from inorganic mercury compounds present in the water or sediments is the major source of this molecule in aquatic environments (Matilainen *et al.*, 1991). Methylmercury is the most hazardous mercury species due to its high stability, its lipid solubility, and its ability to penetrate membranes in living organisms.

Several authors have noted negative relationships between organic carbon and mercury bioaccumulation in fish (Grieb *et al.*, 1990), which would suggest that methylmercury uptake and transfer follows equilibrium partitioning theory. However, methylmercury is highly water soluble and has a K_{ow} that varies dependent upon the pH and ionic strength of water (Major *et al.*, 1991), but is approximately 1.7 under conditions observed in Lower Fox River sediments. Based upon its low K_{ow}, methylmercury would be predicted to principally enter food chains through water exposure and have minimum trophic transfer through food webs, yet the opposite occurs, demonstrating that food web transfer is the most important pathway for birds and mammals (Meili, 1991). One proposed mechanism for food chain biomagnification is in the strong affinity that methylmercury has for the sulfhydryl groups in organic molecules (e.g., proteins) (Meili, 1991). In one study, the formation of methylmercury-sulphur ligands with anthropogenic compounds was shown to increase the uptake and formation of lipophilic complexes in brown trout (Gottofrey and Tjalve, 1991).

Since methylmercury biomagnifies, the top-level predators generally contain the highest concentrations. In organisms near the top of the aquatic food chain, such as piscivorous fish, almost all mercury accumulates in the methylated form, with methylmercury representing over 90 percent of the mercury in fish (Huckabee *et al.*, 1979). Methylmercury is virtually the exclusive form of mercury found in birds and mammals, and exposure is principally a result of the consumption of prey containing methylmercury. Therefore, dietary accumulation of mercury is considered to be the most important pathway of exposure for these organisms.

Mercury Toxicity to Aquatic and Benthic Invertebrates. For freshwater crustaceans, reported LC_{50}s range from 1.3 to 10 μg/L of mercury (Eisler, 1987). From the AQUIRE database, 96-hour LC_{50}s for freshwater invertebrates were as follows: caddisflies (1,200 μg/L), freshwater prawn (4.8 μg/L), amphipod (10 μg/L), and chironomids (220 to 880 μg/L).

The surface water criteria for mercury established by the State of Wisconsin in Chapter NR 105 of the Administrative Code is 0.44 μg/L; well below concentrations reported to cause toxicity. This value will be used as a TRV for effects to water column invertebrates. Risks to benthic invertebrates will be evaluated using the Environment Canada TEL of 0.17 mg/kg dwt (Smith *et al.*, 1996).

Mercury Toxicity to Fish. A LOAEC of 2.37 mg/kg in whole fish tissue and a NOAEC of 0.25 mg/kg were selected as TRVs based primarily on the study by Friedmann *et al.* (1996). These TRVs will be used to assess risks to all selected fish receptors.

Mercury uptake and toxicity have been extensively studied in fish species common to the Lower Fox River. There are numerous references to elevated mercury and methylmercury levels in suckers, large mouth bass, yellow perch, pike, and walleye (Laarman *et al.*, 1976; McMurtry *et al.*, 1989; Cope *et al.*, 1990; Grieb *et al.*, 1990; Wren, 1991).

The method of uptake and bioaccumulation is dependent upon the predominant form (inorganic vs. organic mercury) and the relative concentrations in water or sediment. In lakes with elevated aqueous concentrations of mercury, uptake across the gill membrane is predominant, whereas in systems with low mercury in the water column the food chain route predominates (Boudou, 1991). The uptake of mercury by benthic infauna, which serve as prey species for higher organisms such as fish, has been previously documented (Ciborowski and Corkum, 1988; Odin *et al.*, 1995).

Nearly all (95 to 99 percent) of the mercury contained in fish is methylmercury, even though little of the total mercury in the waters and sediments in freshwater ecosystems exists as methylmercury. Fish do not methylate inorganic mercury within their tissues, but some methylation occurs in their gut. Inorganic mercury is absorbed less efficiently across the gut and gills and is eliminated more rapidly. Methylmercury rapidly penetrates fish gut and gills, binds to red blood cells and is rapidly transported to all organs, including the brain, and is eliminated very slowly relative to its rate of uptake (Beyer *et al.*, 1996).

Mercury can cause a range of lethal and sublethal responses in fish. Studies demonstrate that acute toxicity can occur at less than 0.1 μg/L in a water-only exposure with brook trout, even though EPA's acute water quality criterion is 2.4 μg/L (Eisler, 1987). Mercury residues in lethally exposed fish ranged from 26 to 69 mg/kg ww in the liver, and 5 to 7 mg/kg ww in whole body tissue (Armstrong, 1979).

Sublethal adverse affects of mercury include altered reproduction, growth, behavior, and metabolism. As a teratogen, mercury affects developmental processes, causing a variety of abnormalities that include skeletal deformities, abnormal organ development, optic malformation, and often retardation of development (Weis and Weis, 1991). Maternal-transferred mercury has also been observed to cause developmental abnormalities (Birge, 1979; McKim *et al.*, 1976). Additional sublethal effects include alterations of respiration, locomotion, social

organization, reproduction, feeding, and predator avoidance (Henry and Atchinson, 1991).

Wiener and Spry (1996) summarized a number of studies detailing the tissue concentrations of total mercury in fish related to methylmercury toxicity. Whole body concentrations associated with toxic effects are 3 mg/kg or greater in brook trout, 4 mg/kg in rainbow trout, and 3 mg/kg in walleye (brain tissue).

Friedmann *et al.* (1996) exposed juvenile (8.5 months old) walleye to diets containing methylmercury for 6 months. The two dietary concentrations, 0.1 and 1.0 mg/kg, were deemed to be environmentally relevant and resulted in body burdens of 0.25 and 2.37 mg/kg, respectively. Results indicated that in terms of gonad development, males from the high-dose group had significant multifocal cell atrophy and seriously disrupted architecture. Additionally, growth was significantly reduced in males exposed to the high-dose diet. Decreased growth during the summer can increase the susceptibility of young-of-year fish to predation and winter kill, and, therefore, mortality. Based on these investigations, a methylmercury LOAEC of 2.37 mg/kg and a NOAEC of 0.25 mg/kg were selected as TRVs for whole fish.

Mercury Toxicity to Birds. Methylmercury has been detected in a large number of avian species in the Great Lakes, including piscivorous species such as the common loon (Meyer *et al.*, 1995), herring gulls (Frank and Holdrinet, 1975), and eagles (Bowerman, 1993; Giesy *et al.*, 1995; Wiemeyer *et al.*, 1984), as well as insectivorous birds such as tree swallows and red-winged blackbirds (Bishop *et al.*, 1995). In birds, the principal route of methylmercury exposure is through trophic transfer.

Numerous studies have shown that methylmercury is more toxic to birds than inorganic mercury. Mercury poisoning in birds is characterized by muscular incoordination, falling, slowness, fluffed feathers, withdrawal, hyperactivity, and hypoactivity. Mercury sublethal effects on birds include liver and kidney damage, neurobehavioral effects, reduced food consumption, weight loss, reduced growth and development, and reproductive impairment. Reproductive and behavioral effects can occur at dietary concentrations well below those that cause overt effects (EPA, 1997e).

Methylmercury acute oral toxicity studies yielded the following LD_{50} values (EPA, 1997e): 2.2 to 23.5 mg/kg for mallards (*Anas platyrhynchos*), 11 to 27 mg/kg for quail (*coturnix*), and 28.3 mg/kg northern bobwhite (*Colinus virginianus*).

Three separate endpoints were used as TRVs to assess risk to Fox River receptor bird species from mercury:

1. A LOAEC TRV of 2 mg/kg mercury in liver tissue and an estimated NOAEC TRV of 0.2 mg/kg. These TRVs were based primarily on the study by Fimreite (1971) together with reviews by Scheuhammer (1987) and EPA (1997e).

2. A LOAEC TRV of 0.8 mg/kg mercury in egg and an estimated NOAEC TRV of 0.08 mg/kg mercury in egg were selected, based primarily on the studies of Heinz (1979). These TRVs will be used to assess risks to all bird receptors from mercury based on comparison to whole body or egg residues.

3. A LOAEC TRV of 0.078 mg/kg-BW/day and an estimated NOAEC TRV of 0.008 mg/kg-BW/day was selected to evaluate the dietary risks to piscivorous and carnivorous bird receptors based on dietary intake of methylmercury. These TRVs are based primarily on the studies of Heinz (1974, 1976a, 1976b, 1979) and Heinz and Locke (1975).

Each of the above TRVs is discussed in detail below.

Liver TRV. Mercury residues associated with toxic effects in birds have been reviewed by Scheuhammer (1987), Eisler (1987), and Heinz (1979), and have been summarized by EPA in the Mercury Study Report to Congress (EPA, 1997e).

Eisler (1987) reported that liver mercury concentrations in birds experimentally killed by methylmercury ranged from 17 mg/kg dwt in red-tailed hawks (*Buto jamaicsis*) to 70 mg/kg dwt in jackdaws (*Corvus monedula*), with intermediate concentrations detected in ring-necked pheasants, kestrels (*Falco tinnunculus*), and magpies (*Pica pica*). As reported by Scheuhammer (1987), adult pheasants fed methylmercury in their diet for 12 weeks had liver mercury residues of 2 mg/kg without exhibiting toxic effects, but there was a decrease in hatching success, suggesting generational effects (Fimreite, 1971). Heinz (1979) conducted a multigenerational study where mallards were fed methylmercury, and accumulated approximately 1.5 mg/kg in the liver without adverse effect. Offspring, however, did exhibit aberrant behavior.

The endpoint of reduced hatching success at liver concentrations of 2 mg/kg observed by Fimreite (1971) was considered to be a potentially ecologically

relevant effect, and was selected as a liver LOAEC TRV. Based upon this LOAEC, an estimated NOAEC of 0.2 mg/kg in liver tissue will also be used as a TRV.

Egg Tissue TRV. Reproduction is the most sensitive endpoint identified for methylmercury toxicity to birds and reproductive impairment has been a well evaluated endpoint for mercury toxicity.

Reduction in egg laying and nest site territory fidelity were associated with mean mercury concentrations ranging from 2 to 3 ppm in loon eggs (Barr, 1986). Adverse effects on hatching and fledging were observed when mercury concentrations in common tern eggs exceeded 3.6 mg/kg (Fimreite, 1979).

Bishop *et al.* (1995) reported total mercury NOAECs ranging from 0.22 to 1.0 mg/kg in eggs of common tern, northern gannet (*Sula bassanus*), osprey (*Pandion haliaetus*), Forster's tern, black skimmer (*Rhynchops niger*), Caspian tern, least tern, and herring gull.

Fimreite (1979) reported hatching success LOAECs ranging from 1.3 to 2 mg/kg-egg following mercury exposure in white-tailed sea eagles (*Haliaeetus albicilla*), common loon, and several seed-eating species. This same concentration range for mercury in eggs was also noted to cause reduced hatchability in ring-necked pheasants (Borg *et al.*, 1969). However, earlier research by Fimreite (1971) on mercury toxicity to pheasants indicated that adverse reproductive effects (decreased hatchability and increased embryo mortality) occurred when unhatched eggs contained 0.5 to 1.5 mg/kg mercury.

Wiemeyer *et al.* (1984) collected and analyzed over 100 bald eagle eggs from around the United States and concluded that mercury concentrations of greater than 0.5 mg/kg in eggs may be sufficient to cause adverse effects on reproduction. However, the correlation between reduction in mean 5-year production rate for eagle nests was confounded by the co-occurrence of DDE, which was much more strongly correlated to adverse breeding effects in eagles.

The EPA has stated that the most extensive research on mercury toxicity has been conducted by Heinz at the Patuxent Wildlife Research Center (Heinz, 1974, 1976a, 1976b, 1979; Heinz and Locke, 1975). A three-generation study of effects of methylmercury exposure in mallards indicated that the lowest concentration administered in feed (0.5 mg/kg) caused adverse reproductive effects. The dietary concentration correlated to an egg concentration of 0.8 mg/kg. This LOAEC of 0.8 mg/kg-egg was selected as a TRV and from this a NOAEC TRV of 0.08 mg/kg-egg was estimated. These TRVs will be used to assess mercury residues in both egg and whole body tissues of all bird receptors.

Dietary TRV. Numerous studies have been conducted with captive birds administered methylmercury and mercury noting lethal and sublethal effects. Starlings fed 0.1 mg/kg-BW/day of mercury for 8 weeks were observed to have kidney lesions (Nicholson and Osborn, 1984). Zebra finches fed a diet containing mercury at 1.7 mg/kg-BW/day suffered from neurological impairment and death (Scheuhammer, 1987).

Juvenile goshawks fed chicken muscle and liver that contained 10 to 13 mg/kg methylmercury survived for only 30 to 47 days (Borg *et al.*, 1970). Prior to death, symptoms in the birds included lack of balance, weakness in extremities, and lack of coordination that progressed to an inability to walk or fly.

Adult ring-necked pheasants fed a diet of 4.2 mg/kg mercury experienced reproductive impairment including reduced hatching success and decrease in embryo survival (Heath *et al.*, 1972). In another similar study, adult pheasants fed a diet of 2 to 3 mg/kg of methylmercury for 12 weeks were generally unaffected, but the dosage caused an increase in eggs without shells, a decrease in egg weight, decreased hatchability, and an increase in unfertilized eggs (Fimreite, 1971). Black ducks fed 3 mg/kg dry weight of methylmercury showed reproductive impairment including reduction in clutch size and egg hatchability, in addition, duckling survival was impaired, and lesions were found in nerve tissue in the ducklings (Finley and Stendell, 1978).

As previously discussed, research on mercury toxicity by Heinz and coworkers during the 1970s at the Patuxent Research Center have been some of the most extensive studies performed. When the multigenerational data on mallard ducks initially exposed to mercury dietary doses of 0.5 and 3 mg/kg were combined, it was found that all generations experienced adverse effects from the lowest dose group (0.5 mg/kg) (Heinz, 1974, 1976a; Heinz and Locke, 1975).

In the 0.5 mg/kg dosage group, effects included an increased number of eggs laid outside the box, reduced duckling survival, reduced duckling growth, reduced egg viability, and thinner eggshells (Heinz, 1976b). Also, aberrant behavioral effects were noted in third-generation mallards exposed to 0.5 mg/kg mercury in the diet.

The dietary dose of 0.5 mg/kg was selected as a dietary TRV based on the work of Heinz (1976b, 1979). This LOAEC TRV was converted into daily dose units by assuming a food ingestion rate of 156 g/kg-BW/day (EPA, 1997e), which resulted in a LOAEC TRV of 0.078 mg/kg-BW/day for reproduction and behavior. The estimated NOAEC dietary TRV is 0.008 mg/kg-BW/day. These TRVs were used to assess risk to piscivorous and carnivorous birds by comparison to estimated daily exposure.

Mercury Toxicity to Mammals. A LOAEC TRV of 0.21 mg/kg-BW/day and a NOAEC of 0.084 mg/kg-BW/day were selected as TRVs for mink, based primarily on the studies by Wobeser *et al.* (1976a, 1976b) together with the GLWQI-established subchronic oral doses for mammalian wildlife. These TRVs will be used to assess risk to piscivorous mammals based on comparison to estimated daily exposure concentrations.

Methylmercury in the diet is absorbed with high efficiency in the vertebrate digestive tract, associating rapidly with sulfhydryl-containing molecules in blood. These mobile complexes transport methylmercury to tissues and organs and may facilitate its movement across cell membranes. There is evidence for transports of methylmercury complexes across both the blood-brain and placental barriers. Although it exhibits a range of toxic effects in several target tissues, the primary effects of methylmercury are on the central nervous system. Neurotoxicity occurs in both adults and developing animals. In development, the effect appears to be linked to microtubule formation disruption (EPA, 1997e). Methylmercury ingestion can also cause reduced food intake, weight loss, muscular atrophy, and damage to an animal's heart, lungs, liver, kidneys, and stomach (EPA, 1997e).

Levels of exposure that induce mercury poisoning in mammals varies among species. Death occurs in sensitive mammal species at 0.1 to 0.5 mg/kg-BW/day or 1.0 to 5.0 mg/kg in the diet. Smaller animals (e.g., mink) are generally more susceptible to mercury poisoning than are larger animals (e.g., mule deer and harp seals), perhaps because of different elimination rates or higher exposure amounts (EPA, 1997e).

Toxicity of mercury to mink became a concern to mink ranchers in the 1970s, which coincided with concerns over PCB toxicity. As with invertebrates, fish, and birds, methylmercury was found to have greater toxicity to mammals than elemental mercury. Mink occupy a top trophic position in the aquatic food web and, therefore, are more exposed to contaminants that biomagnify in the food chain. The diet of mink varies with location, time of year, and available prey. Mink mostly consume fish, but also eat small animals, crayfish, birds, and amphibians. Mercury concentrations in mink have been positively correlated with prey items. Mink accumulate about 10 times more mercury on a concentration basis than did predatory fish from the same drainage area (EPA, 1997e).

Toxicity of different forms of mercury were demonstrated in a study by Aulerich *et al.* (1974) using mink fed with 5 ppm methylmercury or 10 ppm mercuric chloride. Mink treated with methylmercury died within 30 days, while exposure to the mercuric chloride did not produce adverse effects over 5 months.

Wobeser *et al.* (1976a) also examined the effects of organic and inorganic mercury on mink by feeding adult female and juvenile ranch mink rations of 50 and 75 percent fish that contained 0.44 mg/kg mercury over a 145-day period. No clinical or pathological signs of mercury poisoning were observed at these exposure concentrations, suggesting a NOAEC of 0.084 mg-Hg/kg-BW/day based upon a food ingestion of 153 g/day and a body weight of 0.8 kg.

In a related study, Wobeser *et al.* (1976b) conducted a chronic dose-response study with female mink. The mink were exposed to varying dietary doses of methylmercuric chloride ranging from 1.1 to 15 mg/kg. The only clinical sign of toxicity in the lowest dose group was a tendency of two of the animals to move more slowly during the last days of the experiment. However, histopathological abnormalities were observed including pale, yellow livers, lesions in the central nervous system, and axonal degeneration. These researchers concluded that if the test had been extended, the damage to the central nervous system in the 1.1 mg/kg group would have been manifested as impaired motor function. Anorexia, posteria ataxia, and lateral recumbency were observed in the other four dose groups. Death occurred within 26 to 36 days at 4.8 mg/kg and within 19 to 26 days at 8.3 mg/kg (Wobeser *et al.*, 1976b). A dose of 1.1 mg/kg is equivalent to 0.21 mg/kg-BW/day for mink.

In the diet study conducted by Wren *et al.* (1987a), a dietary dose of 1 mg/kg methylmercury resulted in traumatic poisoning of female mink, and concentrations reached as high as 44.1 mg/kg ww in liver tissue, as compared to 0.02 mg/kg ww in the controls. While Wobeser *et al.*'s studies (1976a, 1976b) had suggested that 1 mg/kg in the diet was not acutely toxic, the Wren study concluded that the dietary methylmercury consumption, coupled with cold stress during the exposure period, resulted in a synergistic interaction leading to mortality. Despite the apparent acute toxicity observed, there was no apparent effect on reproduction and kit survival, relative to the control group, at 1 mg/kg in the diet (Wren *et al.*, 1987b).

Based on the results of Wobeser *et al.* (1976a, 1976b), TRVs selected for assessing risk to piscivorous mammals were a LOAEC of 0.21 mg/kg-BW/day and a NOAEC of 0.084 mg/kg-BW/day. These TRVs will be compared to estimated dietary intake rates for mink.

6.4 Characterization of Exposure

This section defines the levels of COPCs found in the various environmental media throughout the Lower Fox River and Green Bay. Exposure point concentrations were developed from the Fox River Database (FRDB). The FRDB, discussed in Section 4, contains over 305,000 separate records for levels of

contaminants for all the receptors and measurement endpoints identified in Table 6-2. This includes measurements in water, sediments, fish at several trophic levels, and piscivorous, insectivorous, and carnivorous birds. While the FRDB contains data from 1975 onward, only data collected since 1989 were used in this BLRA.

For each COPC, media, and reach, the FRDB was queried to provide the following:

- Total number of samples,
- Frequency of detection,
- Maximum detected concentration,
- Minimum detected concentration,
- Arithmetic mean concentration,
- Data distribution (normal, log-normal),
- 95% upper confidence limit (UCL) of the mean concentration, and
- Reasonable maximum exposure (RME) concentration.

The age of whole organisms (e.g., fish and birds) at the time of analysis for COPC concentrations was not recorded within the FRDB. It was assumed that the FRDB whole body concentrations were comparable to the selected TRVs, even if the TRVs were derived from organisms at early life stage. This is an inherent uncertainty that will be further discussed in Section 6.6.

For piscivorous birds and mammals, where exposure data may be lacking, exposure will be modeled to evaluate risk, as discussed in Section 6.2.4. For calculation of exposure values, one-half of the sample quantitation limit was used for undetected values (EPA, 1991b). The 95% UCL of the mean is the value that a mean, calculated repeatedly from subsamples of the data population, will not exceed 95 percent of the time. Therefore, there is a 95 percent probability that the true mean of the population does not exceed the 95% UCL. The 95% UCL was calculated from the sample values depending on whether the data were normally, log-normally, or not normally distributed. When the data distribution fit neither a normal or log-normal distribution pattern, the 95% UCL selected was the greater of the two calculated 95% UCLs (normal and log-normal). In cases where data was limited, or where the variability in the data was high, the calculated 95% UCL can exceed the maximum detected concentration. The RME is defined as the lesser of the calculated 95% UCL, or the maximum detected value. Appendix B presents in further detail how data distributions and statistics were calculated.

As an estimate of risk, both the arithmetic mean concentration and the RME concentration are used as exposure point concentrations. The RME is an estimate of the highest average exposure expected to occur at a site. The intent of the RME is to provide an estimate of exposure that is above average, yet still within the range of most exposures. By design, the estimated RME exposures are higher than will be experienced by most receptors in an exposed population. The RME thus provides a degree of protectiveness that encompasses the individual receptors who have a higher likelihood of exposure.

It was assumed that calculation of risk based on the mean and RME statistics was representative of the risk posed to the majority of individuals in an exposed population because:

- The FRDB contains a large number of records, particularly for total PCBs, which reduces the uncertainty surrounding the estimated COPC concentrations to which receptor populations are exposed;

- The FRDB records included data collected over several years (1989 to the present) suggesting that maximum detected concentrations may not best represent current levels of exposure to the receptor population; and

- Where the number of samples analyzed was limited, the data were variable, or few samples had detected COPC concentrations, the maximum measured concentration was the RME.

The only instance where risk was characterized only on the RME concentration was when the calculated mean concentration exceeded the measured maximum concentration. This, on occasion, occurred when the COPC in the media was infrequently detected because of the way in which detection limits were included in statistical calculations. Additionally, the only instance in which mean and maximum concentrations rather than mean and RME concentrations were used for risk evaluation was for bald eagles, which are a federally-listed species, and therefore, a species for which each individual in the population must be protected.

Risk estimates based on using the 95th percentile concentrations of total PCBs for two receptors (walleye and double-crested cormorants) are evaluated as part of the uncertainty analyses and these results are presented in Section 6.6. These 95th percentile concentrations represent high-end exposure concentrations that are experienced by a minority of individuals in the population.

The remainder of this section presents the specific exposure concentrations measured for each media in each river reach and bay zone. The concentrations

of water and sediment COPCs found in each river reach will be compared to the concentrations found in the river reach immediately upstream. Similarly, for the Green Bay zones, concentrations of water and sediment COPCs found in each zone will be compared to the concentrations found in the zones nearest to the Lower Fox River. For Little Lake Butte des Morts, the first river reach examined for risk, COPC concentrations will be compared to concentrations found upstream in the nearby Lake Winnebago. Since the number of samples collected from Lake Winnebago were few, no statistics will be run on these samples; only the minimum and maximum reported concentrations will be presented.

Water Concentrations. Water concentrations, as available, are presented as filtered, unfiltered, and particulate. In estimating ecological risk, unfiltered water concentrations are preferred because unfiltered water is generally what organisms are ingesting. For COPCs that have unfiltered concentrations measured or detected, and where filtered and particulate water concentrations were measured and detected, total water concentrations of COPCs were estimated based on summing the concentrations measured in the filtered and particulate fractions.

Sediment Concentrations. Exposure Effect Concentrations (EECs) for sediments were based upon measurements taken only in the upper 10 cm. While all COPCs were measured at deeper depths, 10 cm represents a reasonable estimate of the biologically active zone for benthic infauna. Ten cm is consistent with chemical and biological sampling done under the ARCS program (EPA, 1996a).

For total PCBs, EECs were calculated on the actual data for the upper 10 cm in the FRDB, but also from the grid values derived in construction of the interpolated bed maps described in Section 2 (see Figures 2-2 through 2-6). Interpolated EECs were included in the risk assessment to: 1) provide a better estimation of reach-wide PCB concentrations, and 2) to provide an estimation of risk that is consistent with use of the bed maps in both the Remedial Investigation and the Feasibility Study.

Methods for generating the grid values used in bed map construction were briefly described in Section 2. Output from the interpolation model had areas within river or bay reaches where grid values were not generated. This was due to either no data, or the grid cells were located too far from other cells with values to credibly extrapolate the concentration intervals. To account for these data gaps, interpolated EECs were calculated by either deleting those grids (labeled I_d), or a value of 0 (zero) was assigned to the grids (I_0), and then calculating the summary statistics.

Tissue Concentrations. For evaluation of ecological risk, tissues evaluated were generally limited to whole bodies, principally because:

- It is whole body prey that are consumed by receptors, and
- Most research has focused on deriving toxicity thresholds based on whole body concentrations.

Unlike water and sediment concentrations that were collected in all river reaches and Green Bay zones, receptor tissue concentrations were not consistently collected in all reaches and zones. This is due in part to not all receptors residing in all locations and, in general, fewer tissue data have been collected.

Section 6.4.1 presents and compares all exposure concentrations on an area and media basis. Section 6.4.2 summarizes these exposure concentrations based on media, and Section 6.4.3 summarizes these exposure concentrations based on area.

6.4.1 Exposure Concentrations by Area and Media

Little Lake Butte des Morts Reach

Water. COPCs analyzed in surface water include lead, mercury, and total PCBs. These data are presented in Table 6-8.

Both filtered and unfiltered lead were detected in the single sample collected. Mercury was not detected in filtered water (i.e., dissolved) (n = 2), however, it was detected in five out of six unfiltered samples. Mercury was also not detected in an upstream sample collected in Lake Winnebago (Table 6-9).

Total PCBs, were not detected in unfiltered samples (n = 6), but they were frequently detected (>82% of the samples) in filtered and particulate samples (n = 46 and 41, respectively) (Table 6-8). Concentrations of total PCBs in filtered and particulate samples were summed to estimate a total water concentration of total PCBs. Detected minimum surface water concentrations were 5.3 times higher in Lake Winnebago (8 ng/L) (Table 6-9) than in the Little Lake Butte des Morts Reach (1.5 ng/L). Detected maximum concentrations were 4.6 times higher in the Little Lake Butte des Morts Reach (59.2 ng/L) than in Lake Winnebago (13 ng/L).

Sediment. COPCs analyzed in surface sediment include arsenic, lead, mercury, 2,3,7,8-TCDD, 2,3,7,8-TCDF, total PCBs, and chlorinated pesticides (dieldrin,

p,p'-DDD, p,p'-DDE, p,p'-DDT). These data are presented in Table 6-10. The only COPC not detected in surface sediment was p,p'-DDE (n = 20).

Of the metals, arsenic was detected in 89 percent (n = 27) of the samples and the maximum concentration (6.8 mg/kg) was just above the maximum concentration measured in Lake Winnebago surface sediment samples (6.0 mg/kg) (Table 6-11). Lead was detected in all samples (n = 27) and the maximum concentration (522 mg/kg) exceeded, by 13 times, the maximum concentration measured in Lake Winnebago surface sediment samples (39.0 mg/kg). Mercury was detected in 83 percent of samples (n = 86) and the maximum concentration (3.3 mg/kg) exceeded, by 19 times, the maximum concentration measured in Lake Winnebago surface sediment samples (0.2 mg/kg) (Table 6-11).

The dioxin 2,3,7,8-TCDD and the furan 2,3,7,8-TCDF were detected at a frequency of 80 and 100 percent, respectively (n = 5). These compounds were not detected upstream in Lake Winnebago sediments.

Concentrations of non-interpolated total PCBs were detected at a frequency of 97 percent (n = 302). The maximum concentration was 130,000 μg/kg which greatly exceeded the maximum concentrations measured in Lake Winnebago surface sediment (36.0 μg/kg). Interpolated PCB sediment concentrations were calculated with the assumption that all grid areas for which no values existed were equal to zero (I_0) or were deleted from the database (I_d). The maximum interpolated PCB values were 60,000 μg/kg for both I_0 and I_d.

Dieldrin, p,p'-DDD, and p,p'-DDT were detected at frequencies of 7, 17, and 10 percent, respectively in surface sediment samples of Little Lake Butte des Morts (n = 15 to 23). The derivative p,p'-DDE was not detected in any samples (n = 20). Contrary to these results, only DDE was detected in Lake Winnebago sediment (n = 3).

Sediment concentrations of PCB coplanar congeners of concern (congener numbers 77, 81, 105, 118, 126, and 169) are given in Table 6-12. Only congener number 169 was not detected (n = 20). Frequency of detection of the other congeners ranged from 44 to 100 percent (n = 18 to 46). RME concentrations of PCB congeners ranged from 0.3 μg/kg (congener 126) to 596 μg/kg (congener 118). These PCB congeners were not detected in Lake Winnebago sediments (n = 3) Table 6-13.

Fish. COPCs analyzed in whole fish identified as important receptors include arsenic, mercury, total PCBs, and chlorinated pesticides (dieldrin, o,p'-DDD, o,p'-DDE, o,p'-DDT, p,p'-DDD, p,p'-DDE, p,p'-DDT). These data are presented in Table

6-14. Fish analyzed in this reach include gizzard shad, golden shiner, yellow perch, carp, and walleye. In all fish analyzed, o,p'-DDD (carp and walleye), o,p'-DDT (carp and walleye), and p,p'-DDT (carp, walleye, and yellow perch) were not detected.

Arsenic was detected in both of the carp samples tested at a mean concentration of 0.14 mg/kg. Mercury was detected in 60 percent of carp samples at a mean concentration of 0.05 mg/kg. Mercury was detected in one walleye sample at a concentration of 0.03 mg/kg (n = 4). Mercury was not detected in yellow perch (n = 2).

Total PCB data were available for carp, gizzard shad, golden shiner, walleye, and yellow perch. Although sample sizes varied, frequencies of detection of total PCBs ranged from 85 to 100 percent. The mean total PCB concentration in carp (1,992 μg/kg), the highest of all the fish tested, was 72 percent higher than the mean concentration in walleye (1,159 μg/kg), which had the next highest mean. The mean concentration in yellow perch (363 μg/kg) was approximately one-third of the mean concentration in walleye. The mean concentrations of total PCBs in golden shiner (993 μg/kg) and gizzard shad (296 μg/kg), both lower trophic level species, had lower concentrations than carp or walleye.

Only carp, walleye, and yellow perch were analyzed for concentrations of chlorinated pesticides (Table 6-14). While carp and walleye were analyzed for all of the pesticides listed above, yellow perch were only analyzed for dieldrin, p,p'-DDD, p,p'-DDE, and p,p'-DDT. Dieldrin was only detected in carp (n = 6). The frequency of detection was 33 percent and the maximum concentration was 1.0 μg/kg. The only pesticide detected in yellow perch (n = 2) was p,p'-DDE at a mean concentration of 9.5 μg/kg. Incidences of pesticide detection were most frequent for p,p'-DDE (detection frequency of 71% or more). Mean concentrations of p,p'-DDE in carp and walleye were 16.9 and 47.6 μg/kg, respectively. The detection of the other DDE isomer (o,p'-DDE) was limited to one carp (5.8 μg/kg) and one walleye (16.0 μg/kg), a detection frequency of 25 percent. DDD (p,p'- isomer) was detected only in carp and walleye where frequencies of detection were 43 and 14 percent, respectively. Maximum concentrations of p,p'-DDD were 5.2 μg/kg for carp and 78.0 μg/kg for walleye.

Concentrations of dioxin, furan, and PCB congeners in whole fish are presented in Table 6-15. Although sample sizes were small, ranging from two to three whole fish for dioxin/furans and one to seven whole fish for PCB congeners, data distributions and statistics were calculated for all detected analytes and detection frequencies were generally greater than 60 percent. The species of fish analyzed for these compounds included carp (dioxins, furans, and PCB congeners), walleye

(dioxins, furans, and PCB congeners), yellow perch (PCB congeners), and golden shiner (PCB congeners). The mean concentration of 2,3,7,8-TCDD was greater in walleye (0.00045 μg/kg) than in carp (0.00025 μg/kg), as was the mean concentration of 2,3,7,8-TCDF in walleye (0.0054 μg/kg) as compared to carp (0.0022 μg/kg). For all species examined, concentrations of PCB congener 118 exceeded concentrations of the other congeners. These differences were greatest for the upper trophic level species, walleye and carp. PCB congener 169 was not detected in any fish except walleye, and in walleye it was only detected in one of seven fish. Overall, PCB congener concentrations greatly exceeded dioxin/furan concentrations where, in all fish examined, mean concentrations of detected PCB congeners ranged from 0.03 μg/kg (congener 126 in golden shiner) to 36.8 μg/kg (congener 118 in walleye).

Birds. COPCs analyzed in birds identified as important receptors include total PCBs and chlorinated pesticides (dieldrin, o,p'-DDD, o,p'-DDE, o,p'-DDT, p,p'-DDD, p,p'-DDE, p,p'-DDT). These data are presented in Table 6-16. The only species analyzed in this reach was tree swallow. The only detected chlorinated pesticide was p,p'-DDE.

Total PCB concentrations were measured in tree swallow eggs (n = 5) and whole bodies (n = 24), with mean concentrations of 2,924 and 2,135 μg/kg, respectively (Table 6-16). Chlorinated pesticides were analyzed in whole tree swallows (n = 18) and only p,p'-DDE was detected with a mean concentration of 155 μg/kg.

For all piscivorous receptors (common terns, Forster's terns, double-crested cormorants, and bald eagles), exposure point concentrations were estimated using the equation presented in Section 6.2.4 and the input parameter values presented in Table 6-4. The fish species selected for modeling were yellow perch (mercury, dieldrin, and p,p'-DDE) and gizzard shad (total PCBs) as a trophic level 3 fish, and walleye as a trophic level 4 fish. Resulting estimated exposure point concentrations are presented in Table 6-17.

Fish concentrations of mercury and dieldrin were not detected; therefore, one-half the detection limit was used to estimate mercury and dieldrin exposure concentrations. The mean estimated exposure concentrations for mercury were 12.5 μg/kg-BW/day (common tern), 11.5 μg/kg-BW/day (Forster's tern), 4.8 μg/kg-BW/day (double-crested cormorant), and 5.7 μg/kg-BW/day (bald eagle). The mean estimated exposure concentrations for total PCBs were 145 μg/kg-BW/day (common tern), 134 μg/kg-BW/day (Forster's tern), 56.0 μg/kg-BW/day (double-crested cormorant), and 207 μg/kg-BW/day (bald eagle). The mean estimated exposure concentrations for dieldrin were 0.6 μg/kg-BW/day (common

tern), 0.6 μg/kg-BW/day (Forster's tern), 0.2 μg/kg-BW/day (double-crested cormorant), and 0.4 μg/kg-BW/day (bald eagle). The mean estimated exposure concentrations for p,p'-DDE were 4.7 μg/kg (common tern), 4.3 μg/kg (Forster's tern), 1.8 μg/kg (double-crested cormorant), and 2.6 μg/kg-BW/day (bald eagle).

PCB coplanar congener concentrations are available for tree swallow eggs and whole bodies. Results of these analyses are presented in Table 6-18. In eggs (n = 5), all congeners of interest (77, 105, 118, 126, and 169) were detected, while in whole body birds (n = 15), only congeners 105, 118, and 126 were detected. For both eggs and whole bodies, PCB congener 118 coeluted with congener 106. For all detected congeners, detection frequencies were 100 percent except PCB 169 and 126. PCB 169 in eggs was detected at a 20 percent frequency and PCB 126 in whole bodies was detected at a 40 percent frequency. Like fish, concentrations of PCB 118/106 exceeded concentrations of the other congeners in both eggs and whole birds.

Mammals. Exposure point concentrations for mink were estimated using the equation presented in Section 6.2.4 and the input parameter values presented in Table 6-4. Carp was the selected fish species for modeling. Non-interpolated and interpolated sediment total PCB concentrations were each used in the exposure modeling. Resulting estimated exposure point concentrations are presented in Table 6-19. The mean estimated exposure concentration for mercury was 14.8 μg/kg-BW/day. The mean estimated exposure concentration for total PCBs (N), total PCBs (I_0), and total PCBs (I_d) were 435, 397, and 400 μg/kg-BW/day, respectively. The mean estimated exposure concentration for dieldrin was 0.6 μg/kg-BW/day. For p,p'-DDE, the mean estimated exposure concentration was 3.2 μg/kg-BW/day.

Appleton to Little Rapids Reach

Water. COPCs analyzed in surface water include arsenic, lead, mercury, 2,3,7,8-TCDF, total PCBs, and the chlorinated pesticides dieldrin, DDD, DDE, and DDT. These data are presented in Table 6-20. Arsenic, 2,3,7,8-TCDF, and all chlorinated pesticides were not detected.

Unfiltered lead, having a mean concentration of 1,397 ng/L, was detected with 100 percent frequency (n = 3). Mercury in filtered samples was detected with 50 percent frequency (n = 2) and in unfiltered samples was detected with 40 percent frequency (n = 5); mean concentrations were 65 and 66 ng/L, respectively. Therefore, approximately all of the mercury was in the dissolved form. As compared to concentrations in Little Lake Butte des Morts (Table 6-8), unfiltered lead concentrations were similar, while unfiltered mercury concentrations were

much lower in the Appleton to Little Rapids Reach—a mean concentration of 66 ng/L as compared to 2,237 ng/L.

Total PCBs were not detected in unfiltered samples (n = 1), but they were frequently detected (>95% of the samples) in filtered and particulate samples (n = 85 and 86, respectively) (Table 6-20). Concentrations of total PCBs in filtered and particulate samples were summed to estimate a total water concentration of total PCBs. The estimated mean concentration of 16.8 ng/L is approximately 40 percent less than the estimated mean concentration of 27.6 ng/L in Little Lake Butte des Morts Reach (Table 6-8).

Sediment. COPCs analyzed in surface sediment include arsenic, lead, mercury, total PCBs, and chlorinated pesticides (dieldrin, p,p'-DDD, p,p'-DDE, p,p'-DDT). These data are presented in Table 6-21. The only COPCs not detected in surface sediment were dieldrin and p,p'-DDE (n = 10 for both).

Ten samples were analyzed for metals (arsenic, lead, and mercury) with detection frequencies of 60 percent for arsenic and 100 percent for lead and mercury. The mean arsenic concentration (4.4 mg/kg) is approximately the same as was measured in the Little Lake Butte des Morts Reach (4.6 mg/kg) (Table 6-10). The mean lead concentration (75.6 mg/kg) is approximately 56 percent less than that measured in the surface sediments of Little Lake Butte des Morts Reach (172 mg/kg). The mean mercury concentration (0.8 mg/kg) is approximately 20 percent less than what was measured in the surface sediments of Little Lake Butte des Morts Reach (1.0 mg/kg).

Concentrations of non-interpolated PCBs were detected at a frequency of 93 percent (n = 131). The mean concentration was 6,751 μg/kg. The mean concentration was approximately half the mean concentration detected in surface sediments in the Little Lake Butte des Morts Reach (10,724 μg/kg). The mean concentrations of interpolated PCBs were 175 μg/kg (I_0) and 1,398 μg/kg (I_d). Interpolated PCB concentrations were approximately 95 percent less (I_0)and 62 percent less (I_d) than those measured in the Little Lake Butte des Morts Reach (Table 6-10).

Chlorinated pesticides (dieldrin, p,p'-DDD, p,p'-DDE, and p,p'-DDT) were analyzed in 10 surface sediment samples, but only p,p'-DDD and p,p'-DDT were detected with detection rates of 20 and 10 percent, respectively. Results are similar to what was found in the Little Lake Butte des Morts Reach except dieldrin was detected in one sample at that location. RME concentrations of p,p'-DDD and p,p'-DDT were 91 percent lower and 93 percent lower,

respectively, than RME concentrations measured in the Little Lake Butte des Morts Reach.

Sediment concentrations of PCB coplanar congeners of concern (congener numbers 77, 81, 105, 118, 126, and 169) are presented in Table 6-22. Only congener number 169 was not detected (n = 13). Frequency of detection of the other congeners ranged from 22 to 100 percent (n = 9 to 21). RME concentrations of PCB congeners ranged from 0.1 µg/kg (congener 126) to 181 µg/kg (congener 118), as compared to RME congener concentrations in Little Lake Butte des Morts Reach which ranged from 0.3 µg/kg (congener 126) to 596 µg/kg (congener 118) (Table 6-12).

Fish. COPCs analyzed in whole fish identified as important receptors include mercury, total PCBs, and chlorinated pesticides (dieldrin, o,p'-DDD, o,p'-DDT, p,p'-DDD, p,p'-DDE, p,p'-DDT). These data are presented in Table 6-23. Fish analyzed in this reach include yellow perch, carp, and walleye. Dieldrin and p,p'-DDT were not detected in any fish (yellow perch, carp, and walleye). The o,p- isomers of DDD and DDT were only analyzed in carp, but were not detected.

Mercury was only detected in 20 percent of the carp (n = 5) and 67 percent of the walleye (n = 3). Mean mercury concentrations in these fish (0.06 mg/kg for carp and 0.14 mg/kg for walleye) were approximately 2 to 3 times greater than concentrations in the Little Lake Butte des Morts Reach (Table 6-14). As in Little Lake Butte des Morts Reach, mercury was not detected in yellow perch (n = 4).

Total PCBs were detected in all three species at mean concentrations of 2,581 µg/kg in carp, 2,737 µg/kg in walleye, and 779 µg/kg in yellow perch. As compared to mean concentrations of total PCBs in these three fish species collected in the Little Lake Butte des Morts Reach, carp concentrations were about 30 percent higher, walleye concentrations were 2.4 times higher, and yellow perch concentrations were 2.1 times higher in the Appleton to Little Rapids Reach.

As indicated above, chlorinated pesticides analyzed in yellow perch, carp, and walleye, but were infrequently detected. Only one walleye (n = 3) had a detected concentration of p,p'-DDD (8.0 µg/kg). In contrast, p,p'-DDE was detected in all three species at a frequency of at least 67 percent. The mean p,p'-DDE concentrations for carp and walleye, were 47.8 and 57.0 µg/kg, respectively, and the concentration in the only yellow perch analyzed was 10.0 µg/kg. As compared to mean concentrations of p,p'-DDE in these three fish species collected in the Little Lake Butte des Morts Reach, carp concentrations were about 2.8 times higher, walleye concentrations were about 20 percent higher, and yellow perch

concentrations were about 5 percent higher in the Appleton to Little Rapids Reach.

Dioxins and furans were not analyzed in the Appleton to Little Rapids Reach. Concentrations of PCB congeners in whole fish are presented in Table 6-24. Congener detection frequencies ranged from 0 to 100 percent and sample sizes were three to five whole fish. The species of fish analyzed for these compounds included carp, walleye, and yellow perch. Congener 81 was only detected in yellow perch (25 percent detection frequency) and congener 169 was only detected in carp (40 percent detection frequency). As was the case for fish in Little Lake Butte des Morts Reach, concentrations of PCB congener 118 exceeded concentrations of the other congeners. Compared to congener concentrations in fish from Little Lake Butte des Morts Reach, congener concentrations in fish from Appleton to Little Rapids Reach were generally greater. In Little Lake Butte des Morts Reach, mean detected PCB congener concentrations ranged from 0.03 μg/kg (congener 126 in golden shiner) to 36.8 μg/kg (congener 118 in walleye) and in Appleton to Little Rapids Reach, mean detected PCB congener concentrations ranged from 0.02 μg/kg (congener 126 in yellow perch) to 80.3 μg/kg (congener 118 in walleye).

Birds. COPCs analyzed in birds identified as important receptors include mercury, total PCBs, and chlorinated pesticides (dieldrin, p,p'-DDD, p,p'-DDE, p,p'-DDT). These data are presented in Table 6-25. The only species analyzed in this reach was bald eagle. Only p,p'-DDT was not detected.

Data for this reach are limited to analysis of one sample of bald eagle liver (mercury) and one egg sample (total PCBs and pesticides). Mercury was detected in the liver sample at a concentration of 1.4 mg/kg. Total PCBs were detected in the egg sample at a concentration of 36,000 μg/kg. Detected pesticides in the egg sample included dieldrin (70.0 μg/kg), p,p'-DDD (160 μg/kg), and p,p'-DDE (1,100 μg/kg).

Because of the lack of data, exposure point concentrations for all piscivorous bird receptors (common terns, Forster's terns, double-crested cormorants, and bald eagles) were estimated using the equation presented in Section 6.2.4 and the input parameter values presented in Table 6-4. The fish species selected for modeling were yellow perch as a trophic level 3 fish, and walleye as a trophic level 4 fish. Resulting estimated exposure point concentrations are presented in Table 6-26.

Fish concentrations of mercury and dieldrin were not detected, therefore, one-half the detection limit was used to estimate mercury and dieldrin exposure

concentrations. The mean estimated exposure concentrations for mercury were 12.3 µg/kg-BW/day (common tern), 11.3 µg/kg-BW/day (Forster's tern), 4.7 µg/kg-BW/day (double-crested cormorant), and 8.6 µg/kg-BW/day (bald eagle). The mean estimated exposure concentrations for total PCBs were 382 µg/kg-BW/day (common tern), 352 µg/kg-BW/day (Forster's tern), 148 µg/kg-BW/day (double-crested cormorant), and 296 µg/kg-BW/day (bald eagle). The mean estimated exposure concentrations for dieldrin were 0.6 µg/kg-BW/day (common tern), 0.6 µg/kg-BW/day (Forster's tern), 0.2 µg/kg-BW/day (double-crested cormorant), and 0.2 µg/kg-BW/day (bald eagle). The mean estimated exposure concentrations for p,p'-DDE were 4.9 µg/kg-BW/day (common tern), 4.5 µg/kg-BW/day (Forster's tern), 1.9 µg/kg-BW/day (double-crested cormorant), and 5.6 µg/kg-BW/day (bald eagle).

No PCB congener data in birds is available for this reach.

Mammals. Exposure point concentrations for mink were estimated using the equation presented in Section 6.2.4 and the input parameter values presented in Table 6-4. Carp was the selected fish species for modeling. Non-interpolated and interpolated sediment total PCB concentrations were each used in the exposure modeling. Resulting estimated exposure point concentrations are presented in Table 6-27. The mean estimated exposure concentration for mercury was 15.5 µg/kg-BW/day. The mean estimated exposure concentration for total PCBs (N), total PCBs (I_0), and total PCBs (I_d) were 527, 494, and 501 µg/kg-BW/day, respectively. The mean estimated exposure concentration for dieldrin was 0.4 µg/kg-BW/day. For p,p'-DDE, the mean estimated exposure concentration was 9.1 µg/kg-BW/day.

Little Rapids to De Pere Reach

Water. COPCs analyzed in surface water included lead, mercury, and total PCBs. These data are presented in Table 6-28. Each of these compounds were detected.

Filtered and unfiltered lead were detected at a frequency of 100 percent (n = 2) and the mean unfiltered concentration of lead (617 ng/L) exceeded the mean filtered concentration (121 ng/L) by a factor of five. This unfiltered concentration of lead, however, was 56 percent less than what was detected in surface water in the Appleton to Little Rapids Reach (Table 6-20). Filtered and unfiltered mercury were detected at a frequency of 67 percent (n = 3) and the mean unfiltered concentration of mercury (3,883 ng/L) exceeded the mean filtered concentration (1,273 ng/L) by a factor of three. Additionally, these mean concentrations of filtered (65.0 ng/L) and unfiltered (66.4 ng/L) mercury concentrations exceeded mean concentrations detected in surface water in the

Appleton to Little Rapids Reach by approximately 20 times and 59 times, respectively.

Total PCBs were not analyzed in unfiltered samples. Total PCBs were analyzed and frequently detected (>95% of the samples) in filtered and particulate samples (n = 98 and 98, respectively) (Table 6-28). Concentrations of total PCBs in filtered and particulate samples were summed to estimate a total water concentration of total PCBs. The estimated mean concentration of 41.1 ng/L is more than twice the estimated mean concentration of 16.8 ng/L in Appleton to Little Rapids Reach (Table 6-20).

Sediment. COPCs analyzed in surface sediment include arsenic, lead, mercury, 2,3,7,8-TCDD, 2,3,7,8-TCDF, total PCBs, and chlorinated pesticides (dieldrin, p,p'-DDD, p,p'-DDE, p,p'-DDT). These data are presented in Table 6-29. The only COPC not detected in surface sediment was dieldrin (n = 19).

Of the metals, arsenic was detected in 90 percent (n = 20) of the samples and the mean concentration (4.6 mg/kg) is about the same concentration as was measured in surface sediment in Appleton to Little Rapids (4.4 mg/kg) (Table 6-21). Lead was detected in all samples (n = 20) and the mean concentration (159 mg/kg) is about twice what was measured in Appleton to Little Rapids (76 mg/kg). Mercury was detected in all samples (n = 74) and the mean concentration (3.5 mg/kg) exceeded by over four times the mean concentration measured in Appleton to Little Rapids (0.8 mg/kg).

The dioxin 2,3,7,8-TCDD and the furan 2,3,7,8-TCDF were both detected at a frequency of 100 percent (n = 2). Dioxin and furan samples were not collected upstream in the Appleton to Little Rapids Reach, but they were detected in Little Lake Butte des Morts Reach. The mean concentrations of 2,3,7,8-TCDD (0.0053 μg/kg) were more than two times the mean concentrations measured in sediment in Little Lake Butte des Morts (0.0025 μg/kg) (Table 6-10). Mean concentrations of 2,3,7,8-TCDF were comparable to mean concentrations measured in sediment in Little Lake Butte des Morts (0.081 and 0.064 μg/kg, respectively).

Concentrations of non-interpolated PCBs were detected at a frequency of approximately 97 percent (n = 209). The mean concentration of total non-interpolated PCBs (4,782 μg/kg) was about 30 percent less than the mean sediment concentration measured in the Appleton to Little Rapids Reach (6,751 μg/kg). The mean concentrations of interpolated PCBs were similar—2,054 μg/kg (I_0) and 2,078 μg/kg (I_d). Mean I_0 PCB concentrations were approximately 12 times those measured in the Appleton to Little Rapids Reach (Table 6-21). Mean

I_d PCB concentrations were approximately 49 percent more than those measured in the Appleton to Little Rapids Reach.

Chlorinated pesticides (dieldrin, p,p'-DDD, p,p'-DDE, and p,p'-DDT) were analyzed in surface sediment samples, and only dieldrin was not detected. Detection frequencies of the other compounds were low, ranging from 21 to 25 percent (n = 14 to 20). As compared to surface sediment concentrations in the Appleton to Little Rapids Reach, the RME p,p'-DDD concentration (2.8 μg/kg) was 65 percent higher, while the RME p,p'-DDT concentration (20.0 μg/kg) was almost six times that found in the Appleton to Little Rapids Reach (Table 6-21). The RME p,p'-DDE concentration was 22.0 μg/kg. The only other upstream location where DDE was detected was Lake Winnebago, where the maximum concentration was 3.5 μg/kg.

Sediment concentrations of PCB coplanar congeners of concern (congener numbers 77, 81, 105, 118, 126, and 169) are given in Table 6-30. Only congener number 169 was not detected (n = 23). Frequency of detection of the other congeners ranged from 22 to 98 percent (n = 22 to 40). RME concentrations of PCB congeners ranged from 0.8 μg/kg (congener 126) to 58.4 μg/kg (congener 118) as compared to RME congener concentrations in Appleton to Little Rapids Reach which ranged from 0.1 μg/kg (congener 126) to 181 μg/kg (congener 118) Table 6-22. These results are consistent with minimum and maximum congeners detected for all other reaches examined thus far, where congener 126 had the minimum RME concentration and congener 118 had the greatest concentration.

Fish. COPCs analyzed in whole fish identified as important receptors include mercury, total PCBs, and chlorinated pesticides (dieldrin, o,p'-DDD, o,p'-DDE, o,p'-DDT, p,p'-DDD, p,p'-DDE, p,p'-DDT). These data are presented in Table 6-31. Fish analyzed in this reach include gizzard shad, golden shiner, yellow perch, carp, and walleye. In all fish analyzed, o,p'-DDD (carp and walleye), o,p'-DDT (carp and walleye), and p,p'-DDT (carp, walleye, and yellow perch) were not detected.

Mercury was detected at a concentrations of 0.15 and 0.16 mg/kg, respectively, in the single carp and walleye analyzed. This concentration in carp is close to the mercury concentration (0.12 mg/kg) detected in the one carp from the Appleton to Little Rapids Reach (Table 6-23). The concentration in walleye is also close to the mean mercury concentration (0.14 mg/kg) measured in carp.

Total PCB data were available for carp, walleye, yellow perch, gizzard shad, and golden shiner (Table 6-31). Although samples sizes varied (n = 1 to 20), all fish had a 100 percent detection frequency. Mean total PCB concentrations in carp (3,919 μg/kg) were the highest of all the fish tested. As compared to mean

concentrations in other fish tested, walleye (3,179 μg/kg) were 19 percent lower, golden shiner (1,020 μg/kg) were 74 percent lower, gizzard shad (347 μg/kg) were more than 10 times lower, and the concentration in the only yellow perch analyzed (627 μg/kg) was 84 percent lower. As compared to mean concentrations of total PCBs collected in the Appleton to Little Rapids Reach (Table 6-23), carp concentrations were 52 percent higher, walleye concentrations were 16 percent higher, and yellow perch concentrations were about 20 percent lower in the Little Rapids to De Pere Reach.

Carp, walleye, and yellow perch were analyzed for chlorinated pesticides (Table 6-31). Yellow perch were only analyzed for dieldrin, p,p'-DDD, p,p'-DDE, and p,p'-DDT. The number of samples analyzed for each species ranged from one (yellow perch) to five (carp). Dieldrin was only detected in a single walleye and the mean concentration was 3.4 μg/kg (n = 4). Dieldrin was not detected in any species in the Appleton to Little Rapids Reach (Table 6-23). Concentrations of p,p'-DDE were measured at a frequency of 100 percent in carp, walleye, and yellow perch where mean concentrations were 74.2, 129, and 16.0 μg/kg, respectively. As compared to mean concentrations in these three fish species collected in the Appleton to Little Rapids Reach, concentrations of p,p'-DDE in carp, walleye, and yellow perch were 55 percent higher, 2.3 times higher, and 60 percent higher in the De Pere to Little Rapids Reach. The only other pesticide compounds detected were o,p'-DDE in walleye (100 percent detection frequency) at a mean concentration of 45.7 μg/kg and p,p'-DDD in carp (60 percent detection frequency) at a mean concentration of 8 μg/kg.

Concentrations of dioxin, furan, and PCB congeners in whole fish are presented in Table 6-32. Dioxin and furan congener detection frequencies ranged from 0 to 100 percent with sample sizes ranging from one to three whole fish. PCB congener detection frequencies ranged from 0 to 100 percent and sample sizes ranged from one to four whole fish. The species of fish analyzed for these compounds included carp (dioxin, furan, and PCB congeners), walleye (dioxin, furan, and PCB congeners), yellow perch (PCB congeners), and golden shiner (PCB congeners). The mean concentration of 2,3,7,8-TCDD were greater in walleye (0.0008 μg/kg) than in carp (0.00055 μg/kg). The mean concentration of 2,3,7,8-TCDF were also greater in walleye (0.01 μg/kg) than in carp (0.0011 μg/kg). PCB congener 81 alone was not detected in carp, walleye, and yellow perch, however, it was detected in golden shiner coeluting with congeners 87 and 115. PCB congener 169 was only detected in one whole walleye (25 percent detection frequency). As was the case for fish in Little Lake Butte des Morts Reach and Appleton to Little Rapids Reach, concentrations of PCB congener 118 exceeded concentrations of the other congeners. Compared to PCB congener concentrations in fish from Appleton to Little Rapids Reach (Table 6-24),

congener concentrations in fish from Little Rapids to De Pere Reach were generally similar. In Appleton to Little Rapids Reach, mean PCB congener concentrations ranged from 0.02 μg/kg (congener 126 in yellow perch) to 80.3 μg/kg (congener 118 in walleye) and in the Little Rapids to De Pere Reach, mean PCB congener concentrations ranged from 0.01 μg/kg (congener 77 in yellow perch) to 77.0 μg/kg (congener 118 in walleye).

Birds. There are no COPC data for bird receptors in this reach. Because of the lack of data, exposure point concentrations for all piscivorous bird receptors (common terns, Forster's terns, double-crested cormorants, and bald eagles) were estimated using the equation presented in Section 6.2.4 and the input parameter values presented in Table 6-4. The fish species selected for modeling were yellow perch (mercury, dieldrin, and p,p'-DDE) and gizzard shad (total PCBs) as a trophic level 3 fish, and walleye as a trophic level 4 fish. Resulting estimated exposure point concentrations are presented in Table 6-33.

Fish concentrations of mercury and dieldrin were not detected; therefore, one-half the detection limit was used to estimate mercury and dieldrin exposure concentrations. The mean estimated exposure concentrations for mercury were 12.7 μg/kg-BW/day (common tern), 11.7 μg/kg-BW/day (Forster's tern), 4.9 μg/kg-BW/day (double-crested cormorant), and 17.4 μg/kg-BW/day (bald eagle). The mean estimated exposure concentrations for total PCBs were 170 μg/kg-BW/day (common tern), 157 μg/kg-BW/day (Forster's tern), 65.6 μg/kg-BW/day (double-crested cormorant), and 427 μg/kg-BW/day (bald eagle). The mean estimated exposure concentrations for dieldrin were 0.6 μg/kg-BW/day (common tern), 0.6 μg/kg-BW/day (Forster's tern), 0.2 μg/kg-BW/day (double-crested cormorant), and 0.2 μg/kg-BW/day (bald eagle). The mean estimated exposure concentrations for p,p'-DDE were 7.8 μg/kg-BW/day (common tern), 7.2 μg/kg-BW/day (Forster's tern), 3.0 μg/kg-BW/day (double-crested cormorant), and 9.6 μg/kg-BW/day (bald eagle).

Mammals. Exposure point concentrations for mink were estimated using the equation presented in Section 6.2.4 and the input parameter values presented in Table 6-4. Carp was the selected fish species for modeling. Non-interpolated and interpolated sediment total PCB concentrations were each used in the exposure modeling. Resulting estimated exposure point concentrations are presented in Table 6-34. The mean estimated exposure concentration for mercury was 46.6 μg/kg-BW/day. The mean estimated exposure concentration for total PCBs (N), total PCBs (I_0), and total PCBs (I_d) were 773, 760, and 760 μg/kg-BW/day, respectively. The mean estimated exposure concentration for dieldrin was 0.4 μg/kg-BW/day. The mean estimated exposure concentration for p,p'-DDE was 14.3 μg/kg-BW/day.

De Pere to Green Bay Reach (Green Bay Zone 1)

Water. COPCs analyzed in surface water include arsenic, lead, mercury, 2,3,7,8-TCDD, 2,3,7,8-TCDF, total PCBs, dieldrin, p,p'-DDD, p,p'-DDE, and p,p'-DDT. These data are presented in Table 6-35. Compounds not detected include 2,3,7,8-TCDD, 2,3,7,8-TCDF, unfiltered dieldrin, p,p'-DDD, p,p'-DDE, and p,p'-DDT.

Unfiltered arsenic was detected at a frequency of 25 percent (n = 4) and this single concentration (1,500 ng/L) is the only detection of arsenic found so far in surface water. Unfiltered lead was detected at a frequency of 75 percent (n = 4) and the mean concentration (3,113 ng/L) is 5 times the mean concentration (617 ng/L) detected in surface water in the Little Rapids to De Pere Reach (Table 6-28).

Filtered, unfiltered, and particulate mercury were detected at frequencies of greater than 90 percent (n = 45, 45, and 32, respectively). The mean unfiltered concentration of mercury (27.5 ng/L) exceeds the mean filtered concentration (4.9 ng/L) by a factor of more than five. The particulate mercury concentration (23.0 ng/L) is just below the unfiltered concentration, indicating that most of the mercury is in the particulate fraction. These mercury concentrations are less than 1 percent of what was measured in the Little Rapids to De Pere Reach, where the mean concentration of filtered mercury is 1,273 ng/L and the mean concentration of unfiltered mercury is 3,383 ng/L.

Total PCBs were not analyzed in unfiltered samples. Total PCBs were analyzed and frequently detected (>90% of the samples) in filtered and particulate samples (n = 143 and 143, respectively) (Table 6-35). Concentrations of total PCBs in filtered and particulate samples were summed to estimate a total water concentration of total PCBs. The estimated mean concentration of 60.9 ng/L is about 48 percent more than the estimated mean concentration of 41.1 ng/L in the Little Rapids to De Pere Reach (Table 6-28).

Filtered and particulate concentrations of chlorinated pesticides were analyzed for p,p'-DDD and p,p'-DDE, but not for dieldrin, and filtered concentrations were not measured for p,p'-DDT. The only other location where pesticides were measured in surface water was in the Appleton to Little Rapids Reach (Table 6-20). Here, only unfiltered samples were analyzed (n = 3) and, as with the analysis conducted in the De Pere to Green Bay Reach, no pesticides were detected in unfiltered samples. For p,p'-DDD and p,p'-DDE, mean filtered concentrations were respectively 55 and 76 percent lower than mean particulate concentrations. The mean particulate concentration of p,p'-DDT (0.07 ng/L) was the lowest, followed by p,p'-DDD (0.11 ng/L), and the p,p'-DDE mean particulate concentration (0.17 ng/L) was the highest.

Sediment. COPCs analyzed in surface sediment include arsenic, lead, mercury, total PCBs, and chlorinated pesticides (dieldrin, p,p'-DDD, p,p'-DDE, p,p'-DDT). These data are presented in Table 6-36. The only COPCs not detected in surface sediment were dieldrin and p,p'-DDT (n = 22 for both).

Of the metals (n = 92), arsenic was detected at a frequency of 72 percent and the mean concentration (10.1 mg/kg) is about twice the concentration measured in surface sediment in Little Rapids to De Pere Reach (4.6 mg/kg) (Table 6-29). Lead was detected at a frequency of 100 percent and the mean concentration (75.7 mg/kg) is about half what was measured in Little Rapids to De Pere Reach (159 mg/kg). Mercury was detected at a frequency of 97 percent and the mean concentration (1.0 mg/kg) was approximately 71 percent lower than the mean concentration measured in Little Rapids to De Pere Reach (3.5 mg/kg).

Non-interpolated PCBs were detected at a frequency of 98 percent (n = 290). The mean concentration (4,184 μg/kg) was close to the mean sediment concentration measured in the Little Rapids to De Pere Reach (4,782 μg/kg). The mean concentrations of interpolated PCBs were 2,950 μg/kg (I_0) and 2,959 μg/kg (I_d). Mean I_0 and I_d PCB concentrations were both approximately 43 percent more than those measured in Green Bay Zone 2.

Chlorinated pesticides (dieldrin, p,p'-DDD, p,p'-DDE, and p,p'-DDT) were analyzed in surface sediment samples, and dieldrin and p,p'-DDT were not detected (Table 6-36). The detection frequency of p,p'-DDD and p,p'-DDE were 14 and 5 percent, respectively (n = 22). As compared to RME surface sediment concentrations in the Little Rapids to De Pere Reach, the RME p,p'-DDD concentration (4.5 μg/kg) was 61 percent higher, while the single p,p'-DDE concentration (1.9 μg/kg) was 91 percent lower than that found in the Little Rapids to De Pere Reach (Table 6-29).

Sediment concentrations of PCB coplanar congeners of concern (congener numbers 77, 81, 105, 118, 126, and 169) are given in Table 6-37. Only congener number 169 was not detected (n = 26). Frequency of detection of the other congeners ranged from 19 to 100 percent (n = 21 to 26). RME concentrations of PCB congeners ranged from 0.2 μg/kg (congener 81) to 27.0 μg/kg (congener 77) as compared to RME congener concentrations in Little Rapids to De Pere Reach, which ranged from 0.8 μg/kg (congener 126) to 58.4 μg/kg (congener 118) (Table 6-30). These results are similar to congener data for all other reaches examined thus far, where congener 126 had the minimum RME concentration and congener 118 had the highest concentration.

Fish. Fish data from this reach was combined with fish data from Green Bay Zone 2 because it was determined that fish in these areas could not be distinguished as independent populations. These data are presented in Section 6.4.5.

Birds. COPCs analyzed in birds identified as important receptors include total PCBs and chlorinated pesticides (dieldrin, o,p'-DDD, o,p'-DDE, o,p'-DDT, p,p'-DDD, p,p'-DDE, p,p'-DDT). These data are presented in Table 6-38. Tree swallow was the only species analyzed in this reach. The only detected chlorinated pesticides were p,p'-DDD and p,p'-DDE.

A total of 22 whole body samples were analyzed for total PCBs and pesticides. Total PCBs were measured in all whole tree swallows at a mean concentration of 3,118 μg/kg. This mean concentration represents an increase of 46 percent compared to mean concentration observed in whole tree swallows from Little Lake Butte des Morts Reach (2,135 μg/kg) (Table 6-16).

Of the pesticides analyzed, only p,p'-DDD and p,p'-DDE were detected; p,p'-DDE was detected in all samples, while p,p'-DDD was only detected in 14 percent of the samples tested. The mean p,p'-DDE concentration (218 μg/kg) was about 40 percent higher relative to the mean concentration in whole tree swallows from the Little Lake Butte des Morts Reach (155 μg/kg). Although p,p'-DDD was not detected in whole tree swallows in the Little Lake Butte des Morts Reach, the mean concentration of p,p'-DDD (6.1 μg/kg) in the De Pere to Green Bay Reach tree swallows was about 36 times lower than the mean concentration of DDE (218 μg/kg) in this reach.

Exposure point concentrations for piscivorous birds were estimated using the equation presented in Section 6.2.4 and the input parameter values presented in Table 6-4. The fish species selected for modeling were alewife as a trophic level 3 fish, and walleye as a trophic level 4 fish. Resulting estimated exposure point concentrations are presented in Table 6-39.

The mean estimated exposure concentrations for mercury were 49.0 μg/kg-BW/day (common tern), 45.2 μg/kg-BW/day (Forster's tern), 18.9 μg/kg-BW/day (double-crested cormorant), and 10.2 μg/kg-BW/day (bald eagle). The mean estimated exposure concentrations for total PCBs were 1,274 μg/kg-BW/day (common tern), 1,175 μg/kg-BW/day (Forster's tern), 492 μg/kg-BW/day (double-crested cormorant), and 750 μg/kg-BW/day (bald eagle). The mean estimated exposure concentrations for dieldrin were 10.3 μg/kg-BW/day (common tern), 9.5 μg/kg-BW/day (Forster's tern), 4.0 μg/kg-BW/day (double-crested cormorant), and 2.7 μg/kg-BW/day (bald eagle). The mean estimated exposure concentrations for p,p'-DDE were 51.1 μg/kg-BW/day (common tern), 47.1 μg/kg-BW/day (Forster's

tern), 19.7 μg/kg-BW/day (double-crested cormorant), and 25.8 μg/kg-BW/day (bald eagle).

Mammals. Exposure point concentrations for mink were estimated using the equation presented in Section 6.2.4 and the input parameter values presented in Table 6-4. Carp was the selected fish species for modeling. Non-interpolated and interpolated sediment total PCB concentrations were each used in the exposure modeling. Resulting estimated exposure point concentrations are presented in Table 6-40. The mean estimated exposure concentration for mercury was 16.5 μg/kg-BW/day. The mean estimated exposure concentration for total PCBs (N), total PCBs (I_0), and total PCBs (I_d) were 1,290, 1,284, and 1,284 μg/kg-BW/day, respectively. The mean estimated exposure concentration for dieldrin was 4.0 μg/kg-BW/day. For p,p'-DDE, the mean estimated exposure concentration was 37.6 μg/kg-BW/day.

Green Bay Zone 2

Water. COPCs analyzed in surface water include lead, mercury, and total PCBs. These data are presented in Table 6-41. Each of these compounds were detected.

Filtered and unfiltered lead were detected at a frequency of 100 percent (n = 2). Concentrations of filtered lead were not measured in Green Bay Zone 1; however, the mean filtered lead concentration in Green Bay Zone 2 (169 ng/L) was only 5 percent of the concentration measured in Green Bay Zone 1 (3,113 ng/L) (Table 6-36). Filtered and unfiltered mercury were detected at frequencies of 20 and 18 percent, respectively (n = 10 and 11, respectively). The mean filtered mercury concentration (391 ng/L) was about 60 percent of the mean unfiltered concentration (629 ng/L), indicating that approximately 40 percent of the mercury is in the particulate fraction. This filtered mercury concentration was approximately 80 times greater than the filtered mercury concentration in Green Bay Zone 1, while the unfiltered mercury concentration was about 23 times greater than what was measured in Green Bay Zone 1.

Total PCBs were not analyzed in unfiltered samples. Total PCBs were analyzed and frequently detected in all filtered and particulate samples (n = 63 and 71, respectively) (Table 6-41). Concentrations of total PCBs in filtered and particulate samples were summed to estimate a total water concentration of total PCBs. The estimated mean concentration of 17.8 ng/L is about 70 percent less than the estimated mean concentration of 60.9 ng/L in De Pere to Green Bay Reach (Table 6-35).

Sediment. COPCs analyzed in surface sediment include arsenic, lead, mercury, total PCBs, and chlorinated pesticides (dieldrin, p,p'-DDD, p,p'-DDE, p,p'-DDT).

These data are presented in Table 6-42. None of the chlorinated pesticides were detected (n = 11).

Of the metals (n = 11), arsenic was detected at a frequency of 91 percent and the mean concentration (2.1 mg/kg) is 79 percent less than the concentration measured in surface sediment in Green Bay Zone 1 (10.1 mg/kg) (Table 6-36). Lead was detected at a frequency of 100 percent and the mean concentration (19.7 mg/kg) is about 74 percent less than the measured concentration in Green Bay Zone 1 (75.7 mg/kg). Mercury was detected at a frequency of 82 percent and the mean concentration (0.5 mg/kg) was half the concentration measured in Green Bay Zone 1 (1.0 mg/kg).

Non-interpolated PCBs were detected at a frequency of 93 percent (n = 15). The mean concentration (251 μg/kg) was only 6 percent of the mean sediment concentration measured in the Green Bay Zone 1 (4,184 μg/kg). The mean concentration of interpolated PCBs was 1,132 μg/kg (I_d).

Sediment concentrations of PCB coplanar congeners of concern (congener numbers 77, 81, 105, 118, 126, and 169) are given in Table 6-43. Only congener number 169 was not detected (n = 11). Frequency of detection of the other congeners ranged from 45 to 100 percent (n = 11 to 15). RME concentrations of PCB congeners ranged from 0.1 μg/kg (congener 126) to 15.9 μg/kg (congener 118) as compared to RME congener concentrations in Green Bay Zone 1 which ranged from 0.2 μg/kg (congener 81) to 27.0 μg/kg (congener 77) (Table 6-37). These results are similar to congener data for all other reaches examined thus far, where congener 126 had the minimum RME concentration and congener 118 had the greatest concentration.

Fish. COPCs analyzed in whole fish identified as important receptors include mercury, total PCBs, and chlorinated pesticides (dieldrin, o,p'-DDD, o,p'-DDE, o,p'-DDT, p,p'-DDD, p,p'-DDE, p,p'-DDT). These data are presented in Table 6-44. Fish analyzed in this reach include gizzard shad, common shiner, emerald shiner, rainbow smelt, alewife, yellow perch, carp, and walleye. In all fish analyzed, o,p'-DDD (gizzard shad, rainbow smelt, carp, and walleye), o,p'-DDT (gizzard shad, rainbow smelt, carp, and walleye), and p,p'-DDT (alewife, gizzard shad, rainbow smelt, yellow perch, carp, and walleye) were not detected.

Mercury was detected in all fish species analyzed, except gizzard shad and yellow perch. The only mercury concentration measured in carp (0.12 mg/kg), and the mean concentration measured in walleye (0.21 mg/kg) are similar to the single concentrations measured in these fish in Little Rapids to De Pere Reach (0.15 and 0.16 mg/kg, respectively) (Table 6-31). Mean mercury concentrations were

comparatively lower in rainbow smelt (0.03 mg/kg) where all fish analyzed (n = 4) had detected levels of mercury and alewife (0.10 mg/kg) where the detection frequency was 40 percent (n = 5).

Total PCBs were detected at a frequency of 100 percent in all fish species analyzed and the mean concentrations in the order of highest to lowest is: carp (6,637 μg/kg), walleye (6,539 μg/kg), common and emerald shiner (3,520 μg/kg), alewife (2,599 μg/kg), gizzard shad (1,852 μg/kg), golden shiner (1,385 μg/kg), yellow perch (1,206 μg/kg), and rainbow smelt (1,049 μg/kg) (Table 6-44). As compared to mean concentrations of total PCBs in fish from the Little Rapids to De Pere Reach (Table 6-31), concentrations increased by 69 percent (carp), 2.1 times (walleye), 92 percent (yellow perch), 5.3 times (gizzard shad), and 36 percent (golden shiner).

Unlike the river reaches evaluated, dieldrin in Green Bay zones 1 and 2 was detected in all species analyzed, except yellow perch (Table 6-44). In contrast to the pattern observed in PCBs, the highest mean level of dieldrin was measured in walleye (37.3 μg/kg), with alewife (21.0 μg/kg) and carp (20.8 μg/kg) following as the next highest concentrations. Mean concentrations measured in gizzard shad (10.5 μg/kg) and rainbow smelt (7.5 μg/kg) were markedly lower. In comparison, dieldrin was not detected in carp or yellow perch in the Little Rapids to De Pere Reach (Table 6-31), and mean concentrations increased by 11 times in walleye.

For the remaining chlorinated pesticides, only p,p'-DDD, p,p'-DDE, and o-p'-DDE were detected in fish (Table 6-44). Concentrations of p,p'-DDE were measured in all species analyzed with the exception of rainbow smelt. Detection frequencies of p,p'-DDE were 36 percent in gizzard shad (n = 22) and 100 percent in all other species (n = 5 to 14). As compared to mean p,p'-DDE concentrations in the Little Rapids to De Pere Reach fish (Table 6-31), carp (197 μg/kg) and walleye (353 μg/kg) concentrations had increased by almost three times and concentrations in yellow perch (32.9 μg/kg) had doubled. As with the analytical results for fish collected from the river reaches, o,p'-DDE and p,p'-DDD were less frequently detected than p,p'-DDE. Detectable concentrations of p,p'-DDD were measured in all species analyzed except yellow perch and detection frequencies ranged from 5 percent (gizzard shad) to 23 percent (carp). Mean concentrations of p,p'-DDD ranged from 7.3 μg/kg (alewife) to 31.8 μg/kg (carp). In the Little Rapids to De Pere Reach, p,p'-DDD was only detected in carp and the maximum concentration was 8.0 μg/kg. Carp and walleye were the only fish analyzed that had detected concentrations of o,p'-DDE and frequencies of detection were 75 and 100 percent, respectively. Mean concentrations were 50 μg/kg (carp) and 85 μg/kg (walleye) and as compared to walleye in the Little Rapids to De Pere Reach, concentrations had increased by 86 percent.

Concentrations of dioxin, furan, and PCB congeners in whole fish are presented in Table 6-45. Dioxin and furan congener detection frequencies ranged from 0 to 100 percent with sample sizes of three whole fish. PCB congener detection frequencies ranged from 0 to 100 percent and sample sizes ranged from 5 to 83 whole fish. The species of fish analyzed for these compounds included carp (dioxin, furan, and PCB congeners), walleye (dioxin, furan, and PCB congeners), yellow perch (PCB congeners), alewife (PCB congeners), and gizzard shad (PCB congeners).

The mean concentration of 2,3,7,8-TCDD was greater in walleye (0.0014 μg/kg) than in carp (0.00098 μg/kg). The mean concentration of 2,3,7,8-TCDF was also greater in walleye (0.017 μg/kg) than in carp (0.0029 μg/kg). As compared to mean concentrations of these compounds in the Little Rapids to De Pere Reach (Table 6-32), 2,3,7,8-TCDD concentrations in carp increased by 78 percent, 2,3,7,8-TCDF concentrations in carp increased by 2.4 times, 2,3,7,8-TCDD concentrations in walleye increased by 75 percent, and 2,3,7,8-TCDF concentrations in walleye increased by 70 percent.

All PCB congeners of interest analyzed (congeners 77, 81, 105, 118, 126, and 169) were detected in carp, walleye, gizzard shad, and rainbow smelt. Congeners 126 and 169 were not analyzed in rainbow smelt. Overall, PCB congener concentrations were higher in walleye than other species analyzed. PCB congener 169 was not detected in yellow perch, alewife, or golden shiner and PCB congener 81 was not detected in yellow perch. As was the case for fish in the river reaches examined, mean concentrations of PCB congener 118 exceeded other congeners and ranged from 26.9 μg/kg (rainbow smelt) to 174 μg/kg (walleye).

Compared to PCB congener concentrations in fish from Little Rapids to De Pere Reach, Green Bay zones 1 and 2 congener concentrations were greater in all fish species analyzed from both locations. These comparisons were limited to carp, walleye, yellow perch, and golden shiner. Detected mean PCB congener concentrations ranged from 0.01 μg/kg (congener 126 in yellow perch) to 174 μg/kg (congener 118 in walleye) compared to a range of 0.01 μg/kg (congener 77 in yellow perch) to 77.0 μg/kg (congener 118 in walleye) in the Little Rapids to De Pere Reach.

Birds. COPCs analyzed in birds identified as important receptors include total PCBs, and chlorinated pesticides (dieldrin, o,p'-DDD, o,p'-DDE, o,p'-DDT, p,p'-DDD, p,p'-DDE, p,p'-DDT). These data are presented in Table 6-46. Species analyzed in this reach were double-crested cormorants, common terns, Forster's terns, and tree swallows. None of the o,p- isomers of DDT and its metabolites were detected.

Analyses included double-crested cormorant eggs, brains, and whole birds; Forster's tern eggs; common tern eggs; and whole tree swallows. The mean concentrations of total PCBs, in order of decreasing concentration were: 13,944 µg/kg (double-crested cormorant eggs), 11,026 µg/kg (whole double-crested cormorant), 5,077 µg/kg (Forster's tern egg), 4,819 µg/kg (common tern egg), 3,700 µg/kg (double-crested cormorant brain), and 2,980 µg/kg (whole tree swallow). This mean concentration of total PCBs in tree swallows represents a decrease of 4 percent compared to the mean concentration observed in whole tree swallows from Green Bay Zone 1 (3,118 µg/kg) (Table 6-38). Detection frequencies of total PCBs in these tissues were all 100 percent and sample sizes ranged from 5 to 74.

Dieldrin was detected in all tissues analyzed except for whole tree swallows at frequencies of 94 to 100 percent. The mean concentrations in order of decreasing concentration were: 224 µg/kg (double-crested cormorant egg), 196 µg/kg (whole double-crested cormorant), 85 µg/kg (common tern egg), 48.2 µg/kg (double-crested cormorant brain), and 47.6 µg/kg (Forster's tern egg).

Concentrations of p,p'-DDD, p,p'-DDE, and p,p'-DDT were detected in at least two of the tissues analyzed. The only tissue where p,p'-DDD was not detected was in double-crested cormorant brain. The mean concentrations of p,p'-DDD in the other tissues analyzed were 2.1 µg/kg (common tern egg), 7.3 µg/kg (whole double-crested cormorant), 15.0 µg/kg (double-crested cormorant egg), and 6.5 µg/kg (whole tree swallow). This mean concentration of total PCBs in tree swallows represents an increase of 7 percent compared to the mean concentrations observed in whole tree swallows from Green Bay Zone 1 (6.1 µg/kg) (Table 6-38). All tissues had detected levels of p,p'-DDE and mean concentrations ranged from 447 µg/kg (Forster's tern egg) to 4,132 µg/kg (double-crested cormorant eggs). The mean concentration of 128 µg/kg p,p'-DDE in whole tree swallows represents a decrease of 41 percent compared to the mean concentration of 218 µg/kg p,p'-DDE in whole tree swallows from Green Bay Zone 1 (Table 6-38). Detected concentrations of p,p'-DDT were only measured in whole double-crested cormorants and double-crested cormorant eggs at concentrations of 8.1 and 7.6 µg/kg, respectively. These patterns of DDT and metabolite detection are similar to what was seen in fish, where concentrations of p,p'-DDE were the highest.

Exposure point concentrations for all piscivorous birds were estimated using the equation presented in Section 6.2.4 and the input parameter values presented in Table 6-4. The fish species selected for modeling were alewife as a trophic level 3 fish, and walleye as a trophic level 4 fish. The COPC concentrations in these fish available for modeling were mercury, total PCBs, dieldrin, and p,p'-DDE. These exposure point concentrations, presented in Table 6-47, were calculated

using combined Green Bay zones 1 and 2 fish data. The only difference between these exposure estimates and the exposure estimates for piscivorous birds in Green Bay Zone 1 (Table 6-39) are the water concentrations that were used in the modeling. These differences in water concentrations did not result in differences of more than one-tenth of a microgram in estimates of exposure concentrations. The mean estimated exposure concentrations for mercury were 49.1 μg/kg-BW/day (common tern), 45.3 μg/kg-BW/day (Forster's tern), 19.0 μg/kg-BW/day (double-crested cormorant), and 10.2 μg/kg-BW/day (bald eagle). The mean estimated exposure concentrations for total PCBs were 1,274 μg/kg-BW/day (common tern), 1,174 μg/kg-BW/day (Forster's tern), 492 μg/kg-BW/day (double-crested cormorant), and 750 μg/kg-BW/day (bald eagle). The mean estimated exposure concentrations for dieldrin were 10.3 μg/kg-BW/day (common tern), 9.5 μg/kg-BW/day (Forster's tern), 4.0 μg/kg-BW/day (double-crested cormorant), and 2.7 μg/kg-BW/day (bald eagle). The mean estimated exposure concentrations for p,p'-DDE were 51.1 μg/kg-BW/day (common tern), 47.1 μg/kg-BW/day (Forster's tern), 19.7 μg/kg-BW/day (double-crested cormorant), and 25.8 μg/kg-BW/day (bald eagle).

PCB congeners and dioxins/furans were analyzed in whole tree swallows (Table 6-48), whole double-crested cormorants, and double-crested cormorant eggs (Table 6-49), common tern eggs (Table 6-50), and Forster's tern eggs (Table 6-51). Mean concentrations of PCB congeners detected in whole tree swallows were 6.1 μg/kg (congener 77), 27.8 μg/kg (congener 118/106), and 0.3 μg/kg (congener 126), respectively. Congeners were not analyzed in tree swallows collected in Green Bay Zone 1, but they were analyzed in whole tree swallows from Little Lake Butte des Morts Reach. As compared to congener concentrations measured in the Little Lake Butte des Morts Reach, congener 105 had increased by over two times, congener 118/106 had increased by 47 percent, and congener 126 had increased by 3 percent.

The only dioxin congeners detected in whole double-crested cormorants were 1,2,3,4,6,7,8,9-OCDD and 2,3,7,8-TCDD at mean concentrations of 0.14 and 0.0047 μg/kg, respectively. Only 2,3,7,8-TCDD was detected in double-crested cormorant eggs at a mean concentration of 0.012 μg/kg. In contrast, in common tern eggs, all dioxin congeners analyzed were detected except for 1,2,3,4,6,7,8,9-OCDF, and in Forster's tern eggs all dioxin congeners were detected. The mean detected dioxin congener concentrations in common tern eggs ranged from 0.0008 μg/kg (2,3,4,6,7,8-HXCDF) to 0.1 μg/kg (1,2,3,4,6,7,8,9-OCDD) and mean dioxin congener concentrations in Forster's tern egg ranged from 0.00044 μg/kg (1,2,3,4,6,7,8-HPCDF) to 0.53 μg/kg (1,2,3,4,6,7,8,9-OCDD).

As with dioxin congener concentrations in double-crested cormorants, PCB congener concentrations in eggs were higher than concentrations in whole bodies. Mean egg congener concentrations ranged from 0.1 μg/kg (PCB congener 169) to 551 μg/kg (PCB congener 118/106) and mean whole body concentrations ranged from 0.1 μg/kg (PCB congener 169) to 379 μg/kg (PCB congener 118/106). Mean PCB congener concentrations in common tern eggs ranged from 2.3 μg/kg (PCB congener 169) to 357 μg/kg (PCB congener 118), and in Forster's tern eggs ranged from 2.6 μg/kg (PCB congener 77) to 283 μg/kg (PCB congener 118). These patterns of PCB congener concentrations are similar to what is seen in fish, where generally concentrations of PCB congener 169 were relatively low and concentrations of PCB congener 118 were relatively high.

Mammals. Exposure point concentrations for mink were estimated using the equation presented in Section 6.2.4 and the input parameter values presented in Table 6-4. Carp was the selected fish species for modeling. Non-interpolated and interpolated sediment total PCB concentrations were each used in the exposure modeling. Resulting estimated exposure point concentrations are presented in Table 6-52. The mean estimated exposure concentration for mercury was 14.0 μg/kg-BW/day. The mean estimated exposure concentration for total PCBs (N), and total PCBs (I_d) were 1,271, and 1,275 μg/kg-BW/day, respectively. The mean estimated exposure concentration for dieldrin was 4.0 μg/kg-BW/day. For p,p'-DDE, the mean estimated exposure concentration was 37.6 μg/kg-BW/day.

Green Bay Zone 3A

Water. COPCs analyzed in surface water include mercury and total PCBs. These data are presented in Table 6-53. Mercury was not detected in filtered or unfiltered samples.

Total PCBs were not analyzed in unfiltered samples. Total PCBs were analyzed and frequently detected (>92% of the samples) in filtered and particulate samples (n = 60 and 66, respectively) (Table 6-53). Concentrations of total PCBs in filtered and particulate samples were summed to estimate a total water concentration of total PCBs. The estimated mean concentration of 4.4 ng/L is about 75 percent less than the estimated mean concentration of 17.8 ng/L in Green Bay Zone 2 (Table 6-41).

Sediment. COPCs analyzed in surface sediment include arsenic, lead, mercury, total PCBs, and chlorinated pesticides (dieldrin, p,p'-DDD, p,p'-DDE, p,p'-DDT). These data are presented in Table 6-54. Mercury was not detected (n = 2), and none of the chlorinated pesticides were detected (n = 11).

Of the metals (n = 2), arsenic and lead were detected at a frequency of 100 percent. The mean arsenic concentration (1.5 mg/kg) is 29 percent less than the concentration measured in surface sediment in Green Bay Zone 2 (2.1 mg/kg) (Table 6-42). The mean lead concentration (1.5 mg/kg) is 92 percent less than the concentration measured in surface sediment in Green Bay Zone 2 (19.7 mg/kg).

Non-interpolated PCBs were detected at a frequency of 87 percent (n = 15). The mean concentration (376 μg/kg) was 50 percent higher than the mean sediment concentration measured in Green Bay Zone 2 (251 μg/kg). The mean concentration of interpolated PCBs was 256 μg/kg (I_d), which was approximately 77 percent less than those measured in Green Bay Zone 2 (Table 6-42).

Sediment concentrations of PCB coplanar congeners of concern (congener numbers 77, 81, 105, 118, 126, and 169) are given in Table 6-55. Congener numbers 126 and 169 were not detected (n = 2). Frequency of detection of the other congeners ranged from 50 to 100 percent (n = 2 to 15). RME concentrations of PCB congeners ranged from 0.1 μg/kg (congener 77) to 25.4 μg/kg (congener 118) as compared to RME congener concentrations in Green Bay Zone 2 which ranged from 0.1 μg/kg (congener 126) to 15.9 μg/kg (congener 118) (Table 6-43). These results are similar to congener data for all other reaches examined thus far, where congener 126 had the minimum RME concentration and congener 118 had the highest concentration.

Fish. COPCs analyzed in whole fish identified as important receptors include mercury, total PCBs, and chlorinated pesticides (dieldrin, o,p'-DDD, o,p'-DDE, o,p'-DDT, p,p'-DDD, p,p'-DDE, p,p'-DDT). These data are presented in Table 6-56. Fish analyzed in this reach include gizzard shad, rainbow smelt, alewife, yellow perch, carp, walleye, and brown trout. In all fish analyzed, o,p'-DDD (rainbow smelt), o,p'-DDE (rainbow smelt), o,p'-DDT (rainbow smelt), p,p'-DDD (gizzard shad, rainbow smelt, carp, and yellow perch) and p,p'-DDT (gizzard shad, rainbow smelt, carp, and yellow perch) were not detected.

Mercury was analyzed in carp (n = 1), yellow perch (n = 2), gizzard shad (n = 1), and rainbow smelt (n = 6), but it was only detected in rainbow smelt at a frequency of 67 percent and at a mean concentration of 0.03 mg/kg. This was the same mean concentration detected in rainbow smelt from Green Bay zones 1 and 2.

Total PCBs were detected at frequencies of 97 to 100 percent in all fish species analyzed with mean concentrations (from highest to lowest) of: walleye (4,155 μg/kg), gizzard shad (3,524 μg/kg), brown trout (3,250 μg/kg), carp (2,642 μg/kg),

alewife (907 μg/kg), rainbow smelt (570 μg/kg), and yellow perch (179 μg/kg) (Table 6-56). Mean concentrations of total PCBs in fish were generally much lower than mean concentrations in Green Bay zones 1 and 2. Mean concentrations decreased by 60 percent (carp), 36 percent (walleye), 92 percent (yellow perch), 65 percent (alewife), and 46 percent (rainbow smelt). The single detection in gizzard shad (3,524 μg/kg), however, was 90 percent higher than the mean concentration in Green Bay zones 1 and 2 (1,852 μg/kg). Brown trout could not be compared because they were not part of the zones 1 and 2 food web model.

Dieldrin was detected in all species analyzed, except yellow perch (n = 2) and gizzard shad (n = 1) (Table 6-56). Mean concentrations in the order of highest to lowest are: brown trout (76.0 μg/kg), walleye (43.4 μg/kg), alewife (21.5 μg/kg), carp (17.9 μg/kg), and rainbow smelt (14.4 μg/kg). In contrast to mean concentrations of total PCBs, where walleye had accumulated the highest concentrations, mean dieldrin concentrations were 75 percent higher in brown trout than in walleye. As compared to mean dieldrin concentrations measured in Green Bay zones 1 and 2 (Table 6-44), carp, walleye, and alewife concentrations remained similar; however, rainbow smelt concentrations increased by 92 percent.

The only other detected chlorinated pesticide in fish was p,p'-DDE. It was detected in all fish analyzed: gizzard shad (n = 1), rainbow smelt (n = 12), yellow perch (n = 2), and carp (n = 1) at detection frequencies that ranged from 17 (rainbow smelt) to 100 percent (gizzard shad and carp). The mean concentration in yellow perch was 6 μg/kg and in rainbow smelt was 30 μg/kg. Concentrations in carp and gizzard shad were 25 and 150 μg/kg, respectively. Concentrations in carp and yellow perch represent a decline of more than 80 percent as compared to mean concentrations measured in Green Bay zones 1 and 2 fish (Table 6-44). The single concentration measured in gizzard shad, however, represents a more than doubling of the mean concentration measured in Green Bay zones 1 and 2 gizzard shad (Table 6-44). The mean concentration in rainbow smelt could not be compared because these fish had no detected concentration of p,p'-DDE in Green Bay zones 1 and 2.

Concentrations of PCB congeners in whole fish are presented in Table 6-57. Where detected, PCB congener detection frequencies ranged from 91 to 100 percent and sample sizes ranged from 1 to 20 whole fish. The species of fish analyzed for these compounds included carp, walleye, yellow perch, brown trout, alewife, gizzard shad, and rainbow smelt. Congeners 126 and 169 were not analyzed in brown trout, alewife, or rainbow smelt. Additionally, for these species, congener 77 coeluted with congener 110 and congener 105 coeluted with congeners 132 and 153. Therefore, carp, walleye, yellow perch, and gizzard shad

were the only fish for which all congeners were measured (congeners 77, 81, 105, 118, 126, and 169) and none of the congeners coeluted. Congener 81 was not detected in yellow perch or gizzard shad, congener 126 was not detected in carp or yellow perch, and congener 169 was not detected in yellow perch or gizzard shad. Overall, PCB congener concentrations were highest in brown trout, followed by walleye, and then gizzard shad, alewife, carp, rainbow smelt, and yellow perch. As in other locations examined, concentrations of PCB congener 118 exceeded other congeners except when compared with coeluting PCB congeners. The mean concentration of congener 118 ranged from 8.3 μg/kg (yellow perch) to 125 μg/kg (walleye).

Compared to PCB congener concentrations in fish from Green Bay zones 1 and 2 (Table 6-45), mean congener concentrations in fish from Green Bay Zone 3A were lower for all congeners in carp, yellow perch, rainbow smelt, and alewife (not including coeluted congeners). In walleye, all mean congener concentrations decreased except congeners 77 and 169 which increased by 78 percent and 5.3 times, respectively. All congeners increased, however, in gizzard shad, except congener 81 which was not detected in Green Bay Zone 3A (n = 1). Of the congeners that did not coelute, detected mean PCB congener concentrations ranged from 0.01 μg/kg (congener 169 in carp) to 125 μg/kg (congener 118 in walleye) compared to a range of 0.01 μg/kg (congener 126 in yellow perch) to 174 μg/kg (congener 118 in walleye) in the Green Bay zones 1 and 2.

Birds. COPCs analyzed in birds identified as important receptors include total PCBs and chlorinated pesticides (dieldrin, p,p'-DDD, p,p'-DDE, p,p'-DDT). These data are presented in Table 6-58. The only species analyzed in this reach was bald eagle. Only p,p'-DDT was not detected.

The only other area where bald eagles were analyzed is the Appleton to Little Rapids Reach (Table 6-23). Mercury in eggs was detected at a frequency of 100 percent (n = 3) and the mean concentration was 0.3 mg/kg. This mean concentration is 80 percent less than the single mercury concentration detected in the bald eagle egg from the Appleton to Little Rapids Reach.

The concentration of total PCBs in a single bald eagle egg (13,000 μg/kg) was 64 percent lower than the egg concentration in the Appleton to Little Rapids Reach (36,000 μg/kg).

Only a single egg was also analyzed for chlorinated pesticides. The dieldrin concentration of 200 μg/kg was 2.9 times higher than the concentration of 70 μg/kg in the Appleton to Little Rapids Reach. The p,p'-DDD concentration of 120 μg/kg was 25 percent less than the p,p'-DDD concentration of 160 μg/kg in

the Appleton to Little Rapids Reach. The p,p'-DDE concentration of 2,400 μg/kg was 2.2 times higher than the p,p'-DDE concentration of 1,100 μg/kg in the Appleton to Little Rapids Reach.

Because of the lack of data, exposure point concentrations for piscivorous bird receptors (common terns, Forster's terns, double-crested cormorants, and bald eagle) were estimated using the equation presented in Section 6.2.4 and the input parameter values presented in Table 6-4. The fish species selected for modeling were alewife (total PCBs and dieldrin) and rainbow smelt (mercury and p,p'-DDE) as a trophic level 3 fish, and walleye as a trophic level 4 fish. The COPC concentrations in these fish available for modeling were mercury, total PCBs, dieldrin, and p,p'-DDE. Resulting estimated exposure point concentrations are presented in Table 6-59. The mean estimated exposure concentrations for mercury were 14.7 μg/kg-BW/day (common tern), 13.6 μg/kg-BW/day (Forster's tern), 5.7 μg/kg-BW/day (double-crested cormorant), and 2.3 μg/kg-BW/day (bald eagle). The mean estimated exposure concentrations for total PCBs were 444 μg/kg-BW/day (common tern), 410 μg/kg-BW/day (Forster's tern), 172 μg/kg-BW/day (double-crested cormorant), and 334 μg/kg-BW/day (bald eagle). The mean estimated exposure concentrations for dieldrin were 10.5 μg/kg-BW/day (common tern), 9.7 μg/kg-BW/day (Forster's tern), 4.1 μg/kg-BW/day (double-crested cormorant), and 2.6 μg/kg-BW/day (bald eagle). The mean estimated exposure concentrations for p,p'-DDE were 14.7 μg/kg-BW/day (common tern), 13.6 μg/kg-BW/day (Forster's tern), 5.7 μg/kg-BW/day (double-crested cormorant), and 2.3 μg/kg-BW/day (bald eagle).

No PCB congener data in birds is available for this reach.

Mammals. Exposure point concentrations for mink were estimated using the equation presented in Section 6.2.4 and the input parameter values presented in Table 6-4. Carp was the selected fish species for modeling. Non-interpolated and interpolated sediment total PCB concentrations were each used in the exposure modeling. Resulting estimated exposure point concentrations are presented in Table 6-60. The mean estimated exposure concentration for mercury was 4.9 μg/kg-BW/day. The mean estimated exposure concentration for total PCBs (N) and total PCBs (I_d) was 507 μg/kg-BW/day. The mean estimated exposure concentration for dieldrin was 3.4 μg/kg-BW/day. The mean estimated exposure concentration for p,p'-DDE was 4.8 μg/kg-BW/day.

Green Bay Zone 3B

Water. COPCs analyzed in surface water include mercury and total PCBs. These data are presented in Table 6-61. Only mercury in filtered samples was not detected, although it was detected in unfiltered samples.

Unfiltered mercury was detected at a frequency of 14 percent (n = 7) and the mean concentration (47.3 ng/L) is about 92 percent less than the mean unfiltered mercury concentration (391 ng/L) measured in Green Bay Zone 2 (Table 6-41). Mercury was not detected in Green Bay Zone 3A.

Total PCBs were not analyzed in unfiltered samples. Total PCBs were analyzed and frequently detected (>88% of the samples) in filtered and particulate samples (n = 40 and 45, respectively) (Table 6-61). Concentrations of total PCBs in filtered and particulate samples were summed to estimate a total water concentration of total PCBs. The estimated mean concentration of 3.7 ng/L is similar to the estimated mean concentration of 4.4 ng/L in Green Bay Zone 3A (Table 6-53) and approximately 80 percent less than the estimated mean concentration of 17.8 ng/L in Green Bay Zone 2 (Table 6-41).

Sediment. COPCs analyzed in surface sediment include arsenic, lead, mercury, total PCBs, and chlorinated pesticides (dieldrin, p,p'-DDD, p,p'-DDE, p,p'-DDT). These data are presented in Table 6-62. None of the chlorinated pesticides were detected (n = 4).

Of the metals (n = 4), arsenic and lead were detected at a frequency of 100 percent and mercury was detected at a frequency of 25 percent. The mean arsenic concentration (8.6 mg/kg) is more than five times greater than the concentration in surface sediment in Green Bay Zone 3A (1.5 mg/kg) (Table 6-54) and approximately four times the concentration in surface sediment in Green Bay Zone 2 (2.1 mg/kg) (Table 6-42). The mean lead concentration (29.9 mg/kg) is about 20 times greater than the concentration measured in surface sediment in Green Bay Zone 3A (1.5 mg/kg) and 1.5 times the concentration in surface sediment in Green Bay Zone 2 (19.7 mg/kg).

Non-interpolated PCBs were detected at a frequency of 88 percent (n = 40). The mean concentration (542 μg/kg) is 44 percent greater than the mean concentration measured in Green Bay Zone 3A (376 μg/kg) (Table 6-54) and is approximately twice the mean concentration measured in Green Bay Zone 2 (251 μg/kg) (Table 6-42). The mean concentration of interpolated PCBs was 482 μg/kg (I_d) which was almost twice the concentration estimated in Green Bay Zone 3A, but 57 percent less than the mean concentration estimated in Green Bay Zone 2.

Sediment concentrations of PCB coplanar congeners of concern (congener numbers 77, 81, 105, 118, 126, and 169) are given in Table 6-63. Congener numbers 126 and 169 were not detected (n = 4). Frequency of detection of the other congeners ranged from 86 to 100 percent (n = 4 to 37). RME concentrations of PCB congeners ranged from 0.8 μg/kg (congener 81) to 31.0

μg/kg (congener 118) as compared to RME congener concentrations in Green Bay Zone 3A which ranged from 0.1 μg/kg (congener 77) to 25.4 μg/kg (congener 118) (Table 6-55). These results are similar to congener data for other reaches examined thus far, where congener 126 had the minimum RME concentration and congener 118 had the highest concentration.

Fish. COPCs analyzed in whole fish identified as important receptors include mercury, total PCBs, and chlorinated pesticides (dieldrin, p,p'-DDD, p,p'-DDE, p,p'-DDT). These data are presented in Table 6-64. Fish analyzed in this reach include gizzard shad, rainbow smelt, alewife, yellow perch, carp, walleye, and brown trout. In all fish analyzed, p,p'-DDD (alewife, gizzard shad, yellow perch, carp, and walleye) and p,p'-DDT (alewife, gizzard shad, yellow perch, carp, and walleye) were not detected.

Mercury was analyzed in carp (n = 2), walleye (n = 3), yellow perch (n = 2), alewife (n = 1), and gizzard shad (n = 1). Mercury was not detected in yellow perch, alewife, or gizzard shad. Mean detected concentrations of mercury in carp and walleye were 0.1 and 0.3 mg/kg, respectively. Mercury was not detected in carp or walleye in Green Bay Zone 3A, but compared to mean concentrations in carp and walleye from Green Bay zones 1 and 2 (Table 6-44), the mean carp concentration had increased by 83 percent and the mean walleye concentration had increased by 19 percent.

Total PCBs were detected in all fish species analyzed and the mean concentrations in the order of highest to lowest were: walleye (6,429 μg/kg), carp (4,947 μg/kg), brown trout (2,223 μg/kg), alewife (1,821 μg/kg), rainbow smelt (733 μg/kg), gizzard shad (635 μg/kg), and yellow perch (154 μg/kg) (Table 6-64). As compared to mean total PCB concentrations in these species from Green Bay Zone 3A (Table 6-56), the concentration in walleye increased by 55 percent, carp increased by 87 percent, brown trout decreased by 32 percent, alewife increased by 100 percent, rainbow smelt increased by 29 percent, gizzard shad decreased by 82 percent, and yellow perch decreased by 14 percent. Compared to mean total PCB concentrations in these species from Green Bay zones 1 and 2 (Table 6-44), the concentration in walleye decreased by 2 percent, carp decreased by 25 percent, alewife decreased by 30 percent, rainbow smelt decreased by 30 percent, gizzard shad decreased by 66 percent, and yellow perch decreased by 87 percent.

Dieldrin was detected in all species analyzed, except yellow perch and gizzard shad (Table 6-64), which was also the case in Green Bay Zone 3A. Mean concentrations in the order of highest to lowest were: brown trout (72.0 μg/kg), walleye (50.1 μg/kg), carp (43.2 μg/kg) alewife (19.1 μg/kg), and rainbow smelt (14.7 μg/kg). As compared to mean dieldrin concentrations measured in Green

Bay Zone 3A fish (Table 6-56), the mean dieldrin concentration increased by 2.4 times in carp, increased by 15 percent in walleye, decreased by 9 percent in alewife, decreased by 5 percent in brown trout, and increased by 2 percent in rainbow smelt. As compared to mean dieldrin concentrations measured in Green Bay zones 1 and 2 fish (Table 6-44), the mean dieldrin concentration increased by 2.1 times in carp, increased by 30 percent in walleye, decreased by 9 percent in alewife, and increased by 96 percent in rainbow smelt.

The only other chlorinated pesticide detected was p,p'-DDE, which was detected in all species at frequencies of 67 percent (walleye) and 100 percent (alewife, gizzard shad, yellow perch, and carp). Mean concentrations of p,p'-DDE in order of highest to lowest were: 207 μg/kg (walleye), 126 μg/kg (carp), 80.0 μg/kg (alewife), 37 μg/kg (gizzard shad), and 21.0 μg/kg (yellow perch), respectively. Compared to Green Bay Zone 3A, the mean concentration of p,p'-DDE in Green Bay Zone 3B was 5.0 times higher in carp, 3.5 times higher in yellow perch, and 75 percent lower in gizzard shad. Comparisons could not be made between walleye and alewife because they were not analyzed in Green Bay Zone 3A. Compared to Green Bay zones 1 and 2 fish, the mean concentration of p,p'-DDE in Green Bay Zone 3B was 36 percent lower in carp and yellow perch, 41 percent lower in walleye, 23 percent lower in alewife, and 42 percent lower in gizzard shad.

Concentrations of PCB congeners in whole fish are presented in Table 6-65. Where detected, PCB congener detection frequencies ranged from 50 to 100 percent and sample sizes ranged from 1 to 25 whole fish. The species of fish analyzed for these compounds included carp, walleye, yellow perch, alewife, brown trout, gizzard shad, and rainbow smelt. Congeners 126 and 169 were not analyzed in rainbow smelt. Additionally, in rainbow smelt congener 77 coeluted with congener 110 and congener 105 coeluted with congeners 132 and 153. In the other fish examined, no congeners coeluted. All congeners measured were detected in brown trout, walleye, and rainbow smelt. Congeners 126 and 169 were not detected in carp, yellow perch, gizzard shad, or alewife; congener 81 was not detected in gizzard shad or yellow perch. Congeners 105 and 118 were detected in all species. Overall, PCB congener concentrations were highest in walleye, followed by carp, and then brown trout, rainbow smelt, alewife, gizzard shad, and yellow perch. Comparison of relative fractions of various congeners indicate that results in rainbow smelt are artificially high due to coeluting congeners.

Compared to PCB congener concentrations in fish from Green Bay Zone 3A (Table 6-57), mean congener concentrations in fish from Green Bay Zone 3B were higher in carp, walleye, and rainbow smelt, about the same in yellow perch, and

lower in alewife, brown trout, and gizzard shad. In walleye, concentrations of congeners 77, 81, and 126 had decreased by 49, 30, and 82 percent, respectively, and congeners 105, 118, and 169 increased by 49 percent, 30 percent and by 12 times when compared to Green Bay zones 1 and 2. The same trend was present when comparing Green Bay Zone 3B to Green Bay Zone 3A where concentrations of congeners 77, 81, and 126 had decreased by 71, 4, and 78 percent, respectively, and congeners 105, 118, and 169 increased by 63 percent, 82 percent, and by 2.2 times. Of the congeners that did not coelute, detected mean PCB congener concentrations ranged from 0.05 μg/kg (congener 77 in alewife) to 227 μg/kg (congener 118 in walleye) as compared to a range of 0.01 μg/kg (congener 169 in carp) to 125 μg/kg (congener 118 in walleye) in Green Bay Zone 3A and a range of 0.01 μg/kg (congener 126 in yellow perch) to 174 μg/kg (congener 118 in walleye) in Green Bay zones 1 and 2.

Birds. COPCs analyzed in birds identified as important receptors include total PCBs, and chlorinated pesticides (dieldrin, o,p'-DDD, o,p'-DDE, o,p'-DDT, p,p'-DDD, p,p'-DDE, p,p'-DDT). These data are presented in Table 6-66. The only species analyzed in this reach was double-crested cormorant. None of the o,p- isomers of DDT and its metabolites were detected.

A total of 21 whole birds were analyzed for total PCBs and 20 whole samples for pesticides. Total PCBs were detected at a frequency of 95 percent with a mean concentration of 5,384 μg/kg. This mean concentration was 51 percent lower than the mean concentration (11,026 μg/kg) observed in whole double-crested cormorants in Green Bay Zone 2 (Table 6-46). Dieldrin was detected at a frequency of 95 percent with a mean concentration of 128 μg/kg. This mean concentration was 35 percent less than the mean concentration (196 μg/kg) in Green Bay Zone 2. The pesticides p,p'-DDD, p,p'-DDE, and p,p'-DDT were detected at frequencies of 15, 100, and 55 percent, respectively, and at mean concentrations of 6.3, 2,010, and 10.9 μg/kg, respectively. As compared to mean concentrations in Green Bay Zone 2, p,p'-DDD decreased by 14 percent, p,p'-DDE decreased by 27 percent, and p,p'-DDT increased by 35 percent.

Because of the lack of data, exposure point concentrations for piscivorous bird receptors (common terns, Forster's terns, double-crested cormorants, and bald eagles) were estimated using the equation presented in Section 6.2.4 and the input parameter values presented in Table 6-4. The fish species selected for modeling were alewife as a trophic level 3 fish, and walleye as a trophic level 4 fish. The COPC concentrations in these fish available for modeling were mercury, total PCBs, dieldrin, and p,p'-DDE. Resulting estimated exposure point concentrations are presented in Table 6-67. The mean estimated exposure concentrations for mercury were 12.3 μg/kg-BW/day (common tern), 11.3 μg/kg-

BW/day (Forster's tern), 4.7 μg/kg-BW/day (double-crested cormorant), and 15.6 μg/kg-BW/day (bald eagle). The mean estimated exposure concentrations for total PCBs were 892 μg/kg-BW/day (common tern), 823 μg/kg-BW/day (Forster's tern), 345 μg/kg-BW/day (double-crested cormorant), and 594 μg/kg-BW/day (bald eagle). The mean estimated exposure concentrations for dieldrin were 9.3 μg/kg-BW/day (common tern), 8.6 μg/kg-BW/day (Forster's tern), 3.6 μg/kg-BW/day (double-crested cormorant), and 5.1 μg/kg-BW/day (bald eagle). The mean estimated exposure concentrations for p,p'-DDE were 39.2 μg/kg-BW/day (common tern), 36.2 μg/kg-BW/day (Forster's tern), 15.1 μg/kg-BW/day (double-crested cormorant), and 16.1 μg/kg-BW/day (bald eagle).

PCB congeners (congeners 77, 105, 118, 126, and 169) were measured in whole double-crested cormorants from Green Bay Zone 3B (n = 16) (Table 6-68). Congener 118 coeluted with congener 106. The mean congener concentrations ranged from 0.1 μg/kg (congener 169) to 215 μg/kg (congener 118/106). These concentrations are less than or equal to mean whole body concentrations measured in Green Bay Zone 2 which ranged from 0.1 μg/kg (congener 169) to 379 μg/kg (congener 118/106) (Table 6-48).

Mammals. Exposure point concentrations for mink were estimated using the equation presented in Section 6.2.4 and the input parameter values presented in Table 6-4. Carp was the selected fish species for modeling. Non-interpolated and interpolated sediment total PCB concentrations were each used in the exposure modeling. Resulting estimated exposure point concentrations are presented in Table 6-69. The mean estimated exposure concentration for mercury was 21.5 μg/kg-BW/day. The mean estimated exposure concentration for total PCBs (N) and total PCBs (I_d) was 949 μg/kg-BW/day. The mean estimated exposure concentration for dieldrin was 8.3 μg/kg-BW/day. The mean estimated exposure concentration for p,p'-DDE was 24.1 μg/kg-BW/day.

Green Bay Zone 4

Water. COPCs analyzed in surface water included mercury and total PCBs. These data are presented in Table 6-70. Mercury was not detected in filtered or unfiltered samples.

Total PCBs were not analyzed in unfiltered samples. Total PCBs were analyzed and frequently detected (>76% of the samples) in filtered and particulate samples (n = 66 and 86, respectively) (Table 6-70). Concentrations of total PCBs in filtered and particulate samples were summed to estimate a total water concentration of total PCBs. The estimated mean concentration of 1.5 ng/L is less than half of both the estimated mean concentration of 3.7 ng/L in Green Bay

Zone 3B (Table 6-61) and the estimated mean concentration of 4.4 ng/L in Green Bay Zone 3A (Table 6-53).

Sediment. COPCs analyzed in surface sediment include arsenic, lead, mercury, total PCBs, and chlorinated pesticides (dieldrin, p,p'-DDD, p,p'-DDE, p,p'-DDT). These data are presented in Table 6-71. None of the chlorinated pesticides were detected (n = 4).

Of the metals (n = 4), arsenic and lead were detected at a frequency of 100 percent and mercury was detected at a frequency of 25 percent. The mean arsenic concentration (5.0 mg/kg) is 42 percent less than the concentration measured in surface sediment in Green Bay Zone 3B (8.6 mg/kg) (Table 6-62), but three times the concentration measured in Green Bay Zone 3A (1.5 mg/kg) (Table 6-54). The mean lead concentration (3.1 mg/kg) is 90 percent less than the concentration measured in surface sediment in Green Bay Zone 3B (29.9 mg/kg), but three times the concentration measured in Green Bay Zone 3A (1.5 mg/kg). The single detection of mercury (0.1 mg/kg) is approximately one-half the single detection in Green Bay Zone 3B (0.2 mg/kg) and mercury was not detected in Green Bay Zone 3A.

Non-interpolated total PCBs were detected at a frequency of 87 percent (n = 31) and the mean concentration (82.9 μg/kg) was 85 percent less than the mean concentration measured in surface sediments of Green Bay Zone 3B (541 μg/kg) and 78 percent less than the mean concentration measured in surface sediments of Green Bay Zone 3A (376 μg/kg). The mean concentration of interpolated PCBs was 45.7 μg/kg (I_d), which was approximately 91 percent less than those measured in Green Bay Zone 3B and 82 percent less than those measured in Green Bay Zone 3A.

Sediment concentrations of PCB coplanar congeners of concern (congener numbers 77, 81, 105, 118, 126, and 169) are given in Table 6-72. Congener numbers 126 and 169 were not detected (n = 4). Frequency of detection of the other congeners ranged from 50 to 90 percent (n = 4 to 31). RME concentrations of PCB congeners ranged from 0.04 μg/kg (congener 77) to 9.1 μg/kg (congener 118) as compared to RME congener concentrations in Green Bay Zone 3B which ranged from 0.8 μg/kg (congener 81) to 31.0 μg/kg (congener 118) (Table 6-63). These results are similar to congener data for all other reaches examined, where congener 77 or 126 had the minimum RME concentration and congener 118 had the highest concentration.

Fish. COPCs analyzed in whole fish identified as important receptors include mercury, total PCBs, and chlorinated pesticides (dieldrin, p,p'-DDD, p,p'-DDE, p,p'-DDT).

These data are presented in Table 6-73. Fish analyzed in this reach include rainbow smelt, alewife, yellow perch, carp, walleye, and brown trout. Each of these COPCs were detected in at least two fish.

Mercury was detected at a frequency of 100 percent in carp, walleye, and yellow perch at mean concentrations of 0.17 mg/kg (carp), 0.21 mg/kg (walleye), and 0.03 mg/kg (yellow perch). This mean concentration in carp increased 55 percent and the mean concentration in walleye decreased by 16 percent as compared to mean concentrations measured in Green Bay Zone 3B. Yellow perch did not have detected concentrations in Zone 3B and none of these fish had detected mercury concentrations in Green Bay Zone 3A.

Total PCBs were detected at a frequency of 100 percent in all fish species analyzed and the mean concentrations in the order of highest to lowest were: carp (2,992 μg/kg), walleye (2,546 μg/kg), brown trout (2,451 μg/kg), alewife (1,036 μg/kg), rainbow smelt (526 μg/kg), and yellow perch (79.8 μg/kg) (Table 6-71). As compared to mean total PCB concentrations in these species from Green Bay Zone 3A (Table 6-56), the concentration in carp increased by 13 percent, walleye decreased by 39 percent, brown trout decreased by 25 percent, alewife increased by 14 percent, rainbow smelt decreased by 8 percent, and yellow perch decreased by 55 percent. As compared to mean total PCB concentrations in these species from Green Bay Zone 3B (Table 6-64), the concentration in carp decreased by 40 percent, walleye decreased by 60 percent, brown trout increased by 10 percent, alewife decreased by 43 percent, rainbow smelt decreased by 28 percent, and yellow perch decreased by 48 percent.

Dieldrin was detected at a 100 percent detection frequency in all species analyzed (Table 6-73). Mean concentrations in the order of highest to lowest were: brown trout (88.2 μg/kg), walleye (46.9 μg/kg), carp (27.7 μg/kg), alewife (20.8 μg/kg), and rainbow smelt (18.1 μg/kg). As compared to mean dieldrin concentrations measured in Green Bay Zone 3A fish (Table 6-56), brown trout concentrations had increased by 16 percent, walleye concentrations increased by 8 percent, carp concentrations increased by 55 percent, alewife concentrations decreased by 3 percent, and rainbow smelt concentrations increased by 26 percent. As compared to mean dieldrin concentrations measured in Green Bay Zone 3B fish (Table 6-64), brown trout concentrations had increased by 23 percent, walleye concentrations decreased by 6 percent, carp concentrations decreased by 36 percent, alewife concentrations increased by 9 percent, and rainbow smelt concentrations increased by 23 percent.

Chlorinated pesticides p,p'-DDD, p,p'-DDE, and p,p'-DDT were analyzed in carp (n = 10), walleye (n = 20), and yellow perch (n = 5). Overall, measured

concentrations of p,p'-DDD, p,p'-DDE, and p,p'-DDT were significantly higher than in any other location evaluated. While not detected in yellow perch, p,p'-DDD and p,p'-DDT were detected at a frequency of at least 90 percent in carp and walleye with mean p,p'-DDD concentrations of 75.8 μg/kg (carp) and 28.7 μg/kg (walleye), and mean p,p'-DDT concentrations of 8.7 μg/kg (carp) and 33.9 μg/kg (walleye). Neither of these compounds was detected in fish from Green Bay zones 3A or 3B. All species in Zone 4 also had detectable concentrations of p,p'-DDE which was detected at a frequency of 100 percent. Mean p,p'-DDE concentrations in carp, walleye, and yellow perch were 885, 479, and 14.8 μg/kg, respectively. As compared to mean p,p'-DDE concentrations measured in Green Bay Zone 3A fish (Table 6-56), carp concentrations had increased by 35 times and yellow perch concentrations had increased by 2.5 times. As compared to mean p,p'-DDE concentrations measured in Green Bay Zone 3B fish (Table 6-64), carp concentrations had increased by seven times, walleye concentrations had increased by more than two times, and yellow perch concentrations had decreased by 30 percent.

Concentrations of PCB congeners in whole fish are presented in Table 6-74. All congeners analyzed were detected in all species at detection frequencies from 93 to 100 percent. The species of fish analyzed included carp, walleye, alewife, brown trout, and rainbow smelt. Sample sizes ranged from 1 to 18 whole fish. Congener 169 was not analyzed in any fish except for walleye. Congener 126 was not analyzed in any fish except for walleye and brown trout, also walleye and brown trout were the only fish not to have coeluting congeners. In the other fish species, congener 77 coeluted with congener 110, and congener 105 coeluted with congeners 132 and 153. Overall, PCB congener concentrations were highest in carp, followed by walleye, and then brown trout, alewife, and rainbow smelt. Elevated PCB congener levels in carp are likely overstated due to coelution effects.

Overall mean PCB congener concentrations in Green Bay Zone 4 walleye were greater than Green Bay Zone 3A (Table 6-57), but less than Green Bay Zone 3B (Table 6-65). In walleye, individual mean concentrations of PCB congeners 77 and 81 were lower in Green Bay Zone 4 than either of the Zone 3 areas. Concentrations of congeners 105, 118, and 126 were between concentrations measured in walleye from the two Green Bay Zone 3 areas. Of the congeners that did not coelute, detected mean PCB congener concentrations ranged from 0.25 μg/kg (congener 126 in walleye) to 137 μg/kg (congener 118 in walleye). In Green Bay Zone 3A, mean PCB congener concentrations ranged from 0.01 μg/kg (congener 169 in carp) to 125 μg/kg (congener 118 in walleye) and they ranged from 0.05 μg/kg (congener 77 in alewife) to 227 μg/kg (congener 118 in walleye) in Green Bay Zone 3B.

Birds. There are no COPC data for bird receptors in this reach. Because of the lack of data, exposure point concentrations for all piscivorous bird receptors (common terns, Forster's terns, double-crested cormorants, and bald eagles) were estimated using the equation presented in Section 6.2.4 and the input parameter values presented in Table 6-4. The fish species selected for modeling were alewife (total PCBs and dieldrin) and yellow perch (mercury and p,p'-DDE) as a trophic level 3 fish, and walleye as a trophic level 4 fish. Resulting estimated exposure point concentrations are presented in Table 6-75. The mean estimated exposure concentrations for mercury were 14.7 μg/kg-BW/day (common tern), 13.6 μg/kg-BW/day (Forster's tern), 5.7 μg/kg-BW/day (double-crested cormorant), and 20.2 μg/kg-BW/day (bald eagle). The mean estimated exposure concentrations for total PCBs were 508 μg/kg-BW/day (common tern), 468 μg/kg-BW/day (Forster's tern), 196 μg/kg-BW/day (double-crested cormorant), and 329 μg/kg-BW/day (bald eagle). The mean estimated exposure concentrations for dieldrin were 10.2 μg/kg-BW/day (common tern), 9.4 μg/kg-BW/day (Forster's tern), 3.9 μg/kg-BW/day (double-crested cormorant), and 3.6 μg/kg-BW/day (bald eagle). For p,p'-DDE, the mean estimated exposure concentrations were 7.3 μg/kg-BW/day (common tern), 6.7 μg/kg-BW/day (Forster's tern), 2.8 μg/kg-BW/day (double-crested cormorant), and 91.2 μg/kg-BW/day (bald eagle).

Mammals. Exposure point concentrations for mink were estimated using the equation presented in Section 6.2.4 and the input parameter values presented in Table 6-4. Carp was the selected fish species for modeling. Non-interpolated and interpolated sediment total PCB concentrations were each used in the exposure modeling. Resulting estimated exposure point concentrations are presented in Table 6-76. The mean estimated exposure concentration for mercury was 32.8 μg/kg-BW/day. The mean estimated exposure concentration for total PCBs (N) and total PCBs (I_d) were 573 and 573 μg/kg-BW/day, respectively. The mean estimated exposure concentration for dieldrin was 5.3 μg/kg-BW/day. For p,p'-DDE, the mean estimated exposure concentration was 169 μg/kg-BW/day.

6.4.2 Summary of Exposure Concentrations by Media

Summarized below are exposure concentrations by media. This summary has been broken into two parts: all COPCs except PCB congeners, followed by PCB congeners.

Water

Concentrations of surface water analytes were reported in up to three different ways: unfiltered concentrations, filtered concentrations, and particulate concentrations. Only total PCBs were analyzed in surface waters in all river reaches and Green Bay zones, but metals were measured in most areas. Dioxins/furans and pesticides were only analyzed in the Appleton to Little Rapids

Reach and Green Bay Zone 1. Unfiltered pesticides (dieldrin, p,p'-DDD, p,p'-DDE, and p,p'-DDT) were only analyzed in the Appleton to Little Rapids Reach and Green Bay Zone 1. They were not detected in either location. Filtered and particulate p,p'-DDD and p,p'-DDE, and particulate DDT were only analyzed in Green Bay Zone 1 and all samples analyzed had detected concentrations of less than 1 ng/L.

Unfiltered concentrations of metals (arsenic, lead, and mercury) were analyzed more frequently than particulate and filtered concentrations. Only mercury (filtered and unfiltered) was analyzed at all locations. Filtered mercury was not detected in Little Lake Butte des Morts and Green Bay zones 3A, 3B, and 4, while unfiltered mercury was not detected in Green Bay zones 3A and 4. Mean, 95% UCL, and maximum unfiltered mercury concentrations for all areas are presented on Figure 6-5.

Filtered and particulate concentrations of PCBs were detected in all river reaches and Green Bay zones and these concentrations were summed to estimated total water concentrations of total PCBs. Estimated mean, 95% UCL, and maximum total PCB concentrations in water are presented on Figure 6-6. Estimated mean total PCB concentrations were greatest in Green Bay Zone 1 (60.9 μg/L) and represented an increase of 2.2 times over the estimated mean total PCB concentrations in Little Lake Butte des Morts (27.6 μg/L). Estimated mean total PCB concentrations rapidly decreased moving further out into Green Bay: concentrations in Zone 2 (17.8 μg/L) were 3.4 times less than concentrations in Zone 1, and concentrations in Zone 3A (4.4 μg/L) were about 4 times less that concentrations in Zone 2.

Sediment

Arsenic, lead, mercury, total PCBs, dieldrin, p,p'-DDT, p,p'-DDE, and p,p'-DDD were analyzed in all river reaches and Green Bay zones. Arsenic, lead, and total PCBs were detected in all river reaches and Green Bay zones. Dioxins/furans were only analyzed in the Little Lake Butte des Morts Reach and Little Rapids to De Pere Reach and were detected in both locations.

Although arsenic and lead were detected in all river reaches and Green Bay zones, mercury was not detected in Green Bay Zone 3A and was infrequently detected in Green Bay zones 3B and 4 (detection frequency of 25 percent). Mean concentrations of arsenic ranged from 1.5 mg/kg (Green Bay Zone 3A) to 10.1 mg/kg (Zone 1). Mean concentrations of lead ranged from 1.5 mg/kg (Green Bay Zone 3A) to 172 mg/kg (Little Lake Butte des Morts Reach). Mean concentrations of mercury ranged from 0.1 mg/kg (Green Bay Zone 4) to 3.5

mg/kg (Little Rapids to De Pere Reach). Mean, 95% UCL, and maximum concentrations of metals are presented on Figure 6-7.

Total PCBs were detected frequently in all river reaches and Green Bay zones. Measured concentrations are reported in three different ways: non-interpolated, interpolated (I_0), and interpolated (I_d) for all of the river reaches, but, as discussed in Section 6.4.1, I_0 concentrations are not presented for zones 2, 3A, 3B, or 4 of Green Bay. In contrast to metals, PCB concentrations generally decreased moving down the river and into the bay. The mean total PCB concentration ranged from 82.9 μg/kg (Green Bay Zone 4) to 10,724 μg/kg (Little Lake Butte des Morts). Mean, 95% UCL, and maximum concentrations of PCBs are presented on Figure 6-8.

Dieldrin was only detected in one sample from the Little Lake Butte des Morts Reach (n = 15) at a concentration of 5.9 μg/kg. DDE was detected in the Little Rapids to De Pere Reach at a mean concentration of 12.5 μg/kg and one sample in Green Bay Zone 1 at concentration of 1.9 μg/kg. DDD was detected in all river reaches and Green Bay Zone 1, but no other Green Bay zones. The maximum detected mean concentration was 17.8 μg/kg (Little Lake Butte des Morts). DDT was detected in all river reaches, but not in any Green Bay zones (including Zone 1). With the exception of p,p'-DDE and p,p'-DDT in the Little Rapids to De Pere Reach, the calculated mean pesticide concentrations exceeded the maximum concentrations and, therefore, are not reported. Similarly, all calculated 95% UCLs exceeded maximum concentrations. As available, mean and maximum concentrations of pesticides are presented on Figure 6-9. Pesticide concentrations were highest in the Little Lake Butte des Morts Reach and Little Rapids to De Pere Reach.

Fish

Fish data from De Pere to Green Bay (Green Bay Zone 1) was combined with fish data from Green Bay Zone 2 because it was determined that fish in these areas could not be distinguished as independent populations. Mercury was analyzed in all river reaches and Green Bay zones. Arsenic was only analyzed in the Little Lake Butte des Morts Reach. Total PCBs and the pesticides dieldrin, p,p'-DDD, p,p'-DDE, and p,p'-DDT were analyzed in all river reaches and Green Bay zones. Pesticides o,p'-DDD and o,p'-DDT were analyzed in all locations except Green Bay Zone 3B and Green Bay Zone 4 and were not frequently detected. The pesticide o,p'-DDE was not analyzed in the Appleton to Little Rapids Reach, Green Bay Zone 3B, or Green Bay Zone 4, and was also not frequently detected. Dioxin and furan analysis was conducted in Little Lake Butte des Morts, Little Rapids to De Pere, and Green Bay zones 1 and 2. PCB congeners were analyzed in all river reaches and Green Bay zones.

Arsenic was only analyzed in carp in the Little Lake Butte des Morts Reach. Mercury was analyzed in carp and yellow perch in all river reaches and Green Bay zones, as was walleye except for Green Bay Zone 3A. For other fish species, mercury was not consistently analyzed: in Green Bay zones 1 and 2, mercury was analyzed in gizzard shad, alewife, and rainbow smelt; in Green Bay Zone 3A, mercury was analyzed in gizzard shad and rainbow smelt; in Green Bay Zone 3B, mercury was analyzed in gizzard shad and alewife; and in Green Bay Zone 4, mercury was analyzed in alewife. Although not detected in all species at every location, mercury was detected in all reaches and zones. The detected mean concentrations ranged from 0.03 mg/kg (rainbow smelt in Green Bay zones 1 and 2 and 3A, and yellow perch in Green Bay Zone 4) to 0.3 mg/kg (walleye in Green Bay Zone 3B). Mean, 95% UCL, and maximum mercury concentrations in all fish species are presented on Figure 6-10.

Total PCBs were detected frequently in all river reaches and Green Bay zones. The range of detection frequency was 85 to 100 percent. The mean total PCB concentration ranged from 79.8 μg/kg (yellow perch from Green Bay Zone 4) to 6,637 μg/kg (carp from Green Bay zones 1 and 2). Mean, 95% UCL, and maximum total PCB concentrations in yellow perch, carp, and walleye are presented on Figure 6-11. Mean, 95% UCL, and maximum total PCB concentrations in forage fish species (gizzard shad, alewife, shiner species, and rainbow smelt) are presented on Figure 6-12.

Dieldrin was analyzed in all river reaches and Green Bay zones and was frequently detected. Detection frequencies and mean concentrations generally increased from Little Lake Butte des Morts to Green Bay Zone 4. Mean concentrations of dieldrin in fish ranged from non-detect to 88.4 μg/kg (brown trout in Green Bay Zone 4). Mean, 95% UCL, and maximum dieldrin concentrations in fish are presented on Figure 6-13.

Pesticides o,p'-DDD and o,p'-DDT were analyzed in all locations except Green Bay Zone 3B and Green Bay Zone 4, and o,p'-DDE was not analyzed in the Appleton to Little Rapids Reach, Green Bay Zone 3B, or Green Bay Zone 4. Neither o,p'-DDD nor o,p'-DDT was detected in any locations they were analyzed, but o,p'-DDE was detected in at least one species in all areas analyzed, except in Green Bay Zone 3A (analyzed only in rainbow smelt). When detected, the mean concentration of o,p'-DDE ranged from 12.5 μg/kg (walleye in Little Lake Butte des Morts) to 85.0 μg/kg (walleye in Green Bay zones 1 and 2).

The pesticides p,p'-DDD, p,p'-DDE, and p,p'-DDT were analyzed in all river reaches and Green Bay zones. The compound p,p'-DDD was detected in all locations except Green Bay Zone 3A and Green Bay Zone 3B and mean

concentrations ranged from 7.3 μg/kg (alewife in Green Bay zones 1 and 2) to 75.8 μg/kg (carp in Green Bay Zone 4). The compound p,p'-DDT was not detected in any location except Green Bay Zone 4, and here the mean concentrations were non-detect in yellow perch, 8.7 μg/kg in carp, and 33.9 μg/kg in walleye. The compound p,p'-DDE was detected in all species in all locations evaluated except rainbow smelt in Green Bay zones 1 and 2. Mean concentrations ranged from 6.0 μg/kg (yellow perch in Green Bay Zone 3A) to 885 μg/kg (carp in Green Bay Zone 4). Mean, 95% UCL, and maximum p,p'-DDE concentrations in fish are presented on Figure 6-14.

Birds

COPCs analyzed in birds in at least one area included mercury, total PCBs, chlorinated pesticides (dieldrin, o,p'-DDD, o,p'-DDE, o,p'-DDT, p,p'-DDD, p,p'-DDE, p,p'-DDT), dioxins/furans, and PCB congeners. No bird data was collected from the Little Rapids to De Pere Reach or Green Bay Zone 4. Bird analyses was the most limited in terms of number of samples and number of analytes as compared to the other media analyzed. Mercury was analyzed in bald eagle liver from the Appleton to Little Rapids Reach and bald eagle egg from Green Bay Zone 3A. Total PCBs, dieldrin, p,p'-DDD, p,p'-DDE, and p,p'-DDT were analyzed in all locations except Little Rapids to De Pere and Green Bay Zone 4. The pesticides o,p'-DDD, o,p'-DDE, and o,p'-DDT were analyzed in all river reaches and Green Bay zones except for Appleton to Little Rapids Reach, Green Bay Zone 3A, and Green Bay Zone 4. Dioxins and furans were only analyzed in Green Bay Zone 2, while PCB congeners were analyzed in the Little Lake Butte des Morts Reach, Green Bay Zone 2, and Green Bay Zone 3B.

Where they were analyzed, total PCBs were detected at a frequency of 100 percent, except for Green Bay Zone 3B where they were detected at a frequency of 95 percent. The mean total PCB concentration ranged from 2,135 μg/kg (whole tree swallow from Little Lake Butte des Morts) to 11,026 μg/kg (whole double-crested cormorants from Green Bay Zone 2). Measured total PCB concentrations in birds are presented on Figure 6-15. As indicated by this figure, the area where the most bird species were sampled was Green Bay Zone 2. This area also contained the highest concentrations of total PCBs, found in double-crested cormorants.

Dieldrin, p,p'-DDD, p,p'-DDE, and p,p'-DDT were analyzed in all locations in which bird samples were taken. Sample location and type included Little Lake Butte des Morts (tree swallow, whole and egg), Appleton to Little Rapids (bald eagle egg), De Pere to Green Bay (whole tree swallow), Green Bay Zone 2 (double-crested cormorant, brain, egg, and whole; common tern egg; and Forster's tern egg), Green Bay Zone 3A (bald eagle egg), and Green Bay Zone 3B (whole

double-crested cormorant). The same locations were analyzed for the pesticides o,p'-DDD, o,p'-DDE, and o,p'-DDT except for where bald eagle eggs were collected and not analyzed for these pesticides. Mean dieldrin concentrations ranged from non-detect to 224 μg/kg (double-crested cormorant egg from Green Bay Zone 2); these data are presented on Figure 6-16. When detected, the pesticide p,p'-DDD concentrations ranged from 6.1 μg/kg (whole tree swallow from Green Bay Zone 1) to 160 μg/kg (bald eagle egg from Appleton to Little Rapids). Detectable concentrations of p,p'-DDE were measured at a 100 percent frequency in all locations with mean concentrations ranging from 155 (whole tree swallow from Little Lake Butte des Morts) to 4,132 μg/kg (double-crested cormorant egg from Green Bay Zone 2). Measured p,p'-DDE concentrations in birds is presented in Figure 6-17. The pesticide p,p'-DDT was only detected in Green Bay Zone 3B at a mean concentration of 10.9 μg/kg. Detectable concentrations of o,p'-DDD, o,p'-DDE, and o,p'-DDT were not measured in any bird samples.

PCB Congeners

PCB congeners were analyzed in surface water, sediment, whole fish, and bird tissues. PCB congeners 77, 81, 105, 118, 126, and 169 were selected for review based on the fact that these congeners can exert dioxin-like toxicity. All congeners were not analyzed at all sampling points. Congeners 126 and 169 were not analyzed in any water samples. In addition, congeners 81 and 105 were not analyzed in surface water from Little Lake Butte des Morts or Little Rapids to De Pere, and congener 81 was not analyzed in Appleton to Little Rapids.

All congeners were analyzed in all sediment samples, although the mean concentrations were not evaluated for congener 77 in Green Bay Zone 4, congener 81 in Little Lake Butte des Morts and Green Bay Zone 1, or congener 126 in Little Lake Butte des Morts.

PCB congeners 126 and 169 often were not analyzed in fish including rainbow smelt in Green Bay zones 1 and 2, alewife, rainbow smelt, and brown trout in Green Bay Zone 3A, rainbow smelt in Green Bay Zone 3B, and alewife and rainbow smelt in Green Bay Zone 4. Congener 169 was not analyzed in brown trout in Green Bay Zone 4 and PCB congener 126 was not evaluated in Little Lake Butte des Morts.

Congener 81 often was not analyzed in bird tissue, including tree swallow (whole and egg) from Little Lake Butte des Morts, double-crested cormorant (whole and egg) from Green Bay Zone 2, and whole double-crested cormorant from Green Bay Zone 3B.

Mean PCB congener concentration data are presented in two formats in order to illustrate trends in PCB fate and bioaccumulation: the absolute mean concentration and the relative percent of each congener in the sum total of all mean congener concentrations. These data are presented on Figures 6-18 through 6-24. These figures allow comparison of the absolute and relative proportions of PCB congeners between various media for each location. Direct comparisons can not be made between sediment and tissue, however, because sediment congener concentrations are expressed on a dry-weight basis and all other media concentrations are expressed on a wet-weight basis. Also, actual mean surface water concentrations were not included on Figures 6-18 through 6-24 because concentrations were too low to view on the graph scale.

Interpretation of Total PCB Congener Concentrations. Mean PCB congener concentrations showed significant increases from lower to higher trophic levels in tissue samples. For example, bird concentrations generally exceeded fish concentrations, and the highest concentrations of PCB congeners in fish tissue were found in carp, walleye, and brown trout, the highest trophic level fish measured.

Interestingly, PCB congener concentrations generally decreased in sediment through the river reaches and Green Bay zones, while corresponding tissue concentrations generally increased. The highest concentrations measured in sediment were in Little Lake Butte des Morts and the lowest were measured in Green Bay Zone 4. The highest PCB congener concentrations in whole fish tissue were located in Green Bay Zone 4 (carp) and the highest mean bird tissue concentrations were measured in double-crested cormorant eggs from Green Bay Zone 2. It is likely that the sediment congener data collected in the river were skewed to known areas of existing contamination.

Interpretation of Relative PCB Congener Concentrations. PCB congener 118 was detected at higher concentrations than other congeners in most sediment and tissue samples. In fact, congener 118 was measured at greater concentrations than any other congener in all locations except Green Bay Zone 1 or when coelution likely elevated results in other congeners. PCB congener 105 was the next most prevalent congener, followed by congener 77. Congeners 126 and 169 were detected with the lowest frequency and at the lowest concentrations.

Surface Water. Surface water concentrations of PCB congeners were determined by summing the results of particulate and filtered analyses. PCB congener 77 appeared to be predominant in surface water samples, but elevated concentrations are likely due to coelution in all locations with PCB congener 110. This pattern is supported by comparing congener 77 data in all locations, where high

concentrations are only measured when coelution occurred. PCB congener 81 was only analyzed in Green Bay (zones 1, 2, 3A, 3B, and 4). PCB congener 105 was analyzed in the Appleton to Little Rapids Reach and all Green Bay zones, but it coeluted with congeners 132 and 153. Concentrations of congener 105 were extremely low in comparison to other congeners measured in surface water. Relative concentrations of congener 105 in other sample media where coelution did not occur had very similar results, indicating that the coeluting congeners may have comprised a small fraction of the measurement in water. This trend seemed consistent through all types of sample media. The other congeners of interest (congeners 126 and 169) were not analyzed in surface water samples.

Sediment. As with the other media analyzed, the proportionally dominant congener in sediment is congener 118, with the exception of Green Bay Zone 1 where the concentrations of congener 77 was equivalent to the concentration of congener 118 in sediment. In the Little Rapids to De Pere Reach, the sediment concentration of congener 77 was about half of the concentration of congener 118, and in all other areas, the relative contribution of congener 77 was even lower. Another important congener in the sediment is congener 105, which is about equally abundant as congener 77 in Green Bay zones 1 and 2, but ranges from twice as abundant to half as abundant as congener 77 in the other areas.

Fish. The relative PCB congener distribution was similar across all species of fish examined. No significant trends were noted, although the relative concentration of congener 105 measured slightly higher in the Green Bay zones than in the river reaches. High concentrations of congener 81 were measured in golden shiner from Little Lake Butte des Morts, Little Rapids to De Pere, and Green Bay zones 1 and 2. In all three cases, congener 81 coeluted with congeners 87 and 115. The difference in congener concentrations in golden shiner from these locations is most likely due to measurement of the coeluting congeners.

Bird. Relative PCB concentrations were very similar to those measured in fish. PCB congener 81 was present at slightly lower relative concentrations in common tern egg and Forster's tern egg than most fish in Green Bay zones 1 and 2. This value could not be confirmed in other media or locations because congener 81 was not measured in any other bird tissue. No other preferences were noted for bioaccumulation of particular congeners.

6.4.3 Summary of Exposure Concentrations by Area

Summarized below are exposure concentrations by area—river reaches and Green Bay zones.

Little Lake Butte des Morts

COPCs detected in the Little Lake Butte des Morts Reach include: arsenic, lead, mercury, total PCBs, dieldrin, o,p'-DDE, p,p'-DDD, and p,p'-DDE.

Metals were detected in water, sediment, and fish, but not in any of the selected bird receptors because they were not analyzed. Metals concentrations measured in all media in the Little Lake Butte des Morts Reach are presented on Figure 6-25. These concentrations are presented on a log scale because concentrations were found to vary widely, with maximum concentrations as low as 0.12 μg/L for lead in water to 522,000 μg/kg for lead in sediment. Only mercury was detected in water, sediment, and fish (carp and walleye). The mercury water resuspension factor was 2.3×10^{-3}, calculated by dividing the mean water concentration (2.2 μg/L) by the mean sediment concentration (955 μg/kg). The mean carp mercury concentration was 48.0 μg/kg and the single detection of walleye measured 30.0 μg/kg.

Total PCBs were detected in water, sediment, fish, and bird receptors. Total PCB concentrations measured in all media in the Little Lake Butte des Morts Reach are presented on Figure 6-26. Maximum concentrations ranged from 0.059 μg/L in water to 130,000 μg/kg in sediment (non-interpolated concentration). Three sediment calculations showed mean concentrations of 10,724 μg/kg (N), 3,284 μg/kg (I_0), and 3,699 μg/kg (I_d). The mean water PCB concentration was 0.028 μg/L. The water resuspension factor, calculated by dividing the mean water PCB concentration by the mean non-interpolated, I_0, and I_d sediment concentrations were 2.6×10^{-6}, 8.5×10^{-6}, and 7.6×10^{-6}, respectively. The mean fish concentrations ranged from 296 μg/kg in gizzard shad to 1,992 μg/kg in carp. As compared to the mean sediment concentrations (non-interpolated, I_0, and I_d), mean fish concentrations were at least 39 percent lower and up to 97 percent lower. Tree swallow concentrations of total PCBs were compared to sediment concentrations rather than fish because tree swallows consume insects and not fish. Measured mean concentrations in tree swallows were 2,924 μg/kg in eggs and 2,135 μg/kg in whole bodies. These mean concentrations were less than what was observed in the sediment.

Chlorinated pesticides were detected in sediment, fish and bird receptors; they were not analyzed in water. Chlorinated pesticide concentrations measured in all media in the Little Lake Butte des Morts Reach are presented on Figure 6-27. Maximum concentrations ranged from 1.0 μg/kg dieldrin in carp to 530 μg/kg p,p'-DDE in whole tree swallows. While in wildlife, concentrations of p,p'-DDE were more frequently detected and detected at higher concentrations than either isomer of DDT or DDD; p,p'-DDE was not detected in sediment. Sediment contained maximum concentrations of 19 μg/kg p,p'-DDD and 50 μg/kg

p,p'-DDT. Frequencies of detection were low for both compounds; 17 percent for p,p'-DDD and 10 percent for p,p'-DDT. As a result, a mean concentration of p,p'-DDT could not be calculated. The o,p- isomers of DDT and its metabolites were not analyzed in sediment. The mean concentrations of p,p'-DDE in fish were 9.5 μg/kg (yellow perch), 16.9 μg/kg (carp), and 47.6 μg/kg (walleye).

Appleton to Little Rapids

COPCs detected in the Appleton to Little Rapids Reach include: arsenic, lead, mercury, total PCBs, dieldrin, p,p'-DDD, p,p'-DDE, and p,p'-DDT.

Metals were detected in water, sediment, fish, and birds. Metals concentrations measured in all media in the Appleton to Little Rapids Reach are presented on Figure 6-28. These concentrations are presented on a log scale because concentrations were found to vary widely, with maximum concentrations as low as 1.8 μg/L for lead in water to 130,000 μg/kg for lead in sediment. Only mercury was detected in water, sediment, fish (carp and walleye), and bird (bald eagle). Only one bald eagle egg and one liver tissue sample were analyzed. The mercury water resuspension factor was 8.6×10^{-5} based on the mean water concentration (0.066 μg/L) and the mean sediment concentration (766 μg/kg).

Total PCBs were detected in water, sediment, fish, and bird receptors. Total PCB concentrations measured in all media in the Appleton to Little Rapids Reach are presented on Figure 6-29. Maximum concentrations ranged from 0.071 μg/L in water to 74,200 μg/kg in sediment (non-interpolated concentration). Sediment calculations showed mean concentrations of 6,751 μg/kg (N), 175 μg/kg (I_0), and 1,398 μg/kg (I_d). The mean water PCB concentration was 0.017 μg/L. The water resuspension factors based on the mean non-interpolated, I_0, and I_d sediment concentrations were 2.5×10^{-6}, 9.7×10^{-5}, and 1.2×10^{-5}, respectively. The mean fish concentrations were 779 μg/kg (yellow perch), 2,581 μg/kg (carp), and 2,737 μg/kg (walleye).

Chlorinated pesticides were detected in sediment, fish, and bird receptors; they were analyzed in water, but not detected. Chlorinated pesticide concentrations measured in all media in the Appleton to Little Rapids Reach are presented on Figure 6-30. Maximum concentrations ranged from 1.7 μg/kg p,p'-DDD in sediment to 1,100 μg/kg p,p'-DDE in a bald eagle egg. While in wildlife, concentrations of p,p'-DDE were more frequently detected and detected at higher concentrations than either isomer of DDT or DDD, p,p'-DDE was not detected in sediment. Sediment contained maximum concentrations of 1.7 μg/kg p,p'-DDD and 3.4 μg/kg p,p'-DDT. Frequencies of detection were low for both compounds, 20 percent for p,p'-DDD and 10 percent for p,p'-DDT. As a result, mean concentrations of these compounds could not be calculated. The o,p-

isomers of DDT and its metabolites were not analyzed in sediment. The mean concentrations of pesticides in fish were 7.5 μg/kg p,p'-DDD in walleye, 47.8 μg/kg p,p'-DDE in carp, and 57.0 μg/kg p,p'-DDE in walleye. There was a detection of 10 μg/kg p,p'-DDE in the only yellow perch analyzed. While dieldrin was not detected in water, sediment, or fish, it was detected at a concentration of 70 μg/kg in the bald eagle egg analyzed. Also detected in the bald eagle egg were p,p'-DDD (160 μg/kg) and p,p'-DDE (1,100 μg/kg).

Little Rapids to De Pere

COPCs detected in the Little Rapids to De Pere Reach include: arsenic, lead, mercury, total PCBs, dieldrin, o,p'-DDE, p,p'-DDD, p,p'-DDE, and p,p'-DDT. No bird tissues were analyzed in this reach.

Metals were detected in water, sediment, and fish. Metals concentrations measured in all media in the Little Rapids to De Pere Reach are presented on Figure 6-31. These concentrations are presented on a log scale because concentrations were found to vary widely, with maximum concentrations as low as 0.124 μg/L for lead in water to 1,400,000 μg/kg for lead in sediment. Only mercury was detected in water, sediment, and fish (carp and walleye). The water resuspension factor for mercury was 1.1×10^{-3}, based on mean concentrations of 3.9 μg/L in water and 3,496 μg/kg in sediment. The mercury concentrations measured in carp (150 μg/kg) and walleye (160 μg/kg) were based on analysis of one of each fish.

Total PCBs were detected in water, sediment, and fish. Total PCB concentrations measured in all media in the Little Rapids to De Pere Reach are presented on Figure 6-32. Maximum concentrations ranged from 0.124 μg/L in water to 40,430 μg/kg in sediment (non-interpolated concentration). Three sediment calculations showed mean concentrations of 4,782 μg/kg (N), 2,054 μg/kg (I_0), and 2,078 μg/kg (I_d). The mean water concentration was 0.041 μg/L. The water resuspension factor based on mean non-interpolated, I_0, and I_d sediment concentrations were 8.6×10^{-6}, 2.0×10^{-5}, and 2.0×10^{-5}, respectively. The mean fish concentrations ranged from 347 μg/kg in gizzard shad to 3,919 μg/kg in carp.

Chlorinated pesticides were detected in sediment and fish receptors; they were not analyzed in water. Chlorinated pesticide concentrations measured in all media in the Little Rapids to De Pere Reach are presented on Figure 6-33. Maximum concentrations ranged from 2.8 μg/kg p,p'-DDD in sediment to 220 μg/kg p,p'-DDE in walleye. In fish, concentrations of p,p'-DDE were more frequently detected and detected at higher concentrations than either isomer of DDT or DDD, p,p'-DDE, and unlike the other reaches so far, p,p'-DDE was detected in

sediment. The mean concentration of p,p'-DDE was 12.5 μg/kg and the mean concentration of p,p'-DDT was 16.5 μg/kg. The mean concentration of p,p'-DDD could not be calculated. The detection frequency of p,p'-DDT and its metabolites was 25 percent or less in the sediment. The o,p- isomers of DDT and its metabolites were not analyzed in sediment. Dieldrin and o,p'-DDE were only detected in walleye at mean concentrations of 3.4 and 45.7 μg/kg, respectively. Carp was the only fish with detected concentrations of p,p'-DDD and the maximum concentration was 8.0 μg/kg. The mean concentrations of p,p'-DDE in fish were 74.2 μg/kg in carp and 129 μg/kg in walleye. In the only yellow perch analyzed, the concentration of p,p'-DDE was 16 μg/kg.

Green Bay Zone 1

COPCs detected in Green Bay Zone 1 include: arsenic, lead, mercury, total PCBs, p,p'-DDD, p,p'-DDE, and p,p'-DDT. Fish data from this zone were combined with fish data from Zone 2. These data are presented in the discussion of Zone 2 data.

Metals were detected in water and sediment. Metals concentrations measured in all media in Green Bay Zone 1 are presented on Figure 6-34. These concentrations are presented on a log scale because concentrations were found to vary widely, with maximum concentrations as low as 0.041 μg/L for mercury in water to 385,567 μg/kg for arsenic in sediment. Arsenic was only detected in one of four water samples at a concentration of 1.5 μg/L. The arsenic water resuspension factor was calculated to be 1.5×10^{-4} using the mean sediment concentration of 10,080 μg/kg. The mean sediment concentration of lead (75,652 μg/kg) resulted in a water resuspension factor of 4.1×10^{-5}, given a mean water concentration of 3.1 μg/L. The mean mercury concentration in sediment (1,031 μg/kg) resulted in a water resuspension factor of 2.7×10^{-5}, given a mean water concentration of 0.028 μg/L.

Total PCBs were detected in water, sediment, and birds. Total PCB concentrations measured in all media in Green Bay Zone 1 are presented on Figure 6-35. Maximum concentrations ranged from 0.194 μg/L in water to 99,000 μg/kg in sediment (non-interpolated concentration). Total PCB mean concentrations in sediment were 4,184 μg/kg (N), 2,950 μg/kg (I_0), and 2,959 μg/kg (I_d). The mean water concentration was 0.061 μg/L. Water resuspension factors based on mean non-interpolated, I_0, and I_d sediment concentrations were 1.5×10^{-5}, 2.1×10^{-5}, and 2.1×10^{-5}, respectively. The mean concentration in whole tree swallows was 3,118 μg/kg, which, when compared to the mean sediment concentrations, resulted in BSAF values of 0.75 (N) and 1.1 (I_0 and I_d).

Chlorinated pesticides were detected in water, sediment, and bird receptors. Chlorinated pesticide concentrations measured in all media in Green Bay Zone 1 are presented on Figure 6-36. Maximum concentrations ranged from 0.00007 μg/L p,p'-DDD in water to 520 μg/kg p,p'-DDE in whole tree swallows. As with sediment from the Little Rapids to De Pere Reach, p,p'-DDE was detected in sediment, but in only one of 22 samples at a concentration of 1.9 μg/kg. The p,p'-DDE water resuspension factor calculated from the single sediment sample and the mean filtered water concentration (0.000041 μg/L) was 2.2×10^{-5}.

Green Bay Zone 2

COPCs detected in Green Bay Zone 2 include: arsenic, lead, mercury, total PCBs, dieldrin, o,p'-DDE, p,p'-DDD, p,p'-DDE, and p,p'-DDT. Fish data from this zone were combined with fish data from Zone 1.

Metals were detected in water, sediment, and fish. Metals concentrations measured in all media in Green Bay Zone 2 are presented on Figure 6-37. These concentrations are presented on a log scale because concentrations were found to vary widely, with maximum concentrations as low as 0.044 μg/L for lead in water to 42,000 μg/kg for lead in sediment. Only mercury was detected in water, sediment, and fish (alewife, rainbow smelt, carp, and walleye). The mean sediment concentration (491 μg/kg) and the mean water concentration (0.63 μg/L) resulted in a water resuspension factor of 1.3×10^{-3}.

Total PCBs were detected in water, sediment, fish, and birds. Total PCB concentrations measured in all media in Green Bay Zone 2 are presented on Figure 6-38. Maximum concentrations ranged from 0.105 μg/L in water to 74,000 μg/kg in double-crested cormorant eggs. Sediment calculations showed mean concentrations of 251 μg/kg (N) and 1,132 μg/kg (I_d). The mean water PCB concentration was 0.018 μg/kg. Water resuspension factors based on mean non-interpolated and I_d sediment concentrations were 7.1×10^{-5}, and 1.6×10^{-5}, respectively. The mean fish concentrations ranged from 1,049 μg/kg in rainbow smelt to 6,637 μg/kg in carp. In double-crested cormorants, egg concentrations of total PCBs exceeded the concentrations in other tissues (whole bodies or brains). Egg concentrations of total PCBs were also available for common and Forster's terns and whole body concentrations were available for tree swallows.

Chlorinated pesticides were detected in fish and bird receptors; and they were analyzed, but not detected, in sediment. They were not analyzed in water. Dieldrin concentrations measured in all media in Green Bay Zone 2 are presented on Figure 6-39. The mean concentrations of dieldrin in fish were: 7.5 μg/kg (rainbow smelt), 10.5 μg/kg (gizzard shad), 20.8 μg/kg (carp), 21.0 μg/kg (alewife), and 37.3 μg/kg (walleye). In double-crested cormorants, egg concentrations of

dieldrin exceeded the concentrations in other tissues (whole bodies or brains). Egg concentrations of dieldrin were also available for common and Forster's terns. Biomagnification factor (BMF) values calculated from the mean concentration of dieldrin in double-crested cormorant eggs (224 μg/kg) ranged from 10.7 in alewife (21.0 μg/L) to 29.9 in rainbow smelt (7.5 μg/L). BMF values calculated from the mean concentration in common tern eggs (85.0 μg/kg) ranged from 4.0 in alewife to 11.3 in rainbow smelt. BMF values calculated from the mean concentration in Forster's tern eggs (47.6 μg/kg) ranged from 2.3 in alewife to 6.3 in rainbow smelt.

DDT and metabolite concentrations measured in all media in Green Bay Zone 2 are presented on Figure 6-40. Maximum concentrations ranged from 2.7 μg/kg p,p'-DDD in Forster's tern egg to 11,000 μg/kg p,p'-DDE in whole bodies and eggs of double-crested cormorants. In fish, concentrations of p,p'-DDE were more frequently detected and detected at higher concentrations than either isomer of DDT or DDD. The mean concentrations of p,p'-DDE in fish were: 64.2 μg/kg (gizzard shad), 104 μg/kg (alewife), 32.9 μg/kg (yellow perch), 197 μg/kg (carp), and 353 μg/kg (walleye). In birds, concentrations of p,p'-DDE were more frequently detected and detected at higher concentrations than p,p'-DDT or p,p'-DDD. In double-crested cormorants, egg concentrations of p,p'-DDE exceeded the concentrations in other tissues (whole bodies or brains). Egg concentrations of p,p'-DDE were also available for common and Forster's terns. BMF values calculated from the mean concentration of p,p'-DDE in double-crested cormorant eggs (4,132 μg/kg) were 39.7 in alewife (104 μg/kg) and 64.4 in gizzard shad (64.2 μg/kg). BMF values calculated from the mean concentration in common tern eggs (666 μg/kg) were 6.4 in alewife and 10.4 in gizzard shad. BMF values calculated from the mean concentration in Forster's tern eggs (447 μg/kg) were 4.3 in alewife and 7.0 in gizzard shad.

Green Bay Zone 3A

COPCs detected in Green Bay Zone 3A include: arsenic, lead, mercury, total PCBs, dieldrin, p,p'-DDD, and p,p'-DDE.

Metals were detected in sediment, fish, and birds. Mercury was analyzed in water and sediment, but not detected. Metals concentrations measured in all media in Green Bay Zone 3A are presented on Figure 6-41. The maximum concentrations of metals ranged from a low of 50.0 μg/kg mercury in fish to a high of 1,900 μg/kg for lead in sediment. Mercury was the only metal analyzed in fish and birds. Of the fish analyzed, mercury was only detected in rainbow smelt at a mean concentration of 26.3 μg/kg. The only bird analyzed for mercury was bald eagle (n = 3). Mercury was detected at a frequency of 100 percent in eggs at a mean

concentration of 273 μg/kg. The BMF calculated from the mean concentration in bald eagle eggs and rainbow smelt was 10.4.

Total PCBs were detected in water, sediment, fish, and bird receptors. Total PCB concentrations measured in all media in Green Bay Zone 3A are presented on Figure 6-42. Maximum concentrations ranged from 0.022 μg/L in water to 13,000 μg/kg in the single bald eagle egg analyzed. Sediment calculations showed concentrations of 376 μg/kg (N) and 256 μg/kg (I_d). The mean water PCB concentration was 0.004 μg/L. Water resuspension factors based on non-interpolated and I_d sediment concentrations were 1.1×10^{-5} and 1.6×10^{-6}, respectively. The mean fish concentrations were 179 μg/kg (yellow perch), 570 μg/kg (rainbow smelt), 907 μg/kg (alewife), 2,642 μg/kg (carp), 3,250 μg/kg (brown trout), and 4,155 μg/kg (walleye). The total PCB concentration measured in the only gizzard shad analyzed was 3,524 μg/kg.

Chlorinated pesticides were detected in fish and bird receptors; they were not analyzed in water, and they were analyzed in sediment, but not detected. Chlorinated pesticide concentrations measured in all media in Green Bay Zone 3A are presented on Figure 6-43. Maximum concentrations ranged from 9.0 μg/kg p,p'-DDE in yellow perch to 2,400 μg/kg p,p'-DDE in the single bald eagle egg analyzed. The mean concentrations of dieldrin in fish were 14.4 μg/kg (rainbow smelt), 21.5 μg/kg (alewife), 17.9 μg/kg (carp), 43.4 μg/kg (walleye), and 76.0 μg/kg (brown trout). The concentration of dieldrin measured in the single bald eagle egg analyzed was 200 μg/kg, resulting in a BMF of 11.2 when compared with carp. The mean concentrations of p,p'-DDE in fish were 6.0 μg/kg in yellow perch and 30 μg/kg in rainbow smelt. Concentrations measured in the single gizzard shad and carp were 150 and 25 μg/kg, respectively. Both p,p'-DDD (120 μg/kg) and p,p'-DDE (2,400 μg/kg) were detected in the single bald eagle egg analyzed. The p,p'-DDE BMF in the bald eagle egg was 96 when compared to the mean p,p'-DDE concentration measured in carp.

Green Bay Zone 3B

COPCs detected in Green Bay Zone 3B include: arsenic, lead, mercury, total PCBs, dieldrin, p,p'-DDD, p,p'-DDE, and p,p'-DDT.

Metals were detected in water, sediment, and fish; they were not analyzed in birds. Metals concentrations measured in all media in Green Bay Zone 3B are presented on Figure 6-44. These concentrations are presented on a log scale because concentrations were found to vary widely, with maximum concentrations as low as 0.090 μg/L for mercury in water to 50,000 μg/kg for lead in sediment. Mercury was the only metal analyzed in fish, and it was only detected in one fish each of carp and walleye. The water resuspension factor calculated from the mean

sediment concentration (109 μg/kg) and the mean water concentration (0.047 μg/L) was 4.3×10^{-4}.

Total PCBs were detected in water, sediment, fish, and bird receptors. Total PCB concentrations measured in all media in Green Bay Zone 3B are presented on Figure 6-45. Maximum concentrations ranged from 0.013 μg/L in water to 20,031 μg/kg in walleye. Sediment calculations showed mean concentrations of 542 μg/kg (N) and 482 μg/kg (I_d). The mean water PCB concentration was 0.004 μg/L. Water resuspension factors based on mean non-interpolated, I_0, and I_d sediment concentrations were 7.4×10^{-6}, 8.4×10^{-6}, and 8.3×10^{-6}, respectively. The mean fish concentrations were 154 μg/kg (yellow perch), 733 μg/kg (rainbow smelt), 1,821 μg/kg (alewife), 4,947 μg/kg (carp), 2,223 μg/kg (brown trout), and 6,429 μg/kg (walleye). The total PCB concentration measured in the only gizzard shad analyzed was 635 μg/kg.

Chlorinated pesticides were detected in fish and bird receptors; and they were analyzed in sediment, but not detected. They were not analyzed in water. Chlorinated pesticide concentrations measured in all media in Green Bay Zone 3B are presented on Figure 6-46. Maximum concentrations ranged from 23.0 μg/kg p,p'-DDE in yellow perch to 6,500 μg/kg p,p'-DDE in whole double-crested cormorants. The mean concentrations of dieldrin in fish were 14.7 μg/kg (rainbow smelt), 19.1 μg/kg (alewife), 43.2 μg/kg (carp), 50.1 μg/kg (walleye), and 72.0 μg/kg (brown trout). The mean concentration of dieldrin measured in whole double-crested cormorants was 128 μg/kg resulting in BMF values of 6.7 compared to alewife and 8.7 compared with rainbow smelt. The mean concentrations of p,p'-DDE in fish were 21.0 μg/kg (yellow perch), 126 μg/kg (carp), and 207 μg/kg (walleye). Concentrations measured in the single alewife and gizzard shad analyzed were 80.0 and 37.0 μg/kg, respectively. Whole double-crested cormorants had mean concentrations of 6.3 μg/kg p,p'-DDD, 10.9 μg/kg p,p'-DDT, and 2,010 μg/kg p,p'-DDE. The mean p,p'-DDE concentration in the whole double-crested cormorants demonstrated BMF values of 25.1 compared to alewife and 54.3 compared to gizzard shad.

Green Bay Zone 4

COPCs detected in Green Bay Zone 4 include: arsenic, lead, mercury, total PCBs, dieldrin, p,p'-DDD, p,p'-DDE, and p,p'-DDT. No bird tissues were analyzed in this zone.

Metals were detected in sediment and fish; they were analyzed in water, but not detected. Metals concentrations measured in all media in Green Bay Zone 4 are presented on Figure 6-47. The maximum concentrations ranged from a low of 40.0 μg/kg for mercury in yellow perch to 8,900 μg/kg for arsenic in sediment.

Mercury was the only metal analyzed in fish, where mean concentrations measured were: 26.0 μg/kg (yellow perch), 168 μg/kg (carp), and 208 μg/kg (walleye).

Total PCBs were detected in water, sediment, and fish. Total PCB concentrations measured in all media in Green Bay Zone 4 are presented on Figure 6-48. Maximum concentrations ranged from 0.004 μg/L in water to 9,620 μg/kg in walleye. Sediment calculations showed mean concentrations of 82.9 μg/kg (N) and 45.7 μg/kg (I_d). The mean water PCB concentration was 0.001 μg/kg. Water resuspension factors based on mean non-interpolated and I_d sediment concentrations were 1.2×10^{-5} and 2.2×10^{-5}, respectively. The mean fish concentrations were 79.8 μg/kg (yellow perch), 526 μg/kg (rainbow smelt), 1,036 μg/kg (alewife), 2,992 μg/kg (carp), 2,451 μg/kg (brown trout), and 2,546 μg/kg (walleye).

Chlorinated pesticides were detected in fish; they were not analyzed in water, and they were analyzed in sediment, but not detected. Chlorinated pesticide concentrations measured in all media in Green Bay Zone 4 are presented on Figure 6-49. Maximum concentrations ranged from 15.0 μg/kg p,p'-DDT in carp to 1,749 μg/kg p,p'-DDE in carp. The mean concentrations of dieldrin in fish were 18.1 μg/kg (rainbow smelt), 20.8 μg/kg (alewife), 27.7 μg/kg (carp), 46.9 μg/kg (walleye), and 88.2 μg/kg (brown trout). The mean concentrations of p,p'-DDE in fish exceeded the mean concentrations of p,p'-DDD and p,p'-DDT in fish and were 14.8 μg/kg (yellow perch), 885 μg/kg (carp), and 479 μg/kg (walleye).

6.5 Risk Characterization

Risk characterization for each assessment endpoint will be based upon the calculated HQs and, as available, population or field study data will be included for comparison. Hazard quotients alone, calculated based on literature values, do not provide enough information for characterizing ecological effects and, therefore, field studies on populations are a recommended supplement to the risk evaluation, particularly when the contamination has a historical basis (EPA, 1994b, 1997a).

Each of the eight assessment endpoints selected for risk evaluation focused on the community function, or survival and reproduction of groups of multiple species which have similar trophic status and potential for exposure. For the derivation of TRVs, however, sublethal endpoints such as fry growth (total PCBs), bird deformity (total PCBs and TCDD-Eq), and abnormal behavior in fish (dieldrin) were evaluated because these were the most sensitive TRVs found, or the only TRVs available. With this in mind, not all calculated HQs were equally

determinative of risk to the assessment endpoints. Therefore, risk conclusions based on specific HQs were weighted with available population data, habitat quality, and field study data for the measurement endpoint species examined.

Use of population, community, or biological field surveys can be an important component of assessing potential toxic effects of chemicals in an ecological risk assessment (EPA, 1994b, 1997a; Menzie *et al.*, 1996). Field survey data of affected receptor populations have been used as part of a weight-of-evidence approach in the Clinch River ecological risk assessment (Sample and Suter, 1999; Suter *et al.*, 1999). For fish, invertebrate, piscivorous, and insectivorous wildlife, one of the assessment endpoints was reduction in abundance, with population data used as an endpoint property to determine if there were significant reductions in the affected areas (Cook *et al.*, 1999). Population level data have also been incorporated to weight-of-evidence assessments in some state programs. The exclusion of field population data in a weight-of-evidence approach was an important critique of the 1998 Draft BLRA by the peer review conducted by the Association for the Environmental Health of Soils (AEHS, 2000), and in the peer review of the of the Hudson River PCB Superfund Site (ERG, 2000).

While HQs and these other kinds of data can not be quantitatively combined, each can inform risk managers on the presence of risk and how these risks may be reduced. Therefore, this weighting process did not result in the distillation of a single conclusive statement regarding overall risk to each assessment endpoint. Rather, the interpretation of overall risk and resulting risk management decisions will be determined by the risk managers. Consideration of the magnitude of uncertainty, discussed in Section 6.6, is also a key component of the risk interpretation process.

The evaluation in the sections below begins with the derivation of HQs by comparing the environmental exposure points that were derived from data in the FRDB and presented in Section 6.4, to the effects criteria developed in Section 6.3. HQs that were calculated based on exposures derived from food web modeling will be termed "estimated" HQs. Where available, estimated HQs will be compared to "measured" HQs—HQs that were derived from tissue residue data. If both the NOAEC and LOAEC HQs are less than 1.0, then it will be assumed that there is no risk. If the NOAEC HQ exceeds 1.0, but the LOAEC HQ is less than 1.0, then it will be assumed that potentially there is risk. If both the LOAEC HQ and the NOAEC HQ exceed 1.0, then it will be assumed that there is risk. HQs will be calculated for both the mean and RME exposure points, but the evaluation of risk will be based principally on the RME.

Secondly, the available population and habitat information presented in Section 2 will be evaluated for relevance to risk by receptor and reach or zone.

Thirdly, field studies that have been conducted on the receptor species within the Lower Fox River and Green Bay will be examined. Some of these studies were cited in the development of the TRVs presented in Section 6.3, but the actual effect will be re-examined here.

Risks will be evaluated based upon the preponderance of evidence individually for each river reach and Green Bay zone, and for the individual assessment endpoints. The risk questions posed in Section 6.2 will be evaluated and summarized at the end of this section.

6.5.1 Estimation of Exposure Point Risks

Little Lake Butte des Morts

Water. COPC data available for risk evaluation in Little Lake Butte des Morts Reach included filtered and unfiltered lead, filtered and unfiltered mercury, and filtered, unfiltered, and particulate total PCBs. Hazard quotients for surface water COPCs detected in Little Lake Butte des Morts Reach are given in Table 6-8. The only HQs that exceeded 1.0 were the mean and RME HQs for unfiltered mercury (5.1 and 16, respectively) and the RME NOAEC HQ (1.1) for estimated total PCBs.

Sediment. COPC data available for risk evaluation in Little Lake Butte des Morts Reach included arsenic, lead, mercury, 2,3,7,8-TCDD, total PCBs, dieldrin, p,p'-DDD, p,p'-DDE, and p,p'-DDT. Hazard quotients for surface sediment COPCs detected in Little Lake Butte des Morts Reach are given in Table 6-10. HQs that exceeded 1.0 included the mean and RME HQs for lead (5.0 and 15, respectively), mercury (5.6 and 8.5, respectively), total PCBs (104 to 723), p,p'-DDD (5.0 and 5.4, respectively), and the RME HQs for 2,3,7,8-TCDD (1.1) and p,p'-DDT (7.1).

Fish. COPC data available for risk evaluation in Little Lake Butte des Morts Reach included arsenic, mercury, total PCBs, dieldrin, o,p'-DDD, o,p'-DDE, o,p'-DDT, p,p'-DDD, p,p'-DDE, and p,p'-DDT. Hazard quotients for fish COPCs detected in Little Lake Butte des Morts Reach are given in Tables 6-77 and 6-78. The only HQs that exceeded 1.0 are those for total PCBs. Specifically, those that exceeded 1.0 included the NOAEC mean and RME HQs for golden shiner (1.3 and 1.5, respectively), carp (2.6 and 3.9, respectively), and walleye (1.5 and 5.0, respectively).

Hazard quotients for PCB and dioxin/furan congeners in fish from the Little Lake Butte des Morts Reach are contained in Table 6-78. None of the estimated total TEQ HQs exceeded 1.0.

Birds. COPC data available for risk evaluation in Little Lake Butte des Morts Reach included total PCBs, dieldrin, o,p'-DDD, o,p'-DDE, o,p'-DDT, p,p'-DDD, p,p'-DDE, and p,p'-DDT. Hazard quotients for bird COPCs detected in Little Lake Butte des Morts Reach are given in Table 6-79. The only HQs that exceeded 1.0 are those for total PCBs. Specifically, those that exceeded 1.0 included the reproduction NOAEC RME HQ for whole body tree swallows (1.1) and the deformity mean and RME NOAEC HQs for tree swallow eggs (3.7 and 4.7, respectively) and whole body tree swallows (2.7 and 6.6, respectively).

HQs were estimated for all piscivorous birds and the COPCs mercury, total PCBs, dieldrin, and p,p'-DDE based on dietary intake (Table 6-80). Estimated HQs that exceeded 1.0 were the mercury mean and RME NOAEC HQs for common tern (1.6 and 1.6, respectively) and Forster's tern (1.4 and 1.5, respectively), and the total PCB mean and RME NOAEC HQs for common tern (1.3 and 2.3, respectively), Forster's tern (1.2 and 2.1, respectively), and bald eagle (1.3 and 3.1, respectively).

Hazard quotients for PCB congeners in tree swallows from the Little Lake Butte des Morts Reach are contained in Table 6-81. Total TEQ mean and RME NOAEC HQs exceeded 1.0 for tree swallow eggs using both the Tillitt *et al.* (1991b) TEFs where HQs were 1.1 and 2.3, respectively, and the Van den Berg *et al.* (1998) TEFs where HQs were 6.9 and 13, respectively. In whole body tree swallows, total TEQ mean and RME NOAEC HQs only exceeded 1.0 when the Van den Berg *et al.* (1998) TEFs were used, and the corresponding mean and RME HQs were 2.1 and 3.7, respectively.

Mink. COPC data are not available for risk evaluation in the Little Lake Butte des Morts Reach. Therefore, the only piscivorous mammal HQs that were evaluated were those that were estimated, based on dietary intake, for the COPCs mercury, total PCBs, dieldrin, and p,p'-DDE (Table 6-82). The only estimated HQs that exceeded 1.0 were those for total PCBs. Sediment was assumed to be a small fraction of the diet, therefore, three HQs were developed for total PCBs based on the sediment concentrations used (N, I_0, and I_d). Total PCB NOAEC HQs ranged from 99 to 170 and total PCB LOAEC HQs ranged from 3.1 to 5.2.

Appleton to Little Rapids Reach

Water. COPC data available for risk evaluation in the Appleton to Little Rapids Reach included unfiltered arsenic; unfiltered lead; filtered and unfiltered mercury;

unfiltered 2,3,7,8-TCDF; filtered, unfiltered, and particulate total PCBs; unfiltered dieldrin; unfiltered p,p'-DDD; unfiltered p,p'-DDE; and unfiltered p,p'-DDT. Hazard quotients for surface water COPCs detected in the Appleton to Little Rapids Reach are given in Table 6-20. The only HQ that exceeded 1.0 was the RME NOAEC HQ (1.2) for estimated total PCBs.

Sediment. COPC data available for risk evaluation in the Appleton to Little Rapids Reach included arsenic, lead, mercury, total PCBs, dieldrin, p,p'-DDD, p,p'-DDE, and p,p'-DDT. Hazard quotients for surface sediment COPCs detected in the Appleton to Little Rapids Reach are given in Table 6-21. HQs that exceeded 1.0 included the mean and RME HQs for lead (2.2 and 2.6, respectively), mercury (4.5 and 10, respectively), and total PCBs (5.5 to 483).

Fish. COPC data available for risk evaluation in the Appleton to Little Rapids Reach included mercury, total PCBs, PCB congeners, dieldrin, o,p'-DDD, o,p'-DDT, p,p'-DDD, p,p'-DDE, and p,p'-DDT. Hazard quotients for fish COPCs detected in the Appleton to Little Rapids Reach are given in Tables 6-83 and 6-84. The only HQs that exceeded 1.0 are those for total PCBs. Specifically, those that exceeded 1.0 included the NOAEC RME HQ for yellow perch (1.6), and the NOAEC mean and RME HQs for carp (3.4 and 4.7, respectively), and walleye (3.6 and 5.1, respectively).

Birds. For birds in this reach, hazard quotients calculated based on data within the FRDB for eagles, and estimated for piscivorous birds by exposure modeling. As discussed previously, risks to insectivorous birds were not estimated for this reach. Hazard quotients for bird COPCs detected in the Appleton to Little Rapids Reach are given in Table 6-85. HQs that exceeded 1.0 included the mercury reproduction RME NOAEC HQ of 7.0 for bald eagle, the total PCB NOAEC and LOAEC RME HQs for bald eagles (7.7 and 4.7, respectively), and the deformity NOAEC and LOAEC RME HQs for bald eagles (45 and 4.5, respectively).

HQs were estimated for all piscivorous birds and the COPCs mercury, total PCBs, dieldrin, and p,p'-DDE based on dietary intake (Table 6-86). Estimated HQs that exceeded 1.0 were the mercury mean and RME NOAEC HQs for common tern (1.5 and 1.5, respectively), Forster's tern (1.4 and 1.4, respectively), and bald eagle (1.1 and 1.8, respectively), and the total PCB mean and RME NOAEC HQs for common tern (3.4 and 5.3, respectively), Forster's tern (3.1 and 4.9, respectively), double-crested cormorant (1.3 and 2.1, respectively), and bald eagle (2.6 and 3.6, respectively).

Mink. COPC data are not available for risk evaluation in the Appleton to Little Rapids Reach. Therefore, the only piscivorous mammal HQs that were evaluated were

those that were estimated, based on dietary intake, for the COPCs mercury, total PCBs, dieldrin, and p,p'-DDE (Table 6-87). The only estimated HQs that exceeded 1.0 were those for total PCBs. Sediment was assumed to be a small fraction of the diet; therefore, three HQs were developed for total PCBs based on the sediment concentrations used (N, I_0, and I_d). Total PCB NOAEC HQs ranged from 124 to 192 and total PCB LOAEC HQs ranged from 3.8 to 5.9.

Little Rapids to De Pere Reach

Water. COPC data available for risk evaluation in the Little Rapids to De Pere Reach included filtered and unfiltered lead, filtered and unfiltered mercury, and filtered and particulate total PCBs. Hazard quotients for surface water COPCs detected in the Little Rapids to De Pere Reach are given in Table 6-28. The only HQs that exceeded 1.0 were the mean and RME HQs for filtered mercury (2.9 and 5.7, respectively) and unfiltered mercury (8.8 to 16, respectively).

Sediment. COPC data available for risk evaluation in the Little Rapids to De Pere Reach included arsenic, lead, mercury, 2,3,7,8-TCDD, total PCBs, dieldrin, p,p'-DDD, p,p'-DDE, and p,p'-DDT. Hazard quotients for surface sediment COPCs detected in the Little Rapids to De Pere Reach are given in Table 6-29. HQs that exceeded 1.0 included the mean and RME HQs for lead (4.6 and 8.0, respectively), mercury (21 and 24, respectively), 2,3,7,8-TCDD (1.3 and 1.7, respectively) total PCBs (65 to 334), p,p'-DDE (8.8 and 15, respectively), and p,p'-DDT (2.4 and 2.9, respectively).

Fish. COPC data available for risk evaluation in the Little Rapids to De Pere Reach included mercury, total PCBs, PCB congeners, dieldrin, o,p'-DDD, o,p'-DDE, o,p'-DDT, p,p'-DDD, p,p'-DDE, and p,p'-DDT. Hazard quotients for fish COPCs detected in the Little Rapids to De Pere Reach are given in Tables 6-88 and 6-89. HQs that exceeded 1.0 included the mercury NOAEC RMEs for carp (6.0) and walleye (6.4), and the total PCB NOAEC mean and RME HQs for golden shiner (1.3 and 1.4, respectively), carp (5.2 and 7.6, respectively), and walleye (4.2 and 6.0, respectively).

Birds. COPC data are not available for risk evaluation in the Little Rapids to De Pere Reach. Therefore, the only HQs that were evaluated were those that were estimated, based on dietary intake, for all piscivorous birds and the COPCs mercury, total PCBs, dieldrin, and p,p'-DDE (Table 6-90). Estimated HQs that exceeded 1.0 were the mercury RME NOAEC HQs for double-crested cormorant (1.2), the mercury mean and RME NOAEC HQs for common tern (1.6 and 3.2, respectively), Forster's tern (1.5 and 2.9, respectively), and bald eagle (2.1 and 2.1, respectively), and the total PCB mean and RME NOAEC HQs for common

tern (1.5 and 1.6, respectively), Forster's tern (1.4 and 1.5, respectively), and bald eagle (3.7 and 5.5, respectively).

Mink. COPC data are not available for risk evaluation in the Little Rapids to De Pere Reach. Therefore, the only piscivorous mammal HQs that were evaluated were those that were estimated, based on dietary intake, for the COPCs mercury, total PCBs, dieldrin, and p,p'-DDE (Table 6-91). The only estimated HQs that exceeded 1.0 were those for total PCBs. Sediment was assumed to be a small fraction of the diet; therefore, three HQs were developed for total PCBs based on the sediment concentrations used (N, I_0, and I_d). Total PCB NOAEC HQs ranged from 190 to 291 and total PCB LOAEC HQs ranged from 5.8 to 8.9.

De Pere to Green Bay Reach (Green Bay Zone 1)

Water. Of the COPCs measured in surface waters of Zone 1, only PCB concentrations exceeded the TRV. Hazard quotients for surface water COPCs detected in Green Bay Zone 1 are given in Table 6-35. Based upon the calculated RME NOAEC, the HQ for particulate total PCBs just exceeded 1.0 (1.1). The mean and RME NOAEC HQs for estimated total PCBs were 1.2 and 1.4, respectively.

Sediment. Risks to benthic organisms within this reach are indicated by the hazard quotients for lead, mercury, PCBs, arsenic, and both DDD and DDE. Hazard quotients for surface sediment COPCs detected in Green Bay Zone 1 are given in Table 6-36. HQs that exceeded 1.0 included the mean and RME HQs for lead (2.2 and 2.7, respectively), mercury (6.1 and 8.1, respectively), total PCBs (93 to 174), and the RME HQs for arsenic (1.4), p,p'-DDD (1.3), and p,p'-DDE (1.3).

Fish. Risks to Zone 1 fish are reported under the Green Bay Zone 2 summary. As previously discussed, these fish represent a single population, and thus the data were combined for risk evaluation.

Birds. Tissue residue-based hazard quotients for insectivorous birds for Green Bay Zone 1 are given in Table 6-92. The only HQs that exceeded 1.0 are those for deformity from total PCBs in whole tree swallows, where the mean and RME NOAEC HQs were 3.9 and 5.6, respectively.

Estimated hazard quotients for piscivorous birds in this reach did not differ from HQs calculated for piscivorous birds in Green Bay Zone 2, and thus are discussed in the next section.

Mink. COPC data are not available for risk evaluation in Green Bay Zone 1. Therefore, the only piscivorous mammal HQs that were evaluated were those that were estimated, based on dietary intake, for the COPCs mercury, total PCBs, dieldrin,

and p,p'-DDE (Table 6-93). The only estimated HQs that exceeded 1.0 were those for total PCBs. Sediment was assumed to be a small fraction of the diet, therefore, three HQs were developed for total PCBs based on the sediment concentrations used (N, I_0, and I_d). Total PCB NOAEC HQs ranged from 321 to 359 and total PCB LOAEC HQs ranged from 9.9 to 11.

Green Bay Zone 2

Water. COPC data available for risk evaluation in Green Bay Zone 2 included filtered and unfiltered lead, filtered and unfiltered mercury, and filtered and particulate total PCBs. Hazard quotients for surface water COPCs detected in Green Bay Zone 2 are given in Table 6-41. The only HQs that exceeded 1.0 were the RME HQs for filtered and unfiltered mercury (5.2 and 11, respectively) and the mean HQ (1.4) for unfiltered mercury.

Sediment. COPC data available for risk evaluation in Green Bay Zone 2 included arsenic, lead, mercury, total PCBs, dieldrin, p,p'-DDD, p,p'-DDE, and p,p'-DDT. Hazard quotients for surface sediment COPCs detected in Green Bay Zone 2 are given in Table 6-42. HQs that exceeded 1.0 included the mean and RME HQs for mercury (2.9 and 8.8, respectively) and total PCBs (7.9 to 37).

Fish. COPC data available for risk evaluation in Green Bay zones 1 and 2 included mercury, total PCBs, PCB congeners, dieldrin, o,p'-DDD, o,p'-DDE, o,p'-DDT, p,p'-DDD, p,p'-DDE, and p,p'-DDT. Hazard quotients for fish COPCs detected in Green Bay zones 1 and 2 are given in Tables 6-94 and 6-95. HQs that exceeded 1.0 included the mercury NOAEC RME HQ for walleye (1.1) and the total PCB NOAEC mean and RME HQs for all fish analyzed, including alewife (3.4 and 4.2, respectively), gizzard shad (2.4 and 2.6, respectively), rainbow smelt (1.4 and 1.5, respectively), common shiner (4.6 and 5.1, respectively), emerald shiner (4.6 and 5.1, respectively), golden shiner (1.8 and 1.9, respectively), yellow perch (1.6 and 2.1, respectively), carp (8.7 and 9.7, respectively), and walleye (8.6 and 10, respectively).

Hazard quotients for PCB and dioxin/furan congeners in fish from Green Bay zones 1 and 2 are contained in Table 6-95. The only total TEQ HQ that exceeded 1.0 was the NOAEC RME HQ for walleye (1.7).

Birds. COPC data available for risk evaluation in Green Bay Zone 2 included total PCBs, dieldrin, o,p'-DDD, o,p'-DDE, o,p'-DDT, p,p'-DDD, p,p'-DDE, and p,p'-DDT. Hazard quotients for bird COPCs detected in Green Bay Zone 2 are given in Table 6-96. Total PCB NOAEC and LOAEC HQs exceeded 1.0 for both double-crested cormorant eggs and whole bodies and for both endpoints—reproduction and deformity. HQs for reproduction ranged from 1.5

to 4.5 and deformity HQs ranged from 1.4 to 26. Total PCB RME NOAEC HQs exceed 1.0 for both common tern and Forster's tern eggs and for both endpoints. The RME NOAEC HQs for reproduction were 1.3 for both species, and the mean and RME NOAEC deformity HQs ranged from 6.0 to 7.8. For tree swallows, the only total PCB HQs that exceeded 1.0 were the mean and RME deformity HQs of 3.7 and 4.4, respectively.

Other HQs that exceeded 1.0 included the NOAEC RME HQ for dieldrin in common tern eggs (1.4), the mean and RME NOAEC HQs for dieldrin in double-crested cormorant eggs (2.2 and 4.4, respectively), the mean and RME NOAEC HQs for dieldrin in double-crested cormorant whole bodies (2.0 and 2.4, respectively), the mean and RME NOAEC HQs and the RME LOAEC HQ for p,p'-DDE in double-crested cormorant eggs (1.4, 2.4, and 1.4, respectively), and the RME NOAEC HQ for p,p'-DDE in whole double-crested cormorants (1.2).

HQs were estimated for all piscivorous birds and the COPCs mercury, total PCBs, dieldrin, and p,p'-DDE based on dietary intake (Table 6-97). Estimated HQs that exceeded 1.0 were the mercury mean and RME NOAEC HQs for common tern (6.1 and 15, respectively), Forster's tern (5.7 and 14, respectively), double-crested cormorant (2.4 and 5.9, respectively), and bald eagle (1.3 and 1.6, respectively); the total PCB mean and RME NOAEC HQs for common tern (11 and 14, respectively), Forster's tern (10 and 13, respectively), double-crested cormorant (4.4 and 5.4, respectively), and bald eagle (6.6 and 7.4, respectively); and the p,p'-DDE mean and RME NOAEC HQs for common tern (2.8 and 3.9, respectively), Forster's tern (2.6 and 3.6, respectively), double-crested cormorant (1.1 and 1.5, respectively), and bald eagle (1.4 and 4.0, respectively). For common tern and Forster's tern, RME LOAEC HQs (1.6 and 1.5, respectively) exceeded 1.0. Total PCB LOAEC HQs also exceeded 1.0 for both of these species. For common tern, the mean total PCB HQ is 1.1 and the RME HQ is 1.4, and for Forster's tern the RME HQ is 1.3.

Hazard quotients for PCB and dioxin/furan congeners in birds from the Green Bay Zone 2 are contained in Table 6-98 (whole tree swallows), Table 6-99 (double-crested cormorant eggs and whole bodies), Table 6-100 (common tern eggs), and Table 6-101 (Forster's tern eggs).

In whole body tree swallows, total TEQ mean and RME HQs only exceeded 1.0 when the Van den Berg *et al.* (1998) TEFs were used, and corresponding mean and RME HQs were 2.1 and 3.7, respectively. Total TEQ mean and RME NOAEC HQs exceeded 1.0 for whole double-crested cormorants using both the Tillitt *et al.* (1991b) TEFs where HQs were 3.0 and 6.4, respectively, and the Van den Berg *et al.* (1998) TEFs where HQs were 35 and 61, respectively. Total TEQ

mean and RME LD_{20} HQs exceeded 1.0 for whole double-crested cormorants using the Van den Berg *et al.* (1998), TEFs where HQs were 1.3 and 2.2, respectively.

Total TEQ mean and RME reproduction NOAEC HQs exceeded 1.0 for double-crested cormorant eggs using both the Tillitt *et al.* (1991b) TEFs where HQs were 3.8 and 5.1, respectively, and the Van den Berg *et al.* (1998) TEFs where HQs were 31 and 46, respectively. Total TEQ mean and RME LD_{20} HQs exceeded 1.0 for whole double-crested cormorants using the Van den Berg *et al.* (1998) TEFs, where HQs were 1.1 and 1.7, respectively. The total TEQ RME LD_{30} HQ (1.1) was just over 1.0. Total TEQ deformity NOAEC HQs that exceeded 1.0 for double-crested cormorant eggs were the RME HQ of 1.5 using the Tillitt *et al.* TEFs, and the mean and RME HQ of 5.7 and 8.5 using the Van den Berg *et al.* TEFs.

Total TEQ mean and RME NOAEC HQs exceeded 1.0 for common tern eggs using both the Tillitt *et al.* (1991b) TEFs, where HQs were 11 and 7.5, respectively, and the Van den Berg *et al.* (1998) TEFs, where HQs were 146 and 110, respectively. Total TEQ mean and RME LD_{20} HQs exceeded 1.0 for whole common tern eggs using the Van den Berg *et al.* (1998) TEFs, where HQs were 5.3 and 4.0, respectively. Total TEQ mean and RME LD_{30} HQs exceeded 1.0 for whole common tern eggs using the Van den Berg *et al.* (1998) TEFs where HQs were 3.3 and 2.5, respectively.

Total TEQ mean and RME NOAEC HQs exceeded 1.0 for Forster's tern eggs using both the Tillitt *et al.* (1991b) TEFs, where HQs were 15 and 3.3, respectively, and the Van den Berg *et al.* (1998) TEFs, where HQs were 175 and 55, respectively. Total TEQ mean and RME LD_{20} HQs exceeded 1.0 for whole Forster's tern eggs using the Van den Berg *et al.* (1998) TEFs, where HQs were 6.4 and 2.0, respectively. Total TEQ mean and RME LD_{30} HQs exceeded 1.0 for whole Forster's tern eggs using the Van den Berg *et al.* (1998) TEFs, where HQs were 4.0 and 1.3, respectively.

Mink. COPC data are not available for risk evaluation in Green Bay Zone 2. Therefore, the only piscivorous mammal HQs that were evaluated were those that were estimated, based on dietary intake, for the COPCs mercury, total PCBs, dieldrin, and p,p'-DDE (Table 6-102). The only estimated HQs that exceeded 1.0 were those for total PCBs. Sediment was assumed to be a small fraction of the diet; therefore, two HQs were developed for total PCBs based on the sediment concentrations used (N and I_d). Total PCB NOAEC HQs ranged from 318 to 354 and total PCB LOAEC HQs ranged from 9.8 to 11.

Green Bay Zone 3A

Water. COPC data available for risk evaluation in Green Bay Zone 3A included filtered and unfiltered mercury, and filtered and particulate total PCBs. Hazard quotients for surface water COPCs detected in Green Bay Zone 3A are given in Table 6-53. No HQs exceeded 1.0.

Sediment. COPC data available for risk evaluation in Green Bay Zone 3A included arsenic, lead, mercury, total PCBs, dieldrin, p,p'-DDD, p,p'-DDE, and p,p'-DDT. Hazard quotients for surface sediment COPCs detected in Green Bay Zone 3A are given in Table 6-54. HQs that exceeded 1.0 included the mean and RME HQs for total PCBs (7.7 to 16).

Fish. COPC data available for risk evaluation in Green Bay Zone 3A included mercury, total PCBs, PCB congeners, dieldrin, o,p'-DDD, o,p'-DDE, o,p'-DDT, p,p'-DDD, p,p'-DDE, and p,p'-DDT. Hazard quotients for fish COPCs detected in Green Bay Zone 3A are given in Tables 6-103 and 6-104. HQs that exceeded 1.0 included the total PCB NOAEC RME HQ for gizzard shad (4.6), and the total PCB NOAEC mean and RME HQs for alewife (1.2 and 1.7, respectively), walleye (5.5 and 6.7, respectively), and brown trout (4.3 and 4.8, respectively).

Birds. COPC data available for risk evaluation in Green Bay Zone 3A included mercury, total PCBs, dieldrin, p,p'-DDD, p,p'-DDE, and p,p'-DDT. Hazard quotients for bird COPCs detected in Green Bay Zone 3A are given in Table 6-105. Total PCB NOAEC and LOAEC HQs for bald eagle eggs exceeded 1.0 for both endpoints—reproduction and deformity, where the reproduction HQs were 2.8 and 1.7, respectively, and the deformity HQs were 16 and 1.6, respectively. The only other bald eagle egg HQ that exceeded 1.0 was the NOAEC RME of 2.0 for dieldrin.

HQs were estimated for all piscivorous birds and the COPCs mercury, total PCBs, dieldrin, and p,p'-DDE based on dietary intake (Table 6-106). Estimated HQs that exceeded 1.0 were the mercury mean and RME NOAEC HQs for common tern (1.8 and 2.5, respectively) and Forster's tern (1.7 and 2.3, respectively), and total PCB mean and RME NOAEC HQs for common tern (4.0 and 5.6, respectively) and Forster's tern (3.7 and 5.1, respectively), double-crested cormorant (1.5 and 2.1, respectively), and bald eagle (2.9 and 4.2, respectively).

Mink. COPC data are not available for risk evaluation in Green Bay Zone 3A. Therefore, the only piscivorous mammal HQs that were evaluated were those that were estimated, based on dietary intake, for the COPCs mercury, total PCBs, dieldrin, and p,p'-DDE (Table 6-107). The only estimated HQs that exceeded 1.0 were those for total PCBs and dieldrin. Sediment was assumed to be a small

fraction of the diet; therefore, two HQs were developed for total PCBs based on the sediment concentrations used (N and I_d). Total PCB NOAEC HQs ranged from 127 to 191 and total PCB LOAEC HQs ranged from 3.9 to 5.9. The only dieldrin HQ that exceeded 1.0 was the RME NOAEC HQ of 1.2.

Green Bay Zone 3B

Water. COPC data available for risk evaluation in Green Bay Zone 3B included filtered and unfiltered mercury, and filtered and particulate total PCBs. Hazard quotients for surface water COPCs detected in Green Bay Zone 3B are given in Table 6-61. No HQs exceeded 1.0.

Sediment. COPC data available for risk evaluation in Green Bay Zone 3B included arsenic, lead, mercury, total PCBs, dieldrin, p,p'-DDD, p,p'-DDE, and p,p'-DDT. Hazard quotients for surface sediment COPCs detected in Green Bay Zone 3B are given in Table 6-62. HQs that exceeded 1.0 included the mean and RME HQs for total PCBs (15 to 26), and the RME HQs for arsenic (1.2), lead (1.4), and mercury (1.1).

Fish. COPC data available for risk evaluation in Green Bay Zone 3B included mercury, total PCBs, dieldrin, p,p'-DDD, p,p'-DDE, and p,p'-DDT. Hazard quotients for fish COPCs detected in Green Bay Zone 3B are given in Tables 6-108 and 6-109. HQs that exceeded 1.0 included the mercury NOAEC RME HQ for walleye (2.6), the total PCB NOAEC RME HQ for rainbow smelt (1.1), the total PCB NOAEC mean and RME HQs for alewife (2.4 and 3.1, respectively), walleye (8.5 and 15, respectively), and brown trout (2.9 and 3.5, respectively), and the p,p'-DDE NOAEC RME HQ for walleye (1.8).

Birds. COPC data available for risk evaluation in Green Bay Zone 3B included total PCBs, dieldrin, o,p'-DDD, o,p'-DDE, o,p'-DDT, p,p'-DDD, p,p'-DDE, and p,p'-DDT. Hazard quotients for bird COPCs detected in Green Bay Zone 3B are given in Table 6-110. Total PCB NOAEC and LOAEC HQs exceeded 1.0 for whole double-crested cormorants and for both endpoints—reproduction and deformity. HQs for reproduction ranged from 0.7 to 3.2 and deformity HQs ranged from 0.7 to 19. Dieldrin mean and RME NOAEC HQs for whole double-crested cormorants (1.3 and 2.4, respectively) also exceeded 1.0. The only other HQ that exceeded 1.0 in this species was a p,p'-DDE RME NOAEC HQ of 1.5.

HQs were estimated for all piscivorous birds and the COPCs mercury, total PCBs, dieldrin, and p,p'-DDE based on dietary intake (Table 6-111). Estimated HQs that exceeded 1.0 were the mercury mean and RME NOAEC HQs for common tern (1.5 and 3.1, respectively), Forster's tern (1.4 and 2.8, respectively), and bald

eagle (1.9 and 3.7, respectively), the total PCB mean and RME NOAEC HQs for common tern (8.0 and 10, respectively), Forster's tern (7.3 and 9.6, respectively), double-crested cormorant (3.1 and 4.0, respectively), and bald eagle (5.2 and 7.2, respectively), and the p,p'-DDE mean and RME NOAEC HQs for common tern (2.2 and 2.2, respectively) and Forster's tern (2.0 and 2.0, respectively). The mercury RME NOAEC HQ for double-crested cormorants (1.2) exceeded 1.0, as did the p,p'-DDE RME NOAEC HQ for bald eagles (1.9).

Hazard quotients for PCB congeners in birds from Green Bay Zone 3B are contained in Table 6-112 (whole double-crested cormorants). Total TEQ mean and RME NOAEC HQs exceeded 1.0 for whole double-crested cormorants using both the Tillitt *et al.* (1991b) TEFs, where HQs were 2.1 and 2.8, respectively, and the Van den Berg *et al.* (1998) TEFs, where HQs were 12 and 18, respectively.

Mink. COPC data are not available for risk evaluation in Green Bay Zone 3B. Therefore, the only piscivorous mammal HQs that were evaluated were those that were estimated, based on dietary intake, for the COPCs mercury, total PCBs, dieldrin, and p,p'-DDE (Table 6-113). The only estimated HQs that exceeded 1.0 were those for total PCBs and dieldrin. Sediment was assumed to be a small fraction of the diet; therefore, two HQs were developed for total PCBs based on the sediment concentrations used (N and I_d). Total PCB NOAEC HQs ranged from 237 to 295 and total PCB LOAEC HQs ranged from 7.3 to 9.1. The only dieldrin HQ that exceeded 1.0 was the RME NOAEC HQ of 1.2.

Green Bay Zone 4

Water. COPC data available for risk evaluation in Green Bay Zone 4 included filtered and unfiltered mercury, and filtered and particulate total PCBs. Hazard quotients for surface water COPCs detected in Green Bay Zone 3B are given in Table 6-68. No HQs exceeded 1.0.

Sediment. COPC data available for risk evaluation in Green Bay Zone 4 included arsenic, lead, mercury, total PCBs, dieldrin, p,p'-DDD, p,p'-DDE, and p,p'-DDT. Hazard quotients for surface sediment COPCs detected in Green Bay Zone 4 are given in Table 6-71. HQs that exceeded 1.0 included the mean and RME HQs for total PCBs (1.1 to 3.7).

Fish. COPC data available for risk evaluation in Green Bay Zone 4 included mercury, total PCBs, dieldrin, p,p'-DDD, p,p'-DDE, and p,p'-DDT. Hazard quotients for fish COPCs detected in Green Bay Zone 4 are given in Tables 6-114 and 6-115. HQs that exceeded 1.0 included the total PCB NOAEC mean and RME HQs for alewife (1.4 and 2.0, respectively), walleye (3.4 and 4.3, respectively), and brown

trout (3.2 and 3.6, respectively), and the p,p'-DDE NOAEC mean and RME HQs for walleye (1.6 and 2.0, respectively).

Birds. COPC data are not available for risk evaluation in Green Bay Zone 4. Therefore, the only HQs that were evaluated were those that were estimated, based on dietary intake, for all piscivorous birds and the COPCs mercury, total PCBs, dieldrin, and p,p'-DDE (Table 6-116). Estimated HQs that exceeded 1.0 were the mercury mean and RME NOAEC HQs for common tern (1.8 and 1.8, respectively), Forster's tern (1.7 and 1.7, respectively), and bald eagle (2.5 and 2.9, respectively), the total PCB mean and RME NOAEC HQs for common tern (4.5 and 6.5, respectively), Forster's tern (4.2 and 6.0, respectively), double-crested cormorant (1.8 and 2.5, respectively), and bald eagle (2.9 and 4.3, respectively), and the p,p'-DDE mean and RME NOAEC HQs for bald eagle (5.0 and 6.5, respectively).

Mink. COPC data are not available for risk evaluation in Green Bay Zone 4. Therefore, the only piscivorous mammal HQs that were evaluated were those that were estimated, based on dietary intake, for the COPCs mercury, total PCBs, dieldrin, and p,p'-DDE (Table 6-117). The only estimated HQs that exceeded 1.0 were those for total PCBs. Sediment was assumed to be a small fraction of the diet; therefore, two HQs were developed for total PCBs based on the sediment concentrations used (N and I_d). Total PCB NOAEC HQs ranged from 143 to 219 and total PCB LOAEC HQs ranged from 4.4 to 6.7.

6.5.2 Risk Summary by Media

This section provides a summary of the observed risks by media: surface water, surface sediment, fish, birds, and mammals.

Water

Hazard quotients that exceeded 1.0 in surface water are presented on Figures 6-50 and 6-51. These HQs included the COPCs mercury and total PCBs, and the mercury HQs were consistently greater than the total PCB HQs. As indicated on these figures, no surface water HQs were greater than 1.0 for any COPC analyzed in Green Bay zones 3A, 3B, and 4.

Sediment

Hazard quotients that exceeded 1.0 in surface sediment are presented on Figures 6-52 through 6-56. These HQs included the COPCs arsenic, lead, mercury, total PCBs, p,p'-DDD, p,p'-DDE, and p,p'-DDT. As indicated on these figures, each area evaluated had HQs that were greater than 1.0. Unlike the HQs that exceeded 1.0 in water, in sediment the PCB HQs consistently exceeded the metal HQs. For metals, lead HQs were highest in the Little Lake Butte des Morts

Reach, and mercury HQs were highest in the Little Rapids to De Pere Reach (Figure 6-54). For total PCBs, HQs were highest in Little Lake Butte des Morts Reach, and decreased fairly consistently moving down the river and out to Green Bay Zone 4 (Figure 6-55). Pesticide HQs exceeded 1.0 in Little Lake Butte des Morts Reach, Little Rapids to De Pere Reach, and Green Bay Zone 1, and were greatest in the Little Rapids to De Pere Reach (Figure 6-56).

Fish

Hazard quotients that exceeded 1.0 in whole fish are presented on Figures 6-57 through 6-62. These HQs included the COPCs mercury, total PCBs, total TEQ, and p,p'-DDE. As indicated on these figures, each area evaluated had HQs that were greater than 1.0. HQs that exceeded 1.0 in fish did not differ by orders of magnitude as they did for COPC HQs in sediment. In concurrence with the water and sediment HQs for mercury, mercury HQs in whole fish were greatest in the Little Rapids to De Pere Reach (Figure 6-59). Total PCB HQs that exceeded 1.0 for all whole fish are presented on Figure 6-60 and for selected fish (alewife, shiner species, walleye, and carp) are presented on Figure 6-61. General trends in total PCB HQs in whole fish are that HQs in lower trophic level fish (alewife and shiners) were less than HQs in higher trophic level fish (walleye and carp). Interestingly, while sediment total PCB HQs were highest in the Little Lake Butte des Morts and then decreased moving out into the bay, total PCB HQs in carp and walleye were greater in the Appleton to Little Rapids Reach than the Little Lake Butte des Morts Reach, and HQs continued increasing in these fish moving down the river. Total PCB HQs in carp were at a maximum in Green Bay zones 1 and 2, and in walleye they were the greatest in Green Bay Zone 4. DDE HQs exceeded 1.0 in only a few fish sampled and in only a few areas (Figure 6-62).

Bird

Hazard quotients that exceeded 1.0 in birds are presented on Figures 6-63 through 6-72. These HQs included the COPCs mercury, total PCBs, total TEQ, dieldrin, and p,p'-DDE. As indicated on these figures, each area evaluated had HQs that were greater than 1.0. Total PCB and TEQ HQs that exceeded 1.0 for all birds (egg and whole body) are presented on Figures 6-65a and 6-65b. Overall, potential risk from deformity exceeded the potential risk from reproductive impairment. Total TEQs were estimated using two TEF systems: Tillitt *et al.* (1991b) and Van den Berg *et al.* (1998). The HQs calculated from the Tillitt *et al.* TEFs more closely approximated the predicted risk from reproductive impairment, and the HQs calculated from the Van den Berg *et al.* TEFs more closely approximated the predicted risk of deformity. All other bird HQs that exceeded 1.0 (those for mercury, dieldrin, and p,p'-DDE) are presented on Figure 6-66. While the mercury HQ (7.0) exceeded all of the other HQs, only p,p'-DDE had a LOAEC HQ (1.4) that exceeded 1.0. Interestingly, while HQs for dieldrin

did not exceed 1.0 for water, sediment, or fish, dieldrin HQs in birds did exceed 1.0 due to bioaccumulation.

Estimated HQs for piscivorous birds are presented on Figures 6-67 through 6-72. Estimated HQs that exceeded 1.0 included the COPCs mercury, total PCBs, and p,p'-DDE and all species modeled (common tern, Forster's tern, double-crested cormorant, and bald eagle). Estimated HQs for mercury were highest in Green Bay zones 1 and 2, where LOAEC RME HQs exceeded 1.0 (Figure 6-69). Estimated total PCB HQs closely resembled the total PCB HQs for fish both in magnitude and trend over all areas (Figure 6-70). For all of Green Bay (zones 1 through 4), only double-crested cormorant and common tern total PCB HQs exceeded 1.0. As compared to the measured total PCB HQs, estimated HQs always exceeded the measured HQs, except for the HQ for bald eagle based on the single egg analyzed (Figure 6-71). Although estimated HQs generally exceeded the measured HQs, these differences were less than an order of magnitude for double-crested cormorants and bald eagles. Estimated HQs for p,p'-DDE that exceeded 1.0 in piscivorous birds are presented on Figure 6-72. Green Bay zones 1 and 2 were the only zones for which estimated HQs exceeded 1.0 for all species; modeled (common tern, Forster's tern, double-crested cormorant, and bald eagle); however, the estimated bald eagle HQ in Green Bay Zone 4 was the highest estimated HQ. The only instance in which measured and estimated HQs could be compared was for double-crested cormorants in Green Bay Zone 2. The estimated HQs of 1.1 (NOAEC mean) and 1.5 (NOAEC RME) closely approximated the measured HQs of 1.4 (NOAEC mean) and 2.4 (NOAEC RME) in double-crested cormorant eggs and 1.2 (NOAEC mean) in whole double-crested cormorants. Although dieldrin HQs did exceed 1.0 in measured bird tissues, estimated HQs for dieldrin in all areas and for all birds did not exceed 1.0.

Piscivorous Mammals

Estimated hazard quotients that exceeded 1.0 in mink are presented on Figures 6-73 through 6-75. The only estimated HQs that exceeded 1.0 were the HQs for total PCBs, but all total PCB HQs (mean and RME, NOAEC and LOAEC) exceeded 1.0. Risk was highest in Green Bay zones 1 and 2 followed by Zone 3B.

6.5.3 Risk Summary by Area

This section provides a summary of the predicted risks by reach and zone. Data and supporting information are arranged below in a format designed to answer the specific risk questions (see Table 6-2) for each reach and zone examined. Summary tables are provided which present the assessment endpoints, risk questions, constituents analyzed, NOAEC and LOAEC RME HQ values evaluated for risk, and the risk conclusions. For this risk assessment it was agreed by BTAG that degree of risk would be determined based on three categories: "no" risk was

concluded when both the NOAEC and LOAEC HQs evaluated were less than 1.0, "potential" risk was concluded when the NOAEC HQ exceeded 1.0 but the LOAEC HQ was less than 1.0, and risk ("yes") was concluded when both the NOAEC and LOAEC HQs evaluated were greater than 1.0. When constituents were analyzed but not detected, it was concluded that no risk existed. However, there were also cases where risk could not be evaluated because constituents were not analyzed. Within this text, where risk is identified, the site contaminant may be referred to as a COC (chemical of concern). Table 6-134 summarizes the risk to each assessment endpoint in each reach and zone.

Summary tables for each reach and zone are explained in more detail in this section.

Little Lake Butte des Morts Reach

A summary of all RME HQs for Little Lake Butte des Morts Reach is presented in Table 6-121 along with the relevant assessment endpoints and risk questions. Figure 6-76 presents HQs that exceeded 1.0 in Little Lake Butte des Morts Reach. Assessment endpoints, risk questions, and determined risk are summarized below.

For the Assessment Endpoint—Functioning Water Column Invertebrate Communities. *Are levels of site contaminants in surface water sufficient to cause adverse alterations to the functioning of water column invertebrate communities?*

Concentrations of chemicals of concern are at sufficient levels to pose risk in Little Lake Butte des Morts. Based on the HQs alone, the data suggest that mercury likely poses risk, and that PCBs in the water column have the potential for causing risk. Lead is not at sufficient concentrations to pose risk, and the remaining COPCs were not analyzed in this reach. The HQ risk estimated from mercury was based upon the maximum detected value, and thus potentially overestimates the risk.

For the Assessment Endpoint—Functioning Benthic Invertebrate Communities. *Are levels of site contaminants in sediment sufficient to cause adverse alterations to the functioning of benthic invertebrate communities?*

Based upon the estimated hazard quotients, there are persistent risks to benthic infaunal communities from contaminants in sediments in the Little Lake Butte des Morts Reach. The calculated HQs, concentrations of lead, mercury, 2,3,7,8-TCDD, total PCBs, p,p'-DDD, and p,p'-DDT in the sediment are at sufficient concentrations to cause adverse alterations to benthic invertebrate communities. Total PCB HQs are 50 to 1,000 times greater than any other COC. Arsenic, dieldrin, and p,p'-DDE are not at sufficient concentrations to pose risk. Sediment

isopleths for PCB distribution (see Figure 2-2) show that elevated concentrations above the applied sediment thresholds are widely distributed throughout Little Lake Butte des Morts.

For the Assessment Endpoint—Benthic Fish Reproduction and Survival. *Are levels of site contaminants sufficient to cause survival or reproductive impairment to benthic fish?*

Concentrations of total PCBs in benthic fish may be sufficiently high to potentially be of risk to benthic fish reproduction or survival. However, when examined on a coplanar-specific basis, PCB HQs are less than 0.1 for both the NOAEC and LOAEC. Arsenic, mercury, and all chlorinated pesticides (dieldrin, o,p'-DDD, o,p'-DDE, o,p'-DDT, p,p'-DDD, p,p'-DDE, and p,p'-DDT) are not at sufficient concentrations to pose risk.

For the Assessment Endpoint—Pelagial Fish Reproduction and Survival. *Are levels of site contaminants sufficient to cause survival or reproductive impairment to pelagial fish?*

The derived hazard quotients for reproduction suggest that concentrations of total PCBs in walleye, but not perch, are sufficiently high to potentially be of risk. However, when examined on a total TEQ basis, both the NOAEC and LOAEC in walleye and perch are less than 1.0. Measured concentrations of mercury and all chlorinated pesticides (dieldrin, o,p'-DDD, o,p'-DDE, o,p'-DDT, p,p'-DDD, p,p'-DDE, and p,p'-DDT) for both fish species are not at sufficient levels to pose risk.

For the Assessment Endpoint—Insectivorous Bird Reproduction and Survival. *Are levels of site contaminants sufficient to cause survival or reproductive impairment or deformity in insectivorous birds?*

Based upon evaluation of the hazard quotients alone, only total PCBs, and PCB congeners are found at sufficient levels to potentially cause survival or reproductive impairment in insectivorous birds. This included HQs for reproductive success, as well as for potential deformity in hatchlings. There are no potential risk effects attributable to the chlorinated pesticides (dieldrin, o,p'-DDD, o,p'-DDE, o,p'-DDT, p,p'-DDD, p,p'-DDE, and p,p'-DDT).

For the Assessment Endpoint—Piscivorous Bird Reproduction and Survival. *Are levels of site contaminants sufficient to cause survival or reproductive impairment or deformity in piscivorous birds?*

Based upon modeled dietary intake of COCs, mercury and total PCBs are estimated to be at sufficient concentrations to cause potential adverse effects on survival, physiology, or reproduction of common and Forster's tern, but not

double-crested cormorants in Little Lake Butte des Morts Reach. It is noted that the estimated NOAEC HQs are low; between 1.0 and 2.0 for both mercury and PCBs. Estimated concentrations of the chlorinated pesticides (dieldrin and p,p'-DDE) are not at sufficient concentrations to pose risk.

For the Assessment Endpoint—Carnivorous Bird Reproduction and Survival. *Are levels of site contaminants sufficient to cause survival or reproductive impairment or deformity in carnivorous birds?*

Modeled dietary intake of total PCBs in carnivorous birds are estimated to be at sufficient concentrations to cause potential adverse effects on survival, physiology, or reproduction. The modeled, diet-based NOAEC HQs estimated low-level risks (RME HQ of 3.2, mean HQ of 1.8), based on total PCBs in fish as the exposure point. Estimated concentrations of mercury and the chlorinated pesticides (dieldrin and p,p'-DDE) are not at sufficient concentrations to pose risk.

For the Assessment Endpoint—Piscivorous Mammal Reproduction and Survival. *Are levels of site contaminants sufficient to cause survival or reproductive impairment to piscivorous mammals?*

Modeled concentrations of total PCBs in the diet of piscivorous mammals are at sufficient concentrations to cause adverse effects on survival or reproduction. Estimated concentrations of mercury and the chlorinated pesticides (dieldrin and p,p'-DDE) are not at sufficient concentrations to pose risk.

Little Lake Butte des Morts Summary. In summary, the results suggest that only measured or estimated concentrations of total PCBs are at sufficient levels to cause risk to benthic invertebrates, and piscivorous mammals. Potential risks from total PCBs are indicated for water column invertebrates, benthic and pelagic fish, and insectivorous, piscivorous, and carnivorous birds. Measured or estimated concentrations of mercury are found to be at sufficient concentrations to cause or potentially cause risk to water column and benthic invertebrates, and piscivorous birds. Concentrations of 2,3,7,8-TCDD, DDD, and DDT are only sufficient to be of risk to benthic invertebrates. Sediment concentrations of elevated PCBs are widespread and persistent throughout the reach. Concentrations of arsenic, dieldrin, and all o,p'- isomers of DDT and its metabolites are not found to pose risk to any assessment endpoint.

Appleton to Little Rapids Reach

A summary of all RME HQs for the Appleton to Little Rapids Reach is presented in Table 6-122 along with the relevant assessment endpoints and risk questions. Figure 6-77 presents HQs that exceeded 1.0 in the Appleton to Little Rapids

Reach. Assessment endpoints, risk questions and determined risk are summarized below.

For the Assessment Endpoint—Functioning Water Column Invertebrate Communities. *Are levels of site contaminants in surface water sufficient to cause adverse alterations to the functioning of water column invertebrate communities?*

There are no to low levels of risks to functioning water column invertebrate communities in the Appleton to Little Rapids Reach. Based on the HQs alone, PCBs in the water column have the potential for causing risk, but this is based on an HQ of 1.2. Arsenic, lead, mercury, and the chlorinated pesticides (dieldrin, p,p'-DDD, p,p'-DDE, p,p'-DDT) are not at sufficient concentrations to pose risk.

For the Assessment Endpoint—Functioning Benthic Invertebrate Communities. *Are levels of site contaminants in sediment sufficient to cause adverse alterations to the functioning of benthic invertebrate communities?*

Benthic infaunal communities in sediments of the Appleton to Little Rapids Reach are potentially at risk. Based on the calculated HQs, concentrations of lead, mercury, and total PCBs in the sediment are sufficient to cause adverse alterations to benthic invertebrate communities. Arsenic, dieldrin, p,p'-DDD, p,p'-DDE, and p,p'-DDT are not at sufficient concentrations to pose risk.

For the Assessment Endpoint—Benthic Fish Reproduction and Survival. *Are levels of site contaminants sufficient to cause survival or reproductive impairment to benthic fish?*

Concentrations of total PCBs in benthic fish may be sufficiently high to cause reproductive or survival impairment to benthic fish populations. Based upon the calculated hazard quotients, only concentrations of bioaccumulated total PCBs in benthic fish may be sufficiently high to potentially be of risk to benthic fish reproduction or survival. HQs calculated based on tissue residue values exceeded 1.0 for the NOAEC (HQ = 4.7), but were less than 1.0 for the LOAEC. Mercury, PCB congeners, and all chlorinated pesticides (dieldrin, o,p'-DDD, o,p'-DDE, o,p'-DDT, p,p'-DDD, p,p'-DDE, p,p'-DDT) are not at sufficient concentrations to pose risk.

For the Assessment Endpoint—Pelagial Fish Reproduction and Survival. *Are levels of site contaminants sufficient to cause survival or reproductive impairment to pelagial fish?*

Concentrations of total PCBs in pelagial fish are sufficiently high to potentially be of risk to reproduction or survival. For walleye and perch, the NOAEC, but not the LOAEC, exceeds 1.0. However, for both fish species, the TEQ-based HQ

is less than 1.0 for both the NOAEC and the LOAEC. Concentrations of mercury, PCB congeners, and all chlorinated pesticides (dieldrin, p,p'-DDD, p,p'-DDE, p,p'-DDT) are not at sufficient concentrations to pose risk.

For the Assessment Endpoint—Insectivorous Bird Reproduction and Survival. *Are levels of site contaminants sufficient to cause survival or reproductive impairment or deformity in insectivorous birds?*

Data were not available for the estimation of risk to insectivorous birds.

For the Assessment Endpoint—Piscivorous Bird Reproduction and Survival. *Are levels of site contaminants sufficient to cause survival or reproductive impairment or deformity in piscivorous birds?*

Estimated concentrations of mercury and total PCBs in the diet of piscivorous birds are at sufficient concentrations to potentially cause adverse effects to survival, physiology, or reproduction. Estimated concentrations of the chlorinated pesticides (dieldrin and p,p'-DDE) are not at sufficient concentrations to pose risk.

For the Assessment Endpoint—Carnivorous Bird Reproduction and Survival. *Are levels of site contaminants sufficient to cause survival or reproductive impairment or deformity in carnivorous birds?*

Measured concentrations in tissues and estimated dietary intake of mercury and PCBs in bald eagles are at sufficient concentrations to cause potential adverse effects. Measured and estimated concentrations of dieldrin and p,p'-DDE and measured concentrations of p,p'-DDD and p,p'-DDT are not at sufficient concentrations to pose risk. Based on the measured concentrations of total PCBs in a single eagle egg collected in 1990, the NOAEC HQ is 45.

For the Assessment Endpoint—Piscivorous Mammal Reproduction and Survival. *Are levels of site contaminants sufficient to cause survival or reproductive impairment to piscivorous mammals?*

Modeled concentrations of total PCBs in the diet of piscivorous mammals are at sufficient concentrations to cause adverse effects on survival or reproduction. Estimated concentrations of mercury and the chlorinated pesticides (dieldrin and p,p'-DDE) are not at sufficient concentrations to pose risk.

Appleton to Little Rapids Summary. In summary, the results taken in total suggest that measured or estimated concentrations of total PCBs are at sufficient levels to

cause risk to benthic invertebrates, carnivorous birds, and piscivorous mammals. Potential risks are indicated for all other receptors except insectivorous birds, for which there are no data. Measured or estimated concentrations of mercury were found to be at sufficient concentrations to cause of potentially cause risk to benthic invertebrates, piscivorous birds, and carnivorous birds. Concentrations of lead are only of risk to benthic invertebrates. Concentrations of all chlorinated pesticides (dieldrin, o,p'-DDD, o,p'-DDE, o,p'-DDT, p,p'-DDD, p,p'-DDE, p,p'-DDT) are not found to pose risk to any assessment endpoint. Surface sediment concentrations of elevated PCBs indicate reach-wide effects, but are likely limited to specific deposits.

Little Rapids to De Pere Reach

A summary of all RME HQs for Little Rapids to De Pere Reach is presented in Table 6-123 along with the relevant assessment endpoints and risk questions. Figure 6-78 presents HQs that exceeded 1.0 in the Little Rapids to De Pere Reach. Assessment endpoints, risk questions, and determined risk are summarized below.

For the Assessment Endpoint—Functioning Water Column Invertebrate Communities. *Are levels of site contaminants in surface water sufficient to cause adverse alterations to the functioning of water column invertebrate communities?*

Based on the calculated HQs alone, mercury is the only COC at concentrations that pose risk to the functioning of water column invertebrate communities with a calculated HQ of 16. Lead and total PCBs are not at sufficient concentrations to pose risk.

For the Assessment Endpoint—Functioning Benthic Invertebrate Communities. *Are levels of site contaminants in sediment sufficient to cause adverse alterations to the functioning of benthic invertebrate communities?*

There are persistent risks to benthic infaunal communities from COCs in sediments in the Little Rapids to De Pere Reach. Based on the calculated HQs, concentrations of lead, mercury, 2,3,7,8-TCDD, total PCBs, p,p'-DDE, and p,p'-DDT in the sediment are at sufficient concentrations to cause adverse alterations to benthic invertebrate communities. All indicators for total PCBs (surface-weighted average concentrations) exceeded the sediment TRV and the HQs are 10 to 1,000 times greater than any other COC. Arsenic, dieldrin, and p,p'-DDD are not at sufficient concentrations to pose risk.

For the Assessment Endpoint—Benthic Fish Reproduction and Survival. *Are levels of site contaminants sufficient to cause survival or reproductive impairment to benthic fish?*

Based upon calculated HQs alone, levels of site mercury and total PCBs in benthic fish are sufficiently high to potentially be of risk to benthic fish reproduction or survival. All chlorinated pesticides (dieldrin, o,p'-DDD, o,p'-DDE, o,p'-DDT, p,p'-DDD, p,p'-DDE, p,p'-DDT) are not at sufficient concentrations to pose risk.

For the Assessment Endpoint—Pelagial Fish Reproduction and Survival. *Are levels of site contaminants sufficient to cause survival or reproductive impairment to pelagial fish?*

Concentrations of mercury and total PCBs in walleye may be sufficiently high to potentially be of risk. However, the TEQ HQs calculated from measured dioxins and PCB coplanar congeners for both walleye and perch are less than or equal to 0.2. Mercury was detected in the single walleye sample analyzed in this reach at levels yielding a NOAEC HQ of 6.4, but a LOAEC HQ of 0.1. Mercury in the single perch sample analyzed was not detected. Concentrations of all chlorinated pesticides (dieldrin, o,p'-DDD, o,p'-DDE, o,p'-DDT, p,p'-DDD, p,p'-DDE, p,p'-DDT) are not at sufficient concentrations to pose risk.

For the Assessment Endpoint—Insectivorous Bird Reproduction and Survival. *Are levels of site contaminants sufficient to cause survival or reproductive impairment or deformity in insectivorous birds?*

Data are not available for the estimation of risk to insectivorous birds.

For the Assessment Endpoint—Piscivorous Bird Reproduction and Survival. *Are levels of site contaminants sufficient to cause survival or reproductive impairment or deformity in piscivorous birds?*

Modeled dietary exposures to concentrations of mercury and total PCBs measured in fish are estimated to be at sufficient concentrations to cause potential adverse effects on survival or reproduction of common and Forster's terns. Mercury, but not total PCB, risks are also indicated for double-crested cormorants. Modeled intake of the chlorinated pesticides (dieldrin and p,p'-DDE) are not at sufficient levels to pose risk.

For the Assessment Endpoint—Carnivorous Bird Reproduction and Survival. *Are levels of site contaminants sufficient to cause survival or reproductive impairment or deformity in carnivorous birds?*

Modeled dietary intake of mercury and total PCBs in the diet of carnivorous birds are at sufficient concentrations to cause potential adverse effects on survival, physiology, or reproduction. The resultant modeled HQs for the NOAEC are 2.2 and 5.6, respectively, while the LOAEC HQs are less than 1.0. Estimated intake

of the chlorinated pesticides (dieldrin and p,p'-DDE) are not at sufficient concentrations to pose risk.

For the Assessment Endpoint—Piscivorous Mammal Reproduction and Survival. *Are levels of site contaminants sufficient to cause survival or reproductive impairment to piscivorous mammals?*

Modeled concentrations of total PCBs in the diet of piscivorous mammals are at sufficient concentrations to cause adverse effects on survival or reproduction. Estimated concentrations of mercury and the chlorinated pesticides (dieldrin and p,p'-DDE) are not at sufficient concentrations to pose risk.

Little Rapids to De Pere Summary. In summary, the results suggest that measured or estimated concentrations of total PCBs are at sufficient levels to cause risk to benthic invertebrates, and piscivorous mammals. Potential risks are indicated for benthic and pelagic fish, and piscivorous and carnivorous birds. There are no data to evaluate insectivorous birds. Measured or estimated concentrations of mercury are found to be at sufficient concentrations to cause, or potentially cause, risk to aquatic invertebrates, benthic invertebrates, pelagic fish, piscivorous birds, and carnivorous birds. There are persistent risks to benthic infaunal communities in sediments from exposure to lead, mercury, 2,3,7,8-TCDD, total PCBs, p,p'-DDE, and p,p'-DDT. Concentrations of arsenic, dieldrin, all o,p'- isomers of DDT and its metabolites, and p,p'-DDD are not sufficient to pose risk to any assessment endpoint.

De Pere to Green Bay Reach (Green Bay Zone 1)

A summary of all RME HQs for Green Bay Zone 1 is presented in Table 6-124, along with the relevant assessment endpoints and risk questions. Figure 6-79 presents HQs that exceeded 1.0 in Green Bay Zone 1. Summaries of risk to fish and piscivorous birds in this zone of Green Bay are presented with the results for Green Bay Zone 2, because the areas are not distinct for the purposes of assessing risk to the fish and bird assessment endpoints. Assessment endpoints, risk questions, and determined risk are summarized below.

For the Assessment Endpoint—Functioning Water Column Invertebrate Communities. *Are levels of site contaminants in surface water sufficient to cause adverse alterations to the functioning of water column invertebrate communities?*

There are no to low levels of risks to functioning water column invertebrate communities in this reach. Based on the HQs alone, the data suggest that only total PCBs in the water column have the potential for causing risk. Arsenic, lead,

mercury, 2,3,7,8-TCDD, 2,3,7,8-TCDF, p,p'-DDD, p,p'-DDE, and p,p'-DDT are not at sufficient concentrations to pose risk.

For the Assessment Endpoint—Functioning Benthic Invertebrate Communities. *Are levels of site contaminants in sediment sufficient to cause adverse alterations to the functioning of benthic invertebrate communities?*

There are persistent risks to benthic infaunal communities in sediments of this reach. Based on the calculated HQs, concentrations of arsenic, lead, mercury, total PCBs, p,p'-DDD, and p,p'-DDE in the sediment are at sufficient concentrations to cause adverse alterations to benthic invertebrate communities. Sediment HQs for total PCBs were 100 times greater than any other COC. Elevated levels of surface sediment PCBs are found ubiquitously throughout Zone 1 (see Figure 2-5). Dieldrin and p,p'-DDT are not at sufficient concentrations to pose risk.

For the Assessment Endpoint—Benthic Fish Reproduction and Survival. *Are levels of site contaminants sufficient to cause survival or reproductive impairment to benthic fish?*

Benthic fish are discussed under Zone 2, below.

For the Assessment Endpoint—Pelagial Fish Reproduction and Survival. *Are levels of site contaminants sufficient to cause survival or reproductive impairment to pelagial fish?*

Pelagial fish are discussed under Zone 2, below.

For the Assessment Endpoint—Insectivorous Bird Reproduction and Survival. *Are levels of site contaminants sufficient to cause survival or reproductive impairment or deformity in insectivorous birds?*

Only total PCBs were found to be at sufficient concentrations to potentially cause survival or reproductive impairment in insectivorous birds. This included HQs for reproductive success, as well as for potential deformity in hatchlings. There are no potential risk effects attributable to the chlorinated pesticides (dieldrin, o,p'-DDD, o,p'-DDE, o,p'-DDT, p,p'-DDD, p,p'-DDE, p,p'-DDT).

For the Assessment Endpoint—Piscivorous Bird Reproduction and Survival. *Are levels of site contaminants sufficient to cause survival or reproductive impairment or deformity in piscivorous birds?*

Risks to piscivorous birds are discussed under Zone 2, below.

For the Assessment Endpoint—Carnivorous Bird Reproduction and Survival. *Are levels of site contaminants sufficient to cause survival or reproductive impairment or deformity in carnivorous birds?*

Risks to carnivorous birds are discussed under Zone 2, below.

For the Assessment Endpoint—Piscivorous Mammal Reproduction and Survival. *Are levels of site contaminants sufficient to cause survival or reproductive impairment to piscivorous mammals?*

Modeled concentrations of total PCBs in the diet of piscivorous mammals are sufficient to cause adverse effects on survival or reproduction. Estimated concentrations of mercury and the chlorinated pesticides (dieldrin and p,p'-DDE) are not sufficient to pose risk.

Green Bay Zone 1 Summary. In summary, the results taken in total suggest that measured or estimated concentrations of total PCBs are at sufficient levels to cause risk to benthic invertebrates and piscivorous mammals. Total PCBs are at sufficient levels to potentially cause risk to aquatic invertebrates and insectivorous birds. Concentrations of dieldrin, all o,p'- isomers of DDT and its metabolites, and p,p'-DDT are not sufficient to pose risk to any of the evaluated assessment endpoints. Risks to fish and birds are discussed below.

Green Bay Zone 2

A summary of all RME HQs for Green Bay Zone 2 is presented in Table 6-125, along with the relevant assessment endpoints and risk questions. Figures 6-80a, 6-80b, and 6-80c present HQs that exceeded 1.0 in Green Bay Zone 2. Summaries of risks to fish and piscivorous birds exposed in zones 1 and 2 of Green Bay are presented and discussed here. Assessment endpoints, risk questions, and determined risk are summarized below.

For the Assessment Endpoint—Functioning Water Column Invertebrate Communities. *Are levels of site contaminants in surface water sufficient to cause adverse alterations to the functioning of water column invertebrate communities?*

There may be risks to functioning water column invertebrate communities in Zone 2 from exposure to mercury. Based on calculated HQs, only mercury is at concentrations that are posing risk to the functioning of water column invertebrate communities. Lead and total PCBs are not at sufficient concentrations to pose risk and the remaining COPCs were not detected in this reach.

For the Assessment Endpoint—Functioning Benthic Invertebrate Communities. *Are levels of site contaminants in sediment sufficient to cause adverse alterations to the functioning of benthic invertebrate communities?*

There are persistent risks to benthic infaunal communities from COCs in sediments of Zone 2. Based on the calculated HQs, concentrations of mercury and total PCBs in the sediment are at sufficient concentrations to cause adverse alterations to benthic invertebrate communities. Sediment HQs for total PCBs are three times greater than the HQ for mercury. Arsenic, lead, dieldrin, p,p'-DDD, p,p'-DDE, and p,p'-DDT are not at sufficient concentrations to pose risk.

For the Assessment Endpoint—Benthic Fish Reproduction and Survival. *Are levels of site contaminants sufficient to cause survival or reproductive impairment to benthic fish?*

Zone 1 and Zone 2 fish are considered to be a single population that freely migrate between the two zones, and thus were evaluated as a single exposure unit for the purposes of characterizing risk. Based upon HQs alone, concentrations of total PCBs and p,p'-DDE in benthic fish are sufficiently high to potentially be of risk to benthic fish reproduction or survival. For both compounds, the calculated NOAEC HQs exceeded 1.0 (9.7 and 2.3, respectively), but was less than 1.0 for the LOAEC. For PCBs, however, it is noted that TEQs calculated from measured dioxins and coplanar congeners in carp tissues are not at levels sufficiently high to represent risk, even though TEQs measured in these fish are an order of magnitude higher than any of the reaches within the river. Mercury and the other chlorinated pesticides (dieldrin, o,p'-DDD, o,p'-DDE, o,p'-DDT, p,p'-DDD, p,p'-DDT) are not at sufficient concentrations to pose risk.

For the Assessment Endpoint—Pelagial Fish Reproduction and Survival. *Are levels of site contaminants sufficient to cause survival or reproductive impairment to pelagial fish?*

Zone 1 and Zone 2 fish are considered to be a single population that freely migrate between the two zones, and thus were evaluated as a single exposure unit for the purposes of characterizing risk. The derived hazard quotients for reproduction suggest that concentrations of total PCBs in all of the pelagial species examined are at levels that potentially could impact reproduction or survival. All of the NOAEC HQs for alewife, gizzard shad, smelt, perch, walleye and both shiner species are between 1.5 and 10, but are consistently less than 1.0 for the LOAEC. However, when examined on a TEQ basis, only the walleye NOAEC HQ exceeded 1.0. Concentrations of mercury and p,p'-DDE produced a NOAEC HQ greater than 1.0 only for walleye (HQ = 1.5); all other species were below 1.0.

For the Assessment Endpoint—Insectivorous Bird Reproduction and Survival. *Are levels of site contaminants sufficient to cause survival or reproductive impairment or deformity in insectivorous birds?*

Based upon evaluation of the hazard quotients alone, only concentrations of total PCBs, PCB congeners, and p,p'-DDE are at sufficient levels to potentially cause survival or reproductive impairment in insectivorous birds. This included HQs for reproductive success, as well as for potential deformity in hatchlings. There are no potential risk effects attributable to the chlorinated pesticides other than p,p'-DDE (dieldrin, o,p'-DDD, o,p'-DDE, o,p'-DDT, p,p'-DDD, p,p'-DDT).

For the Assessment Endpoint—Piscivorous Bird Reproduction and Survival. *Are levels of site contaminants sufficient to cause survival or reproductive impairment or deformity in piscivorous birds?*

Piscivorous birds, while principally nesting in Zone 2, were considered to have opportunity to take fish in both zones and thus were evaluated as a single exposure unit for the purposes of characterizing risk.

Modeled intake and measured concentrations of contaminants show potentials for risk in piscivorous birds. Modeled dietary intake of mercury, total PCBs, and p,p'-DDE in the diet of piscivorous birds are estimated to be sufficiently high to cause potential adverse effects on survival, physiology, and reproduction. This conclusion is supported by measured concentrations of PCBs and DDE in eggs and whole bodies that are above toxicity reference values. Furthermore, measured concentrations of PCB congeners and dieldrin in piscivorous birds are sufficient to cause potential adverse effects on survival, physiology, or reproduction. It is noted that while the modeled dietary intake of dieldrin yielded an HQ below 1.0 (NOAEC HQ = 0.3 for common tern), the measured concentrations in tern and cormorant eggs yielded elevated HQs (NOAEC HQs = 0.6 to 4.4). These results indicate that the birds are consuming fish containing higher concentrations of dieldrin than the exposure modeling would predict. Measured concentrations of the chlorinated pesticides (o,p'-DDD, o,p'-DDE, o,p'-DDT, p,p'-DDD, p,p'-DDT) are not sufficient to pose risk.

For the Assessment Endpoint—Carnivorous Bird Reproduction and Survival. *Are levels of site contaminants sufficient to cause survival or reproductive impairment or deformity in carnivorous birds?*

Carnivorous birds were considered to have the opportunity to take fish in both zones, and thus were evaluated as a single exposure unit for the purposes of characterizing risk.

Modeled dietary intake of mercury, total PCBs and p,p'-DDE in the diet of carnivorous birds are estimated to be at sufficient concentrations to cause potential adverse effects on survival, physiology, or reproduction. The resultant modeled HQs for the NOAEC are 1.6, 7.5 and 4.1, respectively, while the LOAEC HQs are less than 1.0. Estimated intake of dieldrin is not at sufficient concentrations to pose risk.

For the Assessment Endpoint—Piscivorous Mammal Reproduction and Survival. *Are levels of site contaminants sufficient to cause survival or reproductive impairment to piscivorous mammals?*

Modeled concentrations of total PCBs in the diet of piscivorous mammals are estimated to be at sufficient concentrations to cause adverse effects on survival or reproduction. Estimated concentrations of mercury and the chlorinated pesticides (dieldrin and p,p'-DDE) are not at sufficient concentrations to pose risk.

Green Bay Zone 2 Summary. In summary, the results taken in total suggest that measured or estimated concentrations of total PCBs are at sufficient levels to cause risks to benthic invertebrates, carnivorous birds, and piscivorous mammals. Potential risks are indicated for benthic and pelagial fish, and piscivorous birds. Measured or estimated concentrations of mercury are at sufficient concentrations to cause or potentially cause risk to aquatic invertebrates, benthic invertebrates, pelagial fish, piscivorous birds, and carnivorous birds. Measured of estimated concentrations of DDE are at sufficient concentrations to cause, or potentially cause, risk to benthic fish, pelagic fish, insectivorous birds, piscivorous birds, and carnivorous birds.

Green Bay Zone 3A

A summary of all RME HQs for Green Bay Zone 3A is presented in Table 6-126, along with the relevant assessment endpoints and risk questions. Figure 6-81 presents HQs that exceeded 1.0 in Green Bay Zone 3A. Assessment endpoints, risk questions, and determined risk are summarized below.

For the Assessment Endpoint—Functioning Water Column Invertebrate Communities. *Are levels of site contaminants in surface water sufficient to cause adverse alterations to the functioning of water column invertebrate communities?*

Neither mercury or total PCB concentrations are sufficient to pose risk to water column invertebrate communities, based on calculated HQs that are all 0.1 or less. These were the only two COPCs analyzed in this zone.

For the Assessment Endpoint—Functioning Benthic Invertebrate Communities. *Are levels of site contaminants in sediment sufficient to cause adverse alterations to the functioning of benthic invertebrate communities?*

Based on the calculated HQs, concentrations of total PCBs in the sediment are at sufficient concentrations to cause adverse alterations to benthic invertebrate communities, but concentrations of arsenic, lead, mercury, dieldrin, p,p'-DDD, p,p'-DDE, and p,p'-DDT are not at sufficient concentrations to pose risk.

For the Assessment Endpoint—Benthic Fish Reproduction and Survival. *Are levels of site contaminants sufficient to cause survival or reproductive impairment to benthic fish?*

Calculated HQs suggest that concentrations of total PCBs in benthic fish are sufficiently high to potentially be of risk to benthic fish reproduction or survival, and that concentrations of mercury, PCB congeners, and the chlorinated pesticides (dieldrin, p,p'-DDD, p,p'-DDE, p,p'-DDT) are not sufficient to pose risk.

For the Assessment Endpoint—Pelagial Fish Reproduction and Survival. *Are levels of site contaminants sufficient to cause survival or reproductive impairment to pelagial fish?*

Concentrations of total PCBs in pelagial fish are sufficiently high to potentially be of risk to pelagial fish reproduction or survival. Concentrations of mercury, PCB congeners, and the chlorinated pesticides (dieldrin, o,p'-DDD, o,p'-DDE, o,p'-DDT, p,p'-DDD, p,p'-DDE, p,p'-DDT) are not sufficient to pose risk. As indicated, the PCB risk conclusions are not in agreement for total PCBs and PCB congeners. Additionally, when looked at on a species-by-species basis, HQs for rainbow smelt are 1.0 or less, HQs for alewife are between 1.0 and 2.0, HQs for brown trout are between 4.0 and 5.0, and HQs for walleye are between 5.0 and 7.0.

For the Assessment Endpoint—Insectivorous Bird Reproduction and Survival. *Are levels of site contaminants sufficient to cause survival or reproductive impairment or deformity in insectivorous birds?*

Data were not available for the estimation of risk to insectivorous birds.

For the Assessment Endpoint—Piscivorous Bird Reproduction and Survival. *Are levels of site contaminants sufficient to cause survival or reproductive impairment or deformity in piscivorous birds?*

Estimated concentrations of mercury and total PCBs in the diet of piscivorous birds are at sufficient concentrations to cause potential adverse effects on survival, physiology, or reproduction. The estimated concentrations of dieldrin and p,p'-DDE are not sufficient to pose risk. The only species-specific difference noted was regarding the HQs for mercury, where common tern and Forster's tern are at potential risk, but double-crested cormorants are not. Even in the terns, however, risk from mercury is estimated to be approximately half of the risk posed by total PCBs.

For the Assessment Endpoint—Carnivorous Bird Reproduction and Survival. *Are levels of site contaminants sufficient to cause survival or reproductive impairment or deformity in carnivorous birds?*

HQs for bald eagles collected in this zone indicate that carnivorous birds are at risk for reproductive impairment or deformity from total PCBs, and are potentially at risk from dieldrin. HQs based on estimated concentrations of total PCBs in the diet of carnivorous birds are sufficient to cause potential adverse effects on survival, physiology, or reproduction.

For the Assessment Endpoint—Piscivorous Mammal Reproduction and Survival. *Are levels of site contaminants sufficient to cause survival or reproductive impairment to piscivorous mammals?*

Estimated concentrations of total PCBs in the diet of piscivorous mammals are sufficient to cause adverse effects on survival or reproduction. Estimated concentrations of dieldrin in the diet of piscivorous mammals are sufficient to cause potential adverse effects on survival or reproduction. Estimated concentrations of mercury and p,p'-DDE are not sufficient to pose risk.

Green Bay Zone 3A Summary. In summary, the results taken in total suggest that concentrations of total PCBs are at sufficient levels to cause, or potentially cause, risk to benthic invertebrates, benthic fish, pelagic fish, piscivorous birds, carnivorous birds, and piscivorous mammals. There were no data to evaluate insectivorous birds. Mercury concentrations are potentially causing risk to piscivorous birds. Concentrations of dieldrin are a potential risk for carnivorous birds and piscivorous mammals. Concentrations of arsenic, lead, and all o,p'- and p,p'- isomers of DDT and its metabolites were not found to pose risk to any assessment endpoint.

Green Bay Zone 3B

A summary of all RME HQs for Green Bay Zone 3B is presented in Table 6-127, along with the relevant assessment endpoints and risk questions. Figure 6-82 presents HQs that exceeded 1.0 in Green Bay Zone 3B. Assessment endpoints, risk questions, and determined risk are summarized below.

For the Assessment Endpoint—Functioning Water Column Invertebrate Communities. *Are levels of site contaminants in surface water sufficient to cause adverse alterations to the functioning of water column invertebrate communities?*

Neither mercury nor total PCB concentrations are sufficient to pose risk to water column invertebrate communities, based on calculated HQs that were all 0.1 or less. Although these were the only two COPCs analyzed in this zone, analyses in the other river reaches and Green Bay zones suggest that these are the COPCs that are the most likely to have HQs of greater than 1.0.

For the Assessment Endpoint—Functioning Benthic Invertebrate Communities. *Are levels of site contaminants in sediment sufficient to cause adverse alterations to the functioning of benthic invertebrate communities?*

Based on the calculated HQs, concentrations of arsenic, lead, mercury, and total PCBs in the sediment are at sufficient concentrations to cause adverse alterations to benthic invertebrate communities, but concentrations of dieldrin, p,p'-DDD, p,p'-DDE, and p,p'-DDT are not at sufficient concentrations to pose risk. Additionally, the HQs for PCBs are at least 10 times greater than the HQs for other COCs. Benthic invertebrate community investigations of Green Bay zones 1 and 2 indicate that benthic invertebrate communities are impacted by COCs and ammonia. Therefore, these data suggest that site contaminants in sediment are sufficient to cause adverse alterations to the functioning of benthic invertebrate communities.

For the Assessment Endpoint—Benthic Fish Reproduction and Survival. *Are levels of site contaminants sufficient to cause survival or reproductive impairment to benthic fish?*

Calculated HQs suggest that concentrations of mercury, total PCBs, PCB congeners, and the chlorinated pesticides (dieldrin, p,p'-DDD, p,p'-DDE, p,p'-DDT) are not sufficient to pose risk.

For the Assessment Endpoint—Pelagial Fish Reproduction and Survival. *Are levels of site contaminants sufficient to cause survival or reproductive impairment to pelagial fish?*

Based on a summary of the HQs for pelagial fish, concentrations of total PCBs may be sufficient to cause survival or reproductive impairment, and concentrations of mercury and p,p'-DDE in pelagial fish are sufficient to potentially be of risk, but concentrations of PCB congeners and the chlorinated pesticides (dieldrin, p,p'-DDD, p,p'-DDT) are not at sufficient concentrations to pose risk. Total PCB HQs for alewife, rainbow smelt, and brown trout suggest that there is potential risk to survival or reproduction impairment. The only fish for which total PCB HQs suggested risk was for walleye, and this conclusion is limited by the fact that the RME LOAEC HQ was 1.5, and therefore, did not greatly exceed 1.0.

For the Assessment Endpoint—Insectivorous Bird Reproduction and Survival. *Are levels of site contaminants sufficient to cause survival or reproductive impairment or deformity in insectivorous birds?*

Data were not available for the estimation of risk to insectivorous birds.

For the Assessment Endpoint—Piscivorous Bird Reproduction and Survival. *Are levels of site contaminants sufficient to cause survival or reproductive impairment or deformity in piscivorous birds?*

Measured HQs in whole double-crested cormorants indicated that piscivorous birds are at risk for reproductive impairment or deformity from total PCBs and are at potential risk from dieldrin and p,p'-DDE. Estimated HQs suggest that piscivorous birds are at potential risk from mercury, total PCBs, and p,p'-DDE. The only species-specific difference noted was regarding the HQs for p,p'-DDE, where common tern and Forster's tern are at potential risk, but double-crested cormorants are not. This is also the case for mercury in double-crested cormorants, where the NOAEC HQ (1.2) barely exceeded 1.0.

For the Assessment Endpoint—Carnivorous Bird Reproduction and Survival. *Are levels of site contaminants sufficient to cause survival or reproductive impairment or deformity in carnivorous birds?*

Estimated concentrations of mercury, total PCBs, and p,p'-DDE in the diet of carnivorous birds are sufficient to cause potential adverse effects on survival, physiology, or reproduction. Estimated concentrations of dieldrin are not sufficient to pose risk. There are no measured HQs for carnivorous birds in this area, but these results concur with the results for piscivorous birds. Because of the

federal status of the bald eagle, carnivorous birds are assumed to be at risk from these COCs.

For the Assessment Endpoint—Piscivorous Mammal Reproduction and Survival. *Are levels of site contaminants sufficient to cause survival or reproductive impairment to piscivorous mammals?*

Estimated concentrations of total PCBs in the diet of piscivorous mammals are sufficient to cause adverse effects to survival or reproduction. Estimated concentrations of dieldrin in the diet of piscivorous mammals are sufficient to cause potential adverse effects on survival or reproduction. Estimated concentrations of mercury and p,p'-DDE are not sufficient to pose risk.

Green Bay Zone 3B Summary. In summary, the results taken in total suggest that measured or estimated concentrations of total PCBs are at sufficient levels to cause, or potentially cause, risk to benthic invertebrates, pelagial fish, piscivorous birds, carnivorous birds, and piscivorous mammals. There are no data to evaluate insectivorous birds. Mercury concentrations are causing or potentially causing risk to benthic invertebrates, pelagial fish, piscivorous birds, and carnivorous birds. DDE concentrations are causing, or potentially causing, risk to pelagial fish, piscivorous birds, and carnivorous birds. Dieldrin concentrations are potentially causing risk to piscivorous mammals. Arsenic and lead concentrations are only of risk to benthic invertebrates.

Green Bay Zone 4

A summary of all RME HQs for Green Bay Zone 4 is presented in Table 6-128, along with the relevant assessment endpoints and risk questions. Figure 6-83 presents HQs that exceeded 1.0 for Green Bay Zone 4. Assessment endpoints, risk questions, and determined risk are summarized below.

For the Assessment Endpoint—Functioning Water Column Invertebrate Communities. *Are levels of site contaminants in surface water sufficient to cause adverse alterations to the functioning of water column invertebrate communities?*

Neither mercury nor total PCB concentrations are sufficient to pose risk to water column invertebrate communities, based on calculated HQs that were all 0.1 or less. Mercury was not detected in surface water and results for PCB concentrations are approximately one-third the water concentrations measured in Green Bay Zone 3A. Given that surface water concentrations in Zone 4 are the lowest that have been measured in the bay, it suggests that there is no risk to water column invertebrates in other areas of the bay, and adverse alterations to the functioning water column invertebrate communities are not expected.

For the Assessment Endpoint—Functioning Benthic Invertebrate Communities. *Are levels of site contaminants in sediment sufficient to cause adverse alterations to the functioning of benthic invertebrate communities?*

Based on the calculated HQs, concentrations of total PCBs in the sediment are at sufficient concentrations to cause adverse alterations to benthic invertebrate communities, but concentrations of arsenic, lead, mercury, dieldrin, p,p'-DDD, p,p'-DDE, and p,p'-DDT are not at sufficient concentrations to pose risk. Benthic invertebrate community investigations have not been conducted in this zone of Green Bay or in the adjacent Zone 3. HQs for total PCBs are 3.7 (non-interpolated) and 1.4 (I_d interpolated). Overall, these data suggest that site contaminants in sediment are sufficient to cause adverse alterations to the functioning of benthic invertebrate communities.

For the Assessment Endpoint—Benthic Fish Reproduction and Survival. *Are levels of site contaminants sufficient to cause survival or reproductive impairment to benthic fish?*

Data were not available for the estimation of risk to benthic fish.

For the Assessment Endpoint—Pelagial Fish Reproduction and Survival. *Are levels of site contaminants sufficient to cause survival or reproductive impairment to pelagial fish?*

Based on a summary of the HQs for pelagial fish, concentrations of total PCBs and p,p'-DDE in pelagial fish are sufficiently high to potentially be of risk to pelagial fish reproduction or survival, while concentrations of PCB congeners and chlorinated pesticides (dieldrin, p,p'-DDD, p,p'-DDT) are not at sufficient concentrations to pose risk. In terms of specific fish species, rainbow smelt are not at risk from total PCBs, HQs for alewife were between 1.0 and 2.0, and HQs for brown trout and walleye are between 3.0 and 5.0 for total PCBs. Walleye was the only fish analyzed for p,p'-DDE and the resulting HQs were between 1.0 and 2.0.

For the Assessment Endpoint—Insectivorous Bird Reproduction and Survival. *Are levels of site contaminants sufficient to cause survival or reproductive impairment or deformity in insectivorous birds?*

Data were not available for the estimation of risk to insectivorous birds.

For the Assessment Endpoint—Piscivorous Bird Reproduction and Survival. *Are levels of site contaminants sufficient to cause survival or reproductive impairment or deformity in piscivorous birds?*

Estimated concentrations of mercury and total PCBs in the diet of piscivorous birds are sufficient to cause potential adverse effects to survival, physiology, or reproduction. The estimated concentrations of dieldrin and p,p'-DDE are not sufficient to pose risk. The only species-specific difference noted was regarding the HQs for mercury, where common tern and Forster's tern are at potential risk, but double-crested cormorants are not. There are no measured HQs for this zone of Green Bay. For this zone it is assumed that mercury and total PCBs are at sufficient concentrations to cause survival or reproductive impairment in piscivorous birds.

For the Assessment Endpoint—Carnivorous Bird Reproduction and Survival. *Are levels of site contaminants sufficient to cause survival or reproductive impairment or deformity in carnivorous birds?*

Estimated concentrations of mercury, total PCBs, and p,p'-DDE in the diet of carnivorous birds are at sufficient concentrations to cause potential adverse effects on survival, physiology, or reproduction. Estimated concentrations of dieldrin are not at sufficient concentrations to pose risk. There are no measured HQs for this zone of Green Bay. For this zone it is assumed that mercury, total PCBs, and p,p'-DDE are at sufficient concentrations to cause survival or reproductive impairment in carnivorous birds.

For the Assessment Endpoint—Piscivorous Mammal Reproduction and Survival. *Are levels of site contaminants sufficient to cause survival or reproductive impairment to piscivorous mammals?*

Estimated concentrations of total PCBs in the diet of piscivorous mammals are sufficient to cause adverse effects to survival or reproduction. Estimated concentrations of mercury and the chlorinated pesticides (dieldrin and p,p'-DDE) are not sufficient to pose risk.

Green Bay Zone 4 Summary. In summary, these results taken in total suggest that concentrations of total PCBs are at sufficient levels to cause, or potentially cause risk to benthic invertebrates, pelagial fish, piscivorous birds, carnivorous birds, and piscivorous mammals. Concentrations of DDE are causing or potentially causing risk to pelagial fish and carnivorous birds. Concentrations of mercury are causing or potentially causing risk to piscivorous and carnivorous birds.

6.5.4 Field Studies in the Lower Fox River and Green Bay

The previous section summarized risks to receptor species based upon use of hazard quotients, alone. Within the Lower Fox River and Green Bay system, there have been numerous field studies on these same receptor species. While not specifically included in the risk characterization, the studies are presented here, and with the risk characterization and uncertainty (Section 6.6) to provide the risk managers with an integrated tool for decision making (Section 6.7).

Invertebrates

Water Column. There have been no direct studies relating the COPCs to effects on functioning water column communities. Ankley *et al.* (1990) report on the results of an integrated assessment of the biota of zones 1 and 2 conducted in 1988 that included acute and chronic bioassays on the water column alga *Selenastrum capricornutum*, the invertebrate *Ceriodaphnia dubia*, the bacterium *Photobacterium phosphoreum*, and the fathead minnow (*Pimephales promelas*). While these bioassays were conducted using pore water extracted from sediments, the results have relevance to the BLRA as the extracted water would represent a "worst case" exposure scenario.

Ten sediment samples were collected from Zone 1, two from within Zone 2, and an additional reference sample from the upper East River. During the first round of exposure, all extracted pore water from Green Bay was acutely toxic to the fathead minnows, daphnids, and some of the *Selenastrum* stations, but extracts from the East River showed no mortality. All Green Bay extracts also induced chronic toxicity to daphnids. There were no effects to the bacteria.

Acute mortality from exposure to Green Bay pore water extracts were found to be largely due to the presence of ammonia in the sediments. Ankley and colleagues (1990) were able to demonstrate through a Toxicity Identification Evaluation (TIE) that much of the acute toxicity observed was due to the elevated levels of ammonia in the pore water extracts. Hoke *et al.* (1992) questioned attributing all of the acute toxicity characteristics to ammonia, considering that some metals in sediments would be effected by the same TIE procedures, and that metals and organic chemical analyses were not done on the tested sediments. Their work, however, did confirm that at least 10 to 50 percent of the observed toxicity could be accounted for by ammonia. It should be noted that neither study addressed chronic toxicity on the TIE-treated pore water, leaving unanswered the question of whether there are long-term chronic effects from other COPCs present in water samples.

Benthos. Quantitative benthic invertebrate studies have been conducted throughout the Lower Fox River and Green Bay since 1964 (IPS, 1994; WDNR, 1996a, 1996b).

In general, benthic infaunal communities throughout the river and bay are showing increases in diversity and presence of "pollution-intolerant" species (IPS, 1994) in some locations. In the 1960s, most of the soft sediment was generally devoid of infaunal communities. Most stations had both low diversity and low number of species. Through the 1970s and early 80s, an increased number of organisms were observed, but through the late 1980s the soft sediment communities were dominated by oligochaetes and chironomids. Call *et al.* (1991) conducted infaunal analyses in 1988 and 1989 at the same stations within Zone 1 and Zone 2 that Ankley *et al.* (1990) conducted sediment and pore water bioassays. Their results showed that benthic invertebrates at those stations, including the East River reference site, were primarily oligochaetes and chironomids, but that the total number of organisms collected were one to two orders of magnitude lower than the reference site.

Sediment bioassays have also demonstrated the presence of stressors within sediments of the river. Call *et al.* (1991) also conducted bioassays on bulk sediments collected from the same locations as the infaunal samples, using the amphipod *H. azteca*, the oligochaete *Lumbriculus variegatus*, the chironomid *Chironomus riparius*, the mayfly *Hexagenia limbata*, and the fathead minnow *P. promelas*. While initially high toxicity was observed for the mayfly, the TIE work of Ankley *et al.* (1990) had suggested the ammonia was responsible for much of the toxicity observed. As noted, Hoke *et al.* (1992) suggested that the effect was limited to 10 to 50 percent of the observed toxicity. Call *et al.* (1991) also measured uptake of COPCs from the sediments in tested species and found that, with the exception of PCBs, accumulation of other non-polar organics (e.g., polynuclear aromatic hydrocarbons) was minimal. Dioxins and furans were not detected in any of the bioassay tissue analyses.

More recent samples at some stations within the river have shown increased numbers of benthic invertebrates and increased diversity. For example, samples collected from Deposit POG in Little Lake Butte des Morts in 1994 remained principally dominated by chironomids and oligochaetes, but also showed the presence of flatworms, sow bugs, amphipods (*Hyallela azteca*), clams (*Pisidium*) and physid snails that had previously not been observed. However, this was only observed within Little Lake Butte des Morts; the remaining stations through the river remain low in diversity (IPS, 1994).

Infaunal populations within Green Bay are varied, reflecting both the physical diversity of the bay (e.g., depth, substrate) as well as influences from Fox River-related stressors. Infaunal populations in Zone 2, and in the southernmost end of Zone 3 (A & B) remain largely dominated by oligochaetes and chironomids (IPS, 1994; Call *et al.*, 1991). Communities south of a line drawn between the

Peshtigo Reef and Sturgeon Bay have greater species richness and total numbers of organisms than in the inner bay. However, Zone 3A is reported to be more diverse than Zone 3B, suggesting higher water and sediment quality (IPS, 1994) on the west side of the bay. Gammarid amphipods (*Gammarus fasciatus* and *Pontoporeia hoyi*) first appear in abundance in this zone, and are principally absent from the inner bay. The historical data suggest that since the 1980s, this area of the bay has remained relatively consistent in terms of diversity and total species (IPS, 1994).

The course of recovery of benthic infaunal populations has been altered by the introduction of zebra mussel (*Dreissena polymorpha*). The first reported observation was in 1994 in Green Bay at Menominee, Michigan, and by the fall zebra mussels were widespread along the western Door Peninsula (UWSGI, 1994). By 1996, they were reported in Lake Winnebago (UWSGI, 1996), and now are found throughout the entire Fox River system.

These data, taken in total, suggest that the benthic communities within the Lower Fox River and Green Bay, while showing some improvement, remain impaired.

Fish

Benthic Fish. As noted in Section 2, benthic fish species were present throughout the Lower Fox River, even during the 1960s when water quality was generally poor, and generally were thought to be self-reproducing populations. While the fish in Little Lake Butte des Morts, or within Zone 1, could have been recruits from Lake Winnebago or Zone 2, respectively, the presence of abundant carp and bullheads in the middle two reaches supports the position that those fish have been successfully breeding and recruiting even in the presence of the COPCs.

As noted previously, Call *et al.* (1991) conducted bioassays on the cyprinid *P. promelas* with both bulk sediments and pore water from samples collected in Zone 1, and found no acute or chronic toxicity beyond that which was attributable to ammonia. EPA (Ankley *et al.*, 1992) conducted additional long-term (45-day exposure followed by 6-month grow-out) sediment testing on Medaka (*Oryzias latipes*), using sediments collected in Zone 1, and found neither acute nor chronic toxicity.

While recruitment may have been occurring, resident fish were exposed to potential carcinogens that could have resulted in sublethal effects. USFWS personnel had observed an abnormally high incidence of tumors in the brown bullhead (*Ictalurus nebulosus*) collected from the Lower Fox River (P. Bauman, cited in Ankley *et al.*, 1992). To corroborate that observation, EPA undertook sampling of bullhead from Zone 1 for observation and histopathological examination of the

liver. Fish were collected in 1988, and a total of 16 brown bullhead, 118 black bullhead (*I. melas*), and eight yellow bullhead (*I. natalis*) were examined for gross external abnormalities and for liver lesions (Johnson *et al.*, 1992, cited in Ankley *et al.*, 1992). There was a complete absence of either pre-neoplastic or neoplastic lesions in the livers of any of the fish collected. In addition, EPA conducted Salmonella mutagenicity tests using Zone 1 sediments, and found little mutagenic activity in any of the samples. EPA concluded that there was little potential for mutagenicity to fish in the Lower Fox River (Ankley *et al.*, 1992).

These results, taken collectively, suggest that the COPCs present in Lower Fox River and Green Bay sediments were not having adverse effects on the ability of benthic fish to reproduce and recruit.

Piscivorous Fish

Walleye. The decline and later recovery of breeding walleye populations within the Lower Fox River was discussed in Section 2. Walleye are now found in all reaches of the river and bay.

While successful recruitment may be occurring, resident fish are exposed to potential carcinogens that could result in sublethal effects. As part of the NRDA evaluation, the USFWS undertook an evaluation of injuries to walleye (Barron *et al.*, 1999). Walleye were collected from all five zones of the bay and two reference areas (Lake Winnebago and Patten Lake) in 1996, and were measured for levels of total PCBs and PCB congeners. Tissue PCB concentrations were significantly higher in assessment area walleye than in fish collected from the reference areas. Fish health was further assessed using a suite of tests designed to measure parameters that can be adversely affected by PCB exposure. These included examination of tissues for bacterial, viral, and parasitic infections, immunological evaluation of kidney and blood samples, evaluation of liver lesions, and measurement of ethoxyresorufin-O-deethylase (EROD) activity and tissue PCB concentration. Fish in Green Bay also had a significantly higher incidence of liver tumors and pre-tumors. It has been documented that PCBs promote or enhance liver tumor formation (Hendricks *et al.*, 1990); therefore the injury report concluded walleye health has been adversely impacted by PCB exposure.

Lake Trout. As previously discussed in Section 6.3.2, PCBs have been one of the suspected causes of recruitment failure in lake trout (*Salvelinus namacyacush*), which have experienced significant early life-stage mortality in contaminated regions of the Great Lakes (Mac *et al.*, 1985, 1993; Mac, 1988). As part of the NRDA, the USFWS also looked at potential effects on lake trout reproduction. Adverse effects on reproduction were assessed for lake trout based on historical data, information from the scientific literature, and reproduction and laboratory

toxicity studies conducted for the NRDA by the United States Geological Survey (USGS). The toxicity equivalence approach was used to compare historic PCB concentrations in lake trout eggs with toxicity thresholds for embryo mortality. Mean egg total PCB concentrations over time were modeled and compared with LD_{10} and LD_{50} concentrations. The analysis concluded that in the mid-1970s, egg PCB concentrations were sufficient to cause sac fry mortality to some Green Bay lake trout eggs; by 1980, concentrations in less than 1 percent of Lake Michigan lake trout eggs are estimated to have been sufficient to cause mortality. Limited PCB data were available for Green Bay and western Lake Michigan lake trout; analysis of these data suggest PCB concentrations were higher in Green Bay lake trout.

Results of the toxicity studies conducted by the USGS for the NRDA suggested that thiamine deficiency, rather than exposure to PCBs or other TCDD-like compounds, is currently the primary causal factor for fry mortality in Lake Michigan lake trout. The NRDA report determined that current data do not support concluding that lake trout in Green Bay and Lake Michigan are injured by the PCBs in the Lower Fox River or Green Bay

These data, taken in total, do not support a conclusion that PCBs are having significant adverse effects on lake trout health and reproduction, although they may have in the past.

Birds

Tree Swallows. Tree swallows exist throughout the Lower Fox River and Green Bay, and were first examined for uptake of PCBs and other COPCs during the Green Bay Mass Balance study (Ankley *et al.*, 1993). That work documented the accumulation of total and specific planar PCBs, PCDFs, and PCDDs, as well as derived TCDD-Eqs using the H4IIE rat hepatoma bioassay in tree swallows, red-winged blackbirds, Forster's tern, and common terns. The study areas for the tree swallow and red-winged blackbirds were along the river below De Pere, and along the western shore within Zone 2. For the tern species, chicks were collected from the Kidney Island CDF. The concentrations of the PCBs were greatest in eggs and chicks of the two tern species, less in the tree swallows, and least in the red-winged blackbirds. The results from the field studies suggested apparent adverse reproductive effects in the red-winged blackbirds and in Forster's terns (Ankley *et al.*, 1992).

A more thorough examination of potential effects on tree swallows was undertaken by Custer *et al.* (1998) in 1994 and 1995, as previously discussed in Section 6.2.5. The study was designed specifically to examine the accumulation, concentration, and effects of PCBs on reproduction in tree swallows from Little

Lake Butte des Morts Reach (Arrowhead Park) and from Zone 2 (Kidney Island). Custer and colleagues examined the total PCBs in eggs, newly hatched young, and 12 day-old nestlings from nesting boxes placed at Arrowhead Park and on the Kidney Island CDF, and compared these sites to two reference sites at Lake Poygan and High Cliff State Park. The authors found that total PCBs, congeners, dioxins, and furans were similar to concentrations found in tree swallows in 1988, and did not seem to have declined. Clutch size was found to be unaffected by PCB concentration, and was considered to be normal for tree swallows nesting elsewhere. PCB levels in eggs did not affect hatching success at either of the contaminated sites. While total concentrations and rates of accumulations of PCBs and DDE were found to be elevated at both Arrowhead and Kidney Island (relative to the reference sites), those concentrations were not significantly different among clutches where all eggs hatched, some eggs hatched, or no eggs hatched.

These data suggest that there are no population level effects to reproducing tree swallows in the Lower Fox River and Green Bay.

Terns. As noted in Section 2, the status of reproducing tern populations in Green Bay is indeterminate at this time (Figure 2-22). Forster's, common, and Caspian terns remain listed as endangered species in Wisconsin, while Michigan lists the Forster's and common terns as threatened species. The numbers of observed breeding pairs has consistently risen since the 1970s, but there is insufficient information at this time to state that these species have fully recovered.

The potential toxic effects of PCBs on tern reproduction has been well studied throughout the Lower Fox River and Green Bay. Kubiak *et al.* (1989) documented significant impairment of Forster's tern from exposure to chlorinated hydrocarbons in Green Bay in 1983 from eggs collected just south of Long Tail Point (Zone 2), within the South Oconto Marsh (Zone 3A), and compared them with eggs collected from Lake Poygon (reference station). Eggs collected from the Green Bay sites had significantly higher PCB, TCDD, and other PCDD concentrations relative to the reference site. Hatch success was found to be 50 to 75 percent lower in the Green Bay eggs, and was correlated with PCB concentrations.

As noted above, EPA examined the effects of PCBs, TCDD, and TCDF on Forster's tern and common terns from Kidney Island in Zone 2 in 1988, and found apparent adverse reproducing effects in Forster's terns, but not common terns (Ankley *et al.*, 1992).

Hoffman *et al.* (1993) also examined hatching success and morphological development in common tern eggs collected in 1985 from Kidney Island, two sites in Saginaw Bay, Michigan, and two reference sites. Eggs were collected in the field and brought back to the laboratory for artificial incubation and chemical analysis of total PCBs, DDE, and mercury. Total PCBs and mercury measured in eggs collected from Green Bay were the highest amongst all eggs collected, while DDE was not significantly different across all sites. Hatching success was found to be significantly lower than one of the two reference sites, while the incidence of abnormal embryos and chicks was reported higher than any of the other sites examined.

Ludwig and Ludwig (undated report cited in EPA, 2000a) performed a field study during the 1986 nesting season, and looked at rates of deformities and reproductive success in Caspian terns nesting on Gravelly and Gull Islands in upper Green Bay, as well as islands in Lake Michigan, Lake Superior, and Lake Huron. The Lake Huron site served as a reference site. The authors found no evidence of developmental defects in Caspian terns nesting in the upper Green Bay. However, they did observe the lowest hatching rate of all the study areas to be in Saginaw Bay and the upper Green Bay, with hatching success on Gravelly and Gull Islands measured to be 72 and 71 percent, respectively, compared with a range of 81 to 84 percent in the remaining colonies.

A similar study (Kurita and Ludwig, 1988) was performed in 1988 in which Caspian tern eggs were collected from colonies nesting on Gravelly and Gull Islands in the upper Green Bay as well as in Lake Huron, Lake Superior, and Lake Michigan. Eggs were examined for viability and developmental deformities and grouped into four categories: live-normal, dead-normal, infertile, and deformed. The deformed category included both dead- and live-deformed. Unclassifiable and rotten eggs were classified as dead-normal. In the upper Green Bay, 13 Caspian tern eggs were classified as live-normal, three as infertile, and two as deformed. Organochlorine residues were examined in conjunction with these results, but unlike the cormorants, no trends could be established between PCB residues and rates of deformities in Caspian terns.

In 1990, Mora *et al.* (1993) examined productivity and colony site tenacity in relation to PCB concentrations in blood samples collected from Caspian terns nesting in the Great Lakes, including Gravelly and Gull Islands in upper Green Bay. They found that productivity, as measured by the number of eggs laid, hatching success, and fledging success, was not significantly different between the upper Green Bay and the other colonies, even though PCB concentrations measured in the blood samples were greater in Caspian terns collected in upper Green Bay and Saginaw Bay compared with the other colonies. However, the

authors report that the hatching success rates observed in this study, which ranged from 74 to 82 percent for all of the colonies studied, were less than the hatching success of Caspian tern colonies nesting in Texas, where 85 percent success has been observed, and in Finland, where 85 to 95 percent success has been noted. Colony site tenacity was exceptionally low in the upper Green Bay colonies (56.5 percent) compared with the other colonies studied (81.2 to 100 percent). The authors explain that Caspian terns are less likely to return to their original breeding area if they experience poor reproduction during the previous year. When natal site tenacity is examined, a correlation is observed with PCB concentrations in blood samples by region, where natal site tenacity decreases with increasing PCB concentrations. However, this correlation is based on a small number of data points. Therefore, more data is needed to confirm this relationship.

Ludwig *et al.* (1996) summarized a variety of studies in the Great Lakes, including a study in Green Bay conducted from 1987 to 1991, in which field observations of Caspian tern egg death rates and deformity rates were made and either total PCBs or toxicity equivalents (TEQs) were measured in eggs. The Green Bay colonies had the highest deformity and egg death rates of all the Great Lakes colonies studied, except for Saginaw Bay, another region that is known to contain high levels of contamination. However, data specific to the various zones within Green Bay could not be deciphered from the data presented. Nonetheless, the authors found a significant correlation between TEQs and deformity rates in hatched tern chicks and dead eggs as well as egg death rates, although only egg death rates exhibited a strong correlation (r^2 = 0.68). Poor correlations were observed between total PCBs and adverse effects.

Ewins (1994) present the results of a 1991 study on Caspian terns nesting in colonies in the Great Lakes, including two islands (Gravelly and Gull Islands) in the upper Green Bay. Although observations were performed on both islands, eggs were only taken from Gravelly Island. Reproductive output was measured by determining the number of active nests per colony, and by monitoring the nests for numbers of eggs, hatching success, and number of young fledged per nest. Average rates of population change were determined by comparing nest counts for the 1991 study with a count that was conducted in 1980. The results indicated that even though the concentrations of PCBs and DDE in the eggs were highest on Gravelly Island and Saginaw Bay, there was no evidence of an overall adverse reproductive effect on Caspian terns in the upper Green Bay, since the number of young per pair was well above the minimum value of 0.6, established by Ludwig *et al.* (1996) to maintain population stability. Furthermore, a dramatic increase in the number of active Caspian tern nests on Gravelly and Gull Islands in the upper Green Bay was observed from 1980 to 1991. The authors caution

in basing definitive conclusions on this study in light of the results of the study by Mora *et al.* (1993) that indicate that PCBs may be affecting certain reproductive parameters such as natal region fidelity (tendency to return to their original breeding area) in the upper Green Bay.

The results of the above studies are not conclusive that terns are at risk from PCBs in the Lower Fox River and Green Bay. The data presented suggest that PCBs are not associated with adverse effects on endpoints such as hatching success and deformities, but one study found a strong negative correlation between Caspian tern site tenacity and PCBs. This indicates that some subtle reproductive effects may be manifesting themselves in Green Bay as a result of exposure to PCB contamination.

Double-crested Cormorant. The decline and subsequent recovery of double-crested cormorant populations in the Great Lakes was previously discussed in Section 2. Briefly, the number of nesting pairs in all of the Great Lakes decreased from approximately 900 in the early 1950s to a low of 125 in 1973. Cormorants all but disappeared as a nesting bird from Lake Michigan in the 1970s, but have subsequently recovered. The cormorant is now more numerous on the Great Lakes than at any time in previously recorded history (Environment Canada, 2000). Once on the state's endangered species list, the cormorant was delisted in 1986.

That PCBs and other chlorinated hydrocarbons may play a role in the depressed reproduction rate as well as contribute to sublethal effects to hatchlings has been well documented in numerous field studies. Ludwig and Ludwig (in an undated report cited in EPA, 2000a) performed a field study during the 1986 nesting season and looked at rates of deformities and reproductive success in double-crested cormorants nesting on islands in upper Green Bay (Gravelly and Little Gull Islands) as well as in Lake Michigan, Lake Superior, and Lake Huron; Lake Huron was used as the reference site. They found that the rates of deformities were higher in the upper Green Bay compared with all other sites. Nine cormorants were observed with deformities, including crossed bill, chick edema, unabsorbed yolk sac, dwarfism, and an opaque covering over the eye. It is unclear whether the last deformity is chemically-induced, but the other deformities are similar to those observed in the laboratory as a result of exposure to PCBs (Ludwig *et al.*, 1996). In addition, the lowest hatching rates were also observed in the upper Green Bay, with 63 percent hatchability in upper Green Bay versus 74 percent observed in the reference area (Lake Huron).

A similar study (Kurita and Ludwig, 1988) was performed in 1988 in which double-crested cormorant eggs were collected from colonies nesting on Little Gull

Island in the upper Green Bay as well as on islands in Lake Huron, Lake Superior, and Lake Michigan. Eggs were examined for viability and developmental deformities and grouped into four categories: live-normal, dead-normal, infertile, and deformed. The deformed category included both dead- and live-deformed. Unclassifiable and rotten eggs were classified as dead-normal. In the upper Green Bay, a high rate of reproductive abnormalities was observed. Specifically, 18 cormorant eggs were classified as live-normal, 15 as infertile, and eight as deformed. Organochlorine residues were examined in conjunction with these results, and it was found that total PCBs were correlated with the numbers of live deformities in cormorant chicks, while rates of dead-normal, dead-deformed, and infertile eggs were better correlated with coplanar PCBs and other chlorinated hydrocarbons.

Fox *et al.* (1991a, 1991b) performed a review of all studies conducted between 1979 and 1987 in which double-crested cormorants were examined for bill deformities in colonies in the Great Lakes, including Green Bay, as well as four reference areas. They found that the prevalence of chicks with bill defects in Green Bay was markedly greater than all other regions during this time interval. These differences were statistically significant ($p < 0.05$) between Green Bay and the North Channel, Alpena, and Lake Erie, and the difference approached significance ($p < 0.1$) for all other regions. The study also determined that the probability of observing a cormorant chick in Green Bay with a malformed bill was 10 to 32 times greater than for colonies in the reference areas. The incidence of bill defects was significantly greater in Green Bay compared with all other regions studied except for Lake Ontario. Bill defects were observed in 73 percent of the colonies observed in Green Bay, as compared with only 6 percent of the colonies observed in the reference areas. The authors suggest a chemical etiology for the observed bill defects, since an investigation into the cause of similar bill defects in Forster's terns indicated that the defects were associated with increased liver-to-body mass ratios and elevated aryl hydrocarbon hydroxylase (AHH) activity. Furthermore, the authors stated that all three of the more toxic non-ortho PCB congeners have been isolated from tissues of cormorant chicks with crossed bills collected from Green Bay. Two of these congeners are known to cause craniofacial abnormalities in laboratory animals. Although the data presented in this study do not allow one to distinguish between the upper and lower Green Bay colonies, the data clearly demonstrate that craniofacial abnormalities were high in double-crested cormorants nesting in Green Bay as a whole between 1979 and 1987, and that these defects may have been caused by exposure to polychlorinated aromatic hydrocarbons such as PCBs.

Tillitt *et al.* (1992) examined reproductive success of double-crested cormorants from 1986 to 1988 in colonies in and around the Great Lakes. They found that

egg mortality was significantly greater in all of the Great Lakes nesting colonies, including the upper Green Bay colonies (Little Gull, Snake, and Gravelly Islands), where egg mortality ranged from 32 to 39 percent. At the reference area (Lake Winnipegosis), egg mortality was only 8 percent. Total PCB concentrations in eggs ranged from 0.05 and 14.8 μg/g ww. The authors found a significant correlation between total PCB concentrations in eggs and egg mortality (p = 0.045). However, the coefficient of determination (r^2) was only 0.319, indicating that much of the variance in egg mortality was not explained by this general linear model. A significant correlation was also observed between egg mortality and the H4IIE rat hepatoma bioassay-derived 2,3,7,8-tetrachloro-*p*-dibenzodioxin equivalents (TCDD-Eq) concentrations ($p \leq 0.0003$, $r^2 = 0.703$). The eggs were analyzed for total PCBs, polychlorinated dibenzo-*p*-dioxins (PCDD) and polychlorinated dibenzofuran (PCDF)-type planar halogenated hydrocarbons (PHHs), and only PCBs were detected. This indicates that PCBs are the main contaminant associated with the observed egg mortality in double-crested cormorants in the Great Lakes colonies, including upper Green Bay.

Ludwig *et al.* (1996) summarized a variety of studies conducted from 1986 to 1991, in which field observations of double-crested cormorant egg death rates and deformity rates were observed and either total PCBs or TCDD-Eqs were measured in eggs for colonies in the Great Lakes, including the upper Green Bay. Deformity rates were higher in all Great Lakes colonies than at a reference colony. Of all the Great Lakes colonies studied, the upper Green Bay had the highest rate of egg deformities (6.14 per thousand for upper Green Bay versus a range of 0.69 to 3.6 per thousand for the other Great Lakes colonies). Similarly, the egg death rate for Green Bay was higher than any other colony studied, although data specific to the upper Green Bay could not be deciphered from the data presented for Green Bay. PCB concentrations ranged from 0.8 mg/kg ww at the reference colony to 7.3 mg/kg in eggs collected from Green Bay. The authors found a significant correlation between hatching and deformity rates for both PCBs and TCDD-Eqs, indicating that PCBs are playing a large role in the cormorant egg death and deformity rates observed in the upper Green Bay.

More recently, other authors have suggested that reproductive dysfunction or abnormal development in Green Bay may not be entirely related to PCBs. Larson *et al.* (1996) examined the relationship between tissue residues in cormorants with clutch size, hatching success, and the frequency of deformities in chicks on Spider Island off the eastern shore of the Door Peninsula in Lake Michigan. For sample years 1989 and 1990, both clutch size, hatching success, and rate of deformities were significantly different from a reference site in Lake Winnepegosis. They concluded that PCBs likely contributed to the reduced hatchability. However, the authors found that the measured total PCBs (7.6 mg/kg) and TEQ (134 ng/kg)

concentrations in sample eggs from clutches where all eggs produced fledged chicks were not significantly different from total PCBs (8.2 mg/kg) and TEQ (134 ng/kg) concentrations in sample eggs from clutches where none of the eggs hatched. These authors concluded that they could neither support nor reject the hypothesis that environmental chemicals were effecting reproduction.

In the most recent detailed analysis of cormorants in Green Bay, Custer *et al.* (1999) followed the nesting success of 1,570 eggs laid on Cat Island in Green Bay. Mean chemical concentrations in these eggs were: 13.6 mg/kg total PCBs, 3.9 mg/kg DDE, and 0.25 mg/kg dieldrin. Hatching success was positively correlated with eggshell thickness and negatively correlated with DDE and dieldrin concentrations in sample eggs. Logistic regression indicated that concentrations of DDE, but not dieldrin or PCBs, in sample eggs were a significant factor in the hatching success of double-crested cormorant eggs. No relationship was discovered between PCBs and hatching success. Furthermore, there was no correlation found between measured EROD activity in embryo livers and hatching success. PCB concentrations in deformed embryos were also found to not be significantly different from PCB concentrations in embryos that were not deformed. While the PCB concentrations in double-crested cormorant eggs were higher at Cat Island than in other colonies in Green Bay, the frequency of deformities in double-crested cormorants (0%) chicks at Cat Island in 1994 and 1995 was lower than those reported from other Green Bay colonies. Those authors suggest that previously reported relationships between PCBs and egg mortality of cormorants in the Great Lakes was due to DDE, and not to PCB concentrations. The paper concluded that given the rapidly expanding breeding populations of cormorants in the Great Lakes, DDE contamination (and by implication, PCBs) do not seem to be a significant risk factor to double-crested cormorant populations in Green Bay.

Based on the results presented in the studies summarized above, double-crested cormorants have experienced adverse reproductive effects throughout the Lower Fox River and Green Bay. Deformities such as crossed bills, edema, unabsorbed yolk sac, and dwarfism as well as embryo mortality are characteristic of abnormalities observed as a result of exposure to polychlorinated hydrocarbons such as DDE and PCBs. The current reproductive success of the birds within the bay indicates that cormorant populations are not at risk. However, persistence of low levels of abnormal development within the area indicates that some level of risk may remain to individual double-crested cormorants.

Bald Eagles. The presence of organochlorines in bald eagle tissues has long been linked to low reproductive rates. In Green Bay specifically, all former bald eagle nest sites were abandoned by 1970 (Dykstra and Meyer, 1996). Following the ban of

DDT and other organochlorines in the 1970s, bald eagles began to nest again in former nesting areas. For the shores of Lake Michigan, the first recorded nest initiation following the ban did not occur until 1986 on the Peshtigo River. Since that time, nesting pairs on Wisconsin's shores slowly increased to a maximum of five nesting pairs that were found in 1993 to 1995. Bowerman (1993) determined that the reproductive rate of nesting bald eagles in Green Bay was much lower than the reproductive rates of bald eagles nesting in inland areas of Wisconsin and Michigan.

Dykstra and Meyer (1996) further examined the relationship of organochlorine contaminants (PCBs and DDE) on the reproduction of bald eagles in Green Bay. This investigation was also designed to examine the degree of food availability in Green Bay because food availability has been shown to be an additional limiting factor for bald eagle reproduction in uncontaminated areas.

The number of occupied breeding territories of bald eagles in Green Bay has slowly increased since 1987, when there were five, to 12 in 1995. Occupied breeding territories are defined as an area where eggs have been laid and either two eagles are present or nest repairs are visible. In 1994 to 1995, all bald eagle nests within 8 km of the shore of Green Bay were selected as sampling stations. Twelve nests met this criterion and a thirteenth nest located on the eastern side of the Door Peninsula was included in the study because it was assumed that this nest was as influenced as the other nests by contaminants in Green Bay (Dykstra and Meyer, 1996).

The reproductive rate at these nests was measured by aerial survey: twice during the breeding season, once during incubation, and once when the nestlings were 4 to 7 weeks old. The organochlorine contaminants were measured in the only addled egg recovered from a nest in 1995 and in the blood from eight nestlings. A dead nestling was found in 1995, but not analyzed. Food availability was determined by measuring behavior parameters: the food delivery rate to nestlings, the time spent feeding, and rate of adult nest attendance.

Bald eagle productivity rates from 1987 to 1995 in Green Bay and the Lower Fox River are presented in Table 6-118 and summarized in Table 6-119. Mean productivity data for Green Bay, inland Michigan, and inland Wisconsin are presented on Figure 2-26. These data indicated that despite the increase in occupied territories around Green Bay, bald eagle productivity still was low and that the productivity of bald eagles in Green Bay was lower than the productivity of bald eagles in the Lower Fox River. The reported average productivity of bald eagles in Green Bay from 1990 to 1994 was 0.39 young per occupied territory, and results from 1995 also indicated that productivity was low; of five nesting

attempts, only three were successful (Dykstra and Meyer, 1996). Comparatively, the productivity of inland nests averaged 1.09 young per occupied territory and the productivity of nests on the Lower Fox River averaged 1.9 young per occupied territory over the same period.

None of the behavioral parameters measured differed significantly from inland measures, suggesting that food availability was within a normal range.

Average organochlorine concentrations in addled eggs collected in Green Bay between 1986 and 1992 were 35.0 mg/kg total PCBs and 10.3 mg/kg DDE (Table 6-120). Analytical results of the single addled egg collected in 1995 were not available at the time the report was published. Mean plasma concentrations were 0.207 mg/kg PCB and 0.053 mg/kg DDE (Table 6-120). Evidence is stronger that DDE is responsible for low reproduction rather than total PCBs. The mean concentration of DDE measured in Green Bay bald eagle eggs was midway between concentrations of DDE known to cause reproductive failure (15 to 16 mg/kg) and concentrations not associated with adverse effects (up to 3.6 mg/kg). Although total PCB concentrations have not been as well linked to reproductive effects, total PCB concentrations in eggs from Green Bay were higher than what a nationwide study determined to be the threshold for reproductive failure (33 mg/kg). Although thresholds for organochlorine contaminant levels in blood have not been correlated to adverse reproductive effects, the concentrations measured in nestlings from Green Bay were 6 to 14 times higher than concentrations measured in nestlings from inland areas.

The fact that organochlorines were frequently detected in bald eagles from Green Bay, that concentrations were similar to or higher than other highly-contaminated areas across the U.S., and that food availability in Green Bay was not limited suggests that organochlorine contaminants likely are the cause of low reproductive rates in bald eagles in Green Bay.

Bald eagles were one of the species evaluated in the NRDA examination of injuries to avian resources in the Lower Fox River and Green Bay that was conducted by USFWS (Stratus Consulting, 1999c). This report was completed in 1999 and incorporates more recent data in addition to the data summarized by Dykstra and Meyer (1996). Egg contaminant data were presented for years 1987 to 1997, where mean concentrations of total PCBs and DDE were 46.1 and 12.5 mg/kg, respectively. As compared to the mean concentration summarized by Dykstra and Meyer (1996) that included data through 1992, concentrations of both PCBs and DDE between 1992 and 1997 increased in eggs rather than decreased.

The NRDA injury report also had 3 years of additional productivity data (through 1998) compared to the Dykstra and Meyer (1996) report, and productivity in these years was still found to be significantly lower in Green Bay than productivity measured in inland Wisconsin or on the Lower Fox River.

Often, concentrations of PCBs and DDE in eggs are correlated (Stratus Consulting, 1999c). As pointed out in the NRDA report, when the ratio of PCB and DDE egg concentrations and productivity data are compared, it was found that while the ratio of PCB and DDE egg concentrations is highest in the Lower Fox River, productivity is also higher. Conversely, Green Bay has a lower PCB to DDE concentration ratio and a lower productivity rate. These relationships suggest that DDE concentrations rather than PCB concentrations may be limiting productivity, or that PCB concentrations in eggs, although similar between Green Bay and the Lower Fox River, exert less of an effect on Lower Fox River eggs and more of an effect on Green Bay eggs (Stratus Consulting, 1999c). It should be noted, however, that data for the Lower Fox River were limited to 1 year and one egg, and more data are necessary to test these hypotheses.

Bald eagle productivity data were converted into a probability that the nest would produce no young or at least one young and then compared to PCB concentrations in eggs (Stratus Consulting, 1999c). This comparison indicated that the probability that an eagle nest will raise no young steeply increases once egg PCB concentrations exceed the threshold of 20 mg/kg. All but one of the bald eagle eggs analyzed in Green Bay exceed the 20 mg/kg concentration, indicating that adverse effects on productivity from PCBs are likely in Green Bay. In general, however, investigations have noted that productivity is not consistently correlated to individual contaminant concentrations (Donaldson *et al.*, 1999; Wiemeyer *et al.*, 1993; Dykstra and Meyer, 1996) and results of statistical analyses conducted for the NRDA evaluation also concluded that both contaminants may be affecting productivity, and separating their individual effects is not feasible (Stratus Consulting, 1999c).

The role of PCBs in accounting for the differences between the apparent reproductive success of the Kaukauna nest on the Fox River relative to the Green Bay nests is not clear; total PCB levels are similar in eggs. However, the USFWS avian injury report cites unpublished data from Tillitt *et al.* that found that the total TEQs in the Kaukauna eggs is lower. Based on the H4IIE assay method, two eggs from a Peshtigo Marsh nest in 1988 averaged 147.5 TCDD-Eq, while one egg from the Kaukauna nest had only 34 pg/g (Stratus Consulting, 1999c). The lower dioxin-like congeners in the Kaukauna eggs could account in part for the apparent reproductive success.

Piscivorous Mammals

To date, there have not been field studies on the effects of contaminants on piscivorous mammals in the Lower Fox River or Green Bay. Commercially-caught yellow perch from Green Bay were previously sold as feed to mink ranchers in Wisconsin until the fish source was found to be toxic to ranched mink in the 1960s (Aulerich *et al.*, 1973). The source of the toxicity was attributed to elevated levels of PCBs in the fish. The WDNR requires trappers to report otter takes and maintains annual otter trapping data by county. Gilbertson (1988), citing the otter trapping data, noted that no otters were trapped in lower Green Bay and suggested that chlorinated hydrocarbons in fish were potentially responsible. More recent data from WDNR shows that a single otter was collected during the 1998–1999 trapping season and none were collected along the Door Peninsula. However, these observations should be used with caution as other factors such as available habitat must be considered.

Within the Fox River database, there are only three collections of mink tissue submitted for analysis for total PCBs and mercury. A single mink taken within the Oconto River in 1986 showed mercury levels of 190 μg/kg and total PCB levels of 1,500 μg/kg in muscle tissue. A second mink taken in 1986 from Daly Creek, a tributary to the Little River and subsequently the Oconto River, had mercury levels of 190 μg/kg and total PCBs of 2,300 μg/kg from muscle and tissue. The USFWS 1999 NRDA tissue collection had a single mink analysis of liver from a mink collected in Zone 2 with a liver concentration of total PCBs measuring 40.4 μg/kg. In a data point from the scientific literature, Gilbertson (1988) cites a USFWS collection of mink with a total liver PCB concentration of 5,700 μg/kg from a marsh in Green Bay.

6.6 Uncertainty

The sections above evaluated the potential for PCBs and other COPCs to be bioaccumulated from bedded sediments into the aquatic food chain. Using point estimates at the mean and 95th percentile upper confidence limits, the BLRA estimated risks from COPCs to:

- Benthic infauna in all reaches and zones of the river and bay;
- Benthic and pelagic fish in all reaches and zones of the river and bay;
- Piscivorous birds within Zone 1 and Green Bay;

- Carnivorous birds, principally eagles, within Zone 1 and Green Bay; and

- Piscivorous mammals in all sections and zones of the river and bay.

Of the COPCs, PCBs were indicated for all the receptors in all reaches or zones, but mercury and p,p'-DDE were also found to be at levels that are associated with risks to aquatic and aquatic-dependent organisms.

While these are point estimates expressed as hazard quotients, the risk determinations were qualitatively corroborated with field-derived observations of habitat, population information, and measured reproductive impacts in the affected receptor species within the Area of Concern. Thus, there is a reasonable degree of certainty of impacts from exposure to toxic chemicals within the Lower Fox River and Green Bay. Bounding that degree of certainty is the function of the uncertainty analysis.

The goal of this uncertainty analysis is to both qualitatively, and quantitatively to the degree possible, define the degree of confidence that exists with the estimations of effects from exposure to hazardous chemicals in toxic amounts. Bounding the certainty of risk estimates is a developing science. EPA's Superfund Ecological Risk Assessment Guidance (EPA, 1997a) and the Guidelines for Ecological Risk Assessment (EPA, 1998b) provide general instructions on what should be addressed in an uncertainty analysis. Within the ecological risk assessment industry, more recent recommendations for conducting an uncertainty analysis are given in Bartell *et al.* (1992), Suter (1993), and Warren-Hicks and Moore (1998).

Based upon the review of those documents, the following areas of uncertainty were identified within the Lower Fox River/Green Bay BLRA:

- **Conceptual Site Model.** Are the fate and transport, uptake mechanisms, and selected receptors sufficiently understood to adequately characterize the risks to sensitive habitat and species?

- **Data Uncertainty.** Is there adequate data of sufficient quality to support the conclusions of the ecological risk assessment?

- **Temporal Uncertainty.** Do the changes in concentrations over time within the measured media in the river and bay effect the conclusions of the point estimates of risk, and how are those quantified?

- **Spatial Variability.** How do the estimates of COPC distributions within a reach or zone by media effect the conclusions of the point estimates of risk?

- **Toxic Exposure Uncertainty.** How well are the estimates of toxicity supported by observations within the scientific literature?

- **Population Uncertainty.** How well does population information reflect the presence of risk given other confounding variables affecting population levels at any given point in time?

- **Quantitative Uncertainty.** What is the frequency, or probability, of the receptors within the river or bay encountering hazardous chemicals in toxic concentrations?

Each of these sources of uncertainty is addressed below.

6.6.1 Uncertainty in the Conceptual Site Model

The Lower Fox River and Green Bay have been the focus of hundreds of studies that have addressed the sources of PCBs into the river and the fate and distribution of those contaminants in sediments, pore water, downstream transport, biological uptake, and effects on field populations. In addition, there have been numerous studies that have documented the habitat and life histories of important fish, birds, and mammals within the river and bay. The documentation supporting the physical and biological systems are found in the Remedial Investigation (RETEC, 2002a), and were summarized in Section 2. Additional documentation on the physical fate and transport mechanisms, hydraulic conditions, and biological uptake mechanisms within the river and bay may be found in the Model Documentation Technical Report for the Lower Fox River and Green Bay (RETEC, 2002c). Additional support for the conceptual site model is also given in Technical Memorandum 7c: Food Webs of the Lower Fox River and Green Bay (WDNR, 2001). Thus, qualitatively, there is a high degree of certainty that these factors are well understood and adequately characterized in the conceptual site model.

There is some uncertainty as to whether the receptors identified within the conceptual site model adequately represent the ecosystem and other species potentially at risk within the Lower Fox River and Green Bay. The selection of the important receptor species was done in consultation with biologists both within the WDNR and the USFWS. In addition, input on the receptor species was given by biologists and resource managers within EPA, NOAA, and the Oneida and Menominee Nations through the BTAG process. However, despite

this, there remains a class of organisms and a threatened species that was not addressed in this BLRA.

Reptiles, and specifically amphibians (e.g., frogs, turtles), were not addressed in this BLRA. PCB and PCDF accumulation has been demonstrated in most reptiles. Elevated levels of PCBs have been associated with lower hatching rates in eggs of snapping turtles in the Great Lakes (Bishop *et al.*, 1995), sex reversal in turtles and alligators (Willingham and Crews, 1999), endocrine disruption in salamanders (Gendron *et al.*, 1997), and developmental abnormality in toads and frogs (Jung and Walker, 1997). Given the global concerns about declining amphibian populations, omitting amphibians in this BLRA excludes an apparently sensitive group of organisms that would be exposed to PCBs via sediments, surface water, and prey (insect or small fish) species. This was not a deliberate omission, rather an admission that there are no data within the FRDB to evaluate this receptor group, and that there are no uptake models to estimate risk for frogs or other amphibians. Thus, there remains an uncertainty that all risks have been adequately characterized within the system.

Sturgeon are listed as a threatened species in Michigan, but not in Wisconsin. There are few data points within the FRDB to evaluate potential risks to lake sturgeon. Of the seven measurements for total PCBs, the range of concentrations is between 850 μg/kg in Zone 4 up to a high of 5,200 μg/kg in a fish collected from the mouth of the Menominee River in Zone 3A. There are no other data from any other reach or zone. While these values certainly demonstrate exposure and suggest the potential for risk, it is important to note that the Wisconsin part of the Green Bay system is unique in that there have been harvestable fish within the rivers. There has been sport fishery in the Menominee, Peshtigo, and Oconto Rivers, although recently WDNR elected to close the fishery season on the Menominee in order to try and increase the overall population levels within Green Bay. WDNR also recently closed the hook-and-line season on Green Bay and its tributaries, while maintaining an open season on the Menominee River but increasing the size limit to 70 inches. Within the last 3 years, WDNR has observed lake sturgeon in increased numbers attempting to spawn within the Fox River below the De Pere dam (Lychwick, 2000). Ultimately, there are insufficient data to determine if there has been an impact to sturgeon from exposure to contaminants from the Fox River.

6.6.2 Uncertainty in the Data Supporting the Ecological Risk Assessment

As noted previously, conditions on the Lower Fox River and Green Bay have been extensively studied and documented since the 1960s. Section 4 of this BLRA focuses on the adequacy and quality of the data within the Fox River database

used for this ecological risk assessment. The FRDB represents 35 separate studies with 18,800 discrete samples of water, sediments, and tissues, and over 474,000 discrete data points throughout the Lower Fox River and Green Bay between 1971 and 1999. A rigorous evaluation of the quality of the data was undertaken, and only data for which at least partial QA packages could be reviewed were placed into the FRDB. Of the studies between 1971 and 1991, only partial packages could be reviewed, and so those data were used as supporting evidence within the BLRA. There have been several studies completed on the Fox River in the 1990s. All studies conducted after 1992 have fully-validated data packages. Given the temporal and spatial density of the data within the Lower Fox River, there are good reasons to assume that the overall quality of the data is high, and thus the related degree of data uncertainty is low. There were no significant biases or gaps observed within the sediment, fish, or bird sample data.

There is uncertainty in risk estimates using Green Bay sediments and water data. Those data, collected principally as part of the 1989 Mass Balance Study, were collected on a 10-km grid basis. With the exception of some sampling done at the mouth of the Fox River in Zone 2, almost all of the data used in the interpolated bed maps are from the 1989 collection effort. The lack of more recent data raises some temporal uncertainty in the interpolations. While for a large water body such as Green Bay, a 10-km sampling gird yields a fairly dense data set, interpolating between points 10 km apart does increase the uncertainty with the spatial estimates. The potential magnitude of those uncertainties on risk estimates is discussed further in the following sections. In addition, while the Mass Balance data were judged to be suitable for supporting a risk assessment, they lack a complete set of the quality assurance and quality control (QA/QC) documentation. While overall the conduct, execution, and review of at least the partial QA/QC packages suggest that the Mass Balance data are of high quality, the lack of a more recent, corroborative, and fully-validated data set leaves the sediment and water-based point risk estimations highly uncertain.

By contrast, for Green Bay there are a number of more recent, well-documented and fully-validated data sets for some fish (carp and walleye), as well as birds (double-crested cormorants, tree swallows, and eagles). These data sets confirm that PCB exposure throughout Green Bay is still occurring via sediment and waterborne PCBs.

One additional data gap within the evaluation is the limited measurements of metals and the organochlorine pesticides in the surface water. However, this impacts only the ability to assess risks to pelagic invertebrate communities, and the remaining assessment endpoints could be addressed through the other media (e.g., bird tissues) for which data were judged adequate.

Finally, there are relatively too few data on all PCB congeners for all media within the Lower Fox River and Green Bay to make conclusive assessments or predictions of risk. While the FRDB contains numerous congener-specific data points, until relatively recently all of the dioxin-like congeners have not been adequately assessed. For example, while PCB congener 169 has been detected in the fish and birds of the river and bay, there have been too few measurements taken in sediments or water.

6.6.3 Temporal Variability

Uncertainty in the temporal variability reflects the decisions made to include or exclude sediment and fish tissue data from the BLRA exposure estimates. For the Lower Fox River and Zone 2 of Green Bay, the PCB-interpolated sediment bed maps were derived based upon use of the most recent data, relying on older data from as early as 1989 only where there were no other data available to define sediment COPC concentrations. For zones 3A, 3B, and 4, the sediment point estimates were based solely on data collected in the 1989 Mass Balance Study. Point estimates for fish used data collected after 1989, which excluded a significant amount of data collected prior to that from the Lower Fox River and Zone 2. An exception to this was in Zone 4, where there were no other fish tissue data except those collected in 1987. At issue in the temporal uncertainty is an estimate of whether these data aggregations or exclusions resulted in over- or underestimations of risks to aquatic and aquatic-dependent receptors.

The time trends analysis for sediments and fish tissue PCBs was undertaken to specifically address the question of losses or gains in PCB concentrations over time in sediments and fish. The methods and results were previously discussed in Section 2.6, and are presented as an appendix to the RI. For sediments, the results suggest that generally over time, the surface sediment concentrations of PCBs have been steadily decreasing, but that this decline is generally restricted to the top 10 cm. The rate of change in surface sediments is both reach- and deposit-specific. The change averages an annual decrease of 15 percent, but ranges from an increase of 17 percent to a decrease of 43 percent. A large fraction of analyses provided little useful information for projecting future trends because of the lack of statistical significance and the wide confidence limits observed. This is especially true for sediments below the top 4 inches; changes in the sediment PCB concentrations cannot be distinguished from zero—or no change.

Given these conditions, the sediment data used may over- or under-evaluate the risks. The net effect upon the risk calculations thus are dependent upon how much older data were used in the point estimates or interpolated bed maps. Where a sole reliance existed on data from 1989, surface sediment estimations of PCBs could potentially be reduced by as much as two to nine times (assuming a

steady 15 ± 7 percent decrease annually over 10 years). Green Bay was not included in the time trends analysis for sediments, but the observed declines with wide confidence intervals are probably applicable there as well. The risk estimates most affected would be zones 3 (A and B) and 4, where only 1989 data were available to conduct the interpolations and estimate risks. Zone 2 also includes much of the 1989 Mass Balance data, except near the mouth of the river where there are data from 1992 and 1996. Thus, sediment PCB toxicity is likely overestimated in these zones. Conversely, risk estimates made from most of the data used from Little Lake Butte des Morts, Little Rapids to De Pere, and Zone 1 rely on more recent data, and likely are adequate representations of current sediment PCB concentrations. It should be noted again that the apparent decreases in surface sediment concentrations were limited to the upper 4 inches. Flood or other scouring events likely have caused resuspension or re-exposure events, which would change the rate of declines noted above.

Like sediment PCB concentrations, fish tissue PCB concentrations showed a significant but slow rate of change throughout the Lower Fox River and Green Bay. In all of the reaches of the river and in Zone 2, there were steep declines in fish tissue PCB concentrations from the 1970s, but with significant breakpoints in declines beginning around 1980. After the breakpoint, depending upon the fish species, the additional apparent declines were either not significantly different from zero, or were relatively low (5 to 7 percent annually). In addition, there are some increases in fish tissue PCB concentrations. Walleye in Little Lake Butte des Morts show a non-significant increase of 22 percent per year since 1987. Likewise, gizzard shad in Zone 2 show a non-significant increase of 6 percent per year into 1999. These data, taken collectively, suggest that since the breakpoint for tissue declines occurred in the early 1980s and the changes in fish tissue concentrations were no greater than 4 to 7 percent annually, aggregating fish tissue from 1989 does not likely result in any significant biasing of the risk estimations. At worst, the tissue point estimates might overestimate risks by 50 percent (average of 5 percent per year over 10 years), but given that at least some fish tissue concentrations increased, it is reasonable to suggest that some risks were underestimated by at least an equivalent amount.

6.6.4 Spatial Variability

Uncertainty in the spatial variability refers principally to where sediment samples were collected from within the Lower Fox River and Green Bay. Within the river, most sampling efforts are concentrated in areas where there were thick sediment deposits (e.g., A, POG, N, GG/HH, and the SMUs below De Pere). There were no systematic sampling efforts to define PCB concentrations throughout the river. Within the bay, systematic grid sampling was employed, but the spatial uncertainty is higher because of the large distance between sampling points.

Sediment PCB-interpolated bed maps were developed as a means to view the spatial distribution of PCBs. The inverse distance-weighting method was used, as discussed in Section 2.3.1. Spatial variability is a function of the methods used for calculating the sediment interpolations. Interpolated bed maps were developed by WDNR and were discussed in Section 2 and in WDNR Technical Memorandum 2f (WDNR, 2000b).

For the risk assessment, three indicators of sediment quality were used: the mean of the actual values measured, and two estimated sediment weighted average concentrations (SWAC) based upon the interpolated PCB sediment concentrations. As noted previously, within the bed map estimation method, data points more than the interpolation radius from the grid point were not used in the interpolation. If there were no data points within the interpolation radius of a grid point, then no value was interpolated for that grid point. The I_0 and I_d SWACs were derived by either substituting a zero where there was no grid value (I_0) or deleting that grid point (I_d), and then calculating the mean and 95% UCL.

To evaluate the effect of the interpolated values versus the I_0 and I_d -derived SWAC, values were compared against the means derived from the actual data. Total PCB I_0 concentrations, represented as a percentage of the means derived from the actual data, ranged from 3 to 445 percent. Concentrations for each reach were:

- Little Lake Butte des Morts Reach = 31 percent,
- Appleton to Little Rapids Reach = 3 percent,
- Little Rapids to De Pere Reach = 43 percent,
- De Pere to Green Bay Reach (Green Bay Zone 1) = 71 percent,
- Green Bay Zone 2 = 445 percent,
- Green Bay Zone 3A = 65 percent,
- Green Bay Zone 3B = 88 percent, and
- Green Bay Zone 4 = 43 percent.

Results of I_d interpolation were similar to I_0 interpolation. Total PCB I_d concentrations, represented as a percentage of the non-interpolated concentration of total PCBs, ranged from 21 to 451 percent. Concentrations for each reach were:

- Little Lake Butte des Morts Reach = 34 percent,
- Appleton to Little Rapids Reach = 21 percent,
- Little Rapids to De Pere Reach = 43 percent,
- De Pere to Green Bay Reach (Green Bay Zone 1) = 71 percent,
- Green Bay Zone 2 = 451 percent,

- Green Bay Zone 3A = 68 percent,
- Green Bay Zone 3B = 89 percent, and
- Green Bay Zone 4 = 55 percent.

The calculations demonstrate that in general, using the interpolated SWAC yields a lower estimation of sediment-based risk than use of the actual data themselves. This is an expected result. As noted previously, data collection in the Lower Fox River has been biased toward deposits with higher PCB concentrations (e.g., Deposits A, POG, N, SMU 56/57). The higher data density in those "hot" deposits, or SMUs, would bias the data. The purpose of the interpolation is to project concentrations across the entire reach or zone and eliminate the apparent bias. The exception to the above discussion is Zone 2, where the interpolated means were approximately 450 times the observed data. It is not clear why this condition exists.

These calculations show that with the exception of the Appleton to Little Rapids Reach, the two interpolation means were relatively similar. This reach is unique within the river in that there are few deposits within this fast-flowing section, and the non-interpolated grid points in the area are high. This resulted in the interpolated means being much lower than the data-derived means, and in a large difference between the two interpolated means.

6.6.5 Toxic Exposure Uncertainty

Point estimates of exposure concentrations were compared in the BLRA to point estimates of toxicity in the literature to yield the hazard quotients. While the rationale used to select the most representative value from the literature was presented in Section 6.3, there remain uncertainties associated with effects concentrations above or below the selected TRV, selection of TRVs from one species and applying to another, interpretation between NOAECs and LOAECs based on application of uncertainty factors, or application of different sets of toxicity equivalent factors from the literature. These are discussed below with reference to the appropriate receptor groups.

Benthic Infauna

Risks to benthic infauna in the point estimation were determined by comparing the calculated sediment PCB 95th percent UCL to the ARCS Threshold Effects Concentration of 31.6 μg/kg. The ARCS SEC is one of several sediment PCB effects concentrations derived from multiple sediment endpoints in the toxicological literature. These values were recently reviewed by MacDonald *et al.* (2000), and are presented in Table 6-129. The range of concentrations reported in the scientific literature for sediment-based PCB effects ranged from a low of 3 μg/kg (Neff *et al.*, 1986), to a severe effect level of 5,300 μg/kg (Persuad *et al.*,

1993). Based on a statistical analysis of the various data sets that comprise these sediment effects concentrations, MacDonald *et al.* (2000) developed and evaluated consensus-based sediment effect concentrations of PCBs. Those authors established a Threshold Effect Concentration (TEC) of 40 μg/kg, a Moderate Effect Concentration (MEC) of 400 μg/kg, and an Extreme Effect Concentration (EEC) of 1,700 μg/kg of PCBs in sediments. It should be noted that the terms "Threshold," "Moderate," or "Extreme" are all subjective terms in the literature, and are never adequately defined for the purposes of making a management decision.

The ARCS SEC used in this BLRA are practicably equivalent to the consensus-based TEC (31.6 versus 40 μg/kg), and would not result in any significant changes to the estimated HQs if the latter value was applied. Both the SEC and TEC values do represent the low end (high risk) of the risk estimation spectrum; application of the MEC would yield HQs of approximately one order of magnitude lower than the TEC, and use of the EEC would result in HQs over 400 times lower than those currently used in the risk estimates. The probability of encountering those alternative levels in each of the reaches and zones is discussed in more detail in the next section.

Fish

Selection of an appropriate PCB TRV for fish within the Lower Fox River/Green Bay system was complicated principally by two conditions: toxicity has not been adequately characterized for all receptors, and both laboratory and field studies were evaluated and were not in agreement regarding toxic thresholds. There are several well-supported studies relating to salmonids, but none relating to percids (walleye and perch), and little for cyprinids (carp). Toxicity results of these studies, however, have differed widely depending on whether the studies were conducted using laboratory-reared fish or field-collected fish. Laboratory studies suggest that PCB toxicity thresholds are one to two orders of magnitude greater than toxicity thresholds experienced by fish in the field. The USFWS in their NRDA examination of fish injury have suggested that these differences are because field-collected fish are often thiamine deficient and that this thiamine deficiency causes fish to be more sensitive to PCB toxicity. This hypothesis, however, has not been well tested.

Birds

Selection of an appropriate PCB TRV for birds within the Lower Fox River/Green Bay system was complicated principally by three conditions: most toxicity studies on birds have been conducted in the field where multiple contaminants are present; tissue data within the FRDB are dominantly around Green Bay Zone 2 and therefore, most area-specific bird tissue concentrations were estimated using

modeling; and PCB TEFs that have been proposed for birds differ widely depending on the source. For birds within the Lower Fox Fiver and Green Bay system, studies have indicated that both PCBs, DDE, and mercury may be concentrating in eggs. Determining the relative contribution of these contaminants to observed toxicity is very uncertain. TEFs have been derived for birds based on different species and different analytical methods as discussed in Section 6.3.2. The two TEFs selected for use in the risk assessment were the TEFs proposed by Tillitt *et al.* (1991b) and those proposed by the World Health Organization (WHO) (Van den Berg *et al.*, 1998). The Tillitt *et al.* TEFs were derived based on toxicity in some site-specific receptors found in Green Bay, while the WHO TEFs compiled a larger data set which included the Tillitt *et al.* TEFs in their evaluation. When the RME NOAEC HQs derived using both TEFs are compared, it is found that for all bird species the WHO-estimated HQs were 5 to 16 times greater than the Tillitt *et al.*-estimated HQs. Therefore, there is uncertainty regarding the level of risk posed by PCB congeners to birds.

Mammals

Of the selected receptors evaluated for risk, risk to piscivorous mammals is comparatively more certain. Several studies have been conducted using mink, a very sensitive species known to be present within the Lower Fox River and Green Bay system. Toxicological information on other piscivorous mammals is not readily available, but it is assumed that other species that may be present in the system, such as river otters, are adequately protected from the conclusions based on the mink exposure modeling and HQ generation.

6.6.6 Alternative Exposure Point Uncertainty

Exposure point concentrations in the BLRA were based upon the mean, the 95% UCL on the mean from water, or the maximum measured value in sediment, and tissue data in the FRDB. An alternative way to evaluate risks to receptor species is to estimate the 90th percentile concentrations for tissue concentrations of total PCBs. To evaluate the potential effects on risk estimation using the 90th percentile, hazard quotients were re-estimated for two representative species; walleye and double crested cormorants. It should be noted that for limited sample sizes, 90th percentile concentrations could not be calculated. To estimate a 90th percentile requires a minimum of 10 samples; these were not available for either or both species on all reaches or zones.

Available 90th percentile calculated COPC concentrations and resulting hazard quotients for walleye are presented in Tables 6-130 and 6-131. Risk evaluation of the 90th percentile concentrations would result in only two changes to the risk conclusions determined based on the evaluation of the RME concentrations. Hazard quotients for the total PCB NOEL for walleye in Green Bay Zone 1

increase from 10 to 14 using the 90th percentile. The risk determination for walleye from total PCBs would change from "potential risk" to "likely risk" in Green Bay zones 1 and 2, and risk from mercury in Green Bay Zone 4 would change from "no risk" to "potential risk." The net conclusions of the ecological risk assessment for piscivorous fish would be negligibly affected by using the 90th percentile.

Available 90th percentile calculated COPC concentrations and resulting hazard quotients for double-crested cormorants are presented in Tables 6-132 and 6-133. Risk evaluation of the 90th percentile concentrations would result in only one change to the risk conclusions determined based on the evaluation of the RME concentrations. Risk to double-crested cormorants from p,p'-DDE would change from "potential risk" to "likely risk" in Green Bay Zone 3B. Because of the limited 90th percentile data in fish appropriate as prey for double-crested cormorants, dietary concentrations could not be modeled. However, use of the 90th percentile would not appreciably affect the risk determinations for piscivorous birds.

6.6.7 Population Uncertainty

As noted previously, while population level endpoints can be an appropriate tool to assess risk, the population data discussed in the BLRA were not collected specifically for risk assessment. There is some uncertainty introduced given the potential for other confounding environmental factors that may effect the absence or abundance of receptors within the Lower Fox River and Green Bay. These can include such things as immigration, emigration, food availability, habitat suitability and availability, species competition, predation, and weather. For example, while the risk assessment concludes that PCBs are at sufficient concentrations to affect mink reproduction within the river and bay, Section 2 documented that there is limited habitat for mink—especially along the river. While contaminant conditions exist that potentially would jeopardize mink health along the river corridor, the absence of mink due to absence of habitat must be considered.

Likewise, the apparent increase in populations of walleye and cormorants suggest little or no current risks to these species. Increases in walleye populations have occurred since the 1980s, and are directly linked to improvement in water quality and habitat in the Lower Fox River, and not necessarily to decreases in contaminants. Evidence that some risks persist is evidenced in the apparent presence of pre-cancerous lesions. Cormorant population increases may be related to decreases in contaminant concentrations, but are also likely tied to increases in available prey (fish). Like walleye, sublethal conditions appear to persist within

the cormorant population. Given a shift in food or habitat conditions, those risks could be potentially of greater concern.

6.6.8 Quantitative Uncertainty

The goal of an ecological risk assessment is to estimate the type, magnitude, and probability of occurrence of effects resulting from exposure to a stressor (Warren-Hicks and Moore, 1998). Bartell and colleagues suggest that one important feature of the risk analysis is the explicit, quantitative consideration of uncertainties in the analysis and the expression of the final estimated effect as a probability (Bartell *et al.*, 1992).

While the type of quantitative estimates conducted by Moore *et al.* (1999) for risks to mink on the Clinch River would be useful, only the data for benthic infauna for the Lower Fox River were thought to be amenable to a quantitative analysis.

For this BLRA, quantitative uncertainty to PCB risks is expressed as the frequency of exposure to PCBs at effect levels known to cause harmful effects. While the point estimate calculation used a single selected toxicity reference value, the frequency estimates presented below evaluate the range of values listed in the literature (as compared to the frequency of occurrence of PCB concentration in sediments and fish tissues), since these are the principal routes of exposure for the receptors identified as being at risk.

Benthic Infauna

Those values, along with the ARCS SEC used in this BLRA, and the ARCS No Effects Concentration (NEC) are evaluated below to the frequency of occurrence of sediment PCBs within the Lower Fox River and Green Bay.

The data used to determine frequency of exposure to PCBs are the interpolated sediment grid values previously discussed in Section 2. The interpolated values provide a basis for evaluating risks over an entire reach or zone. The limits of those interpolations were discussed above. As noted previously, there are two ways of plotting the frequency of those interpolated values; either including grids for which no interpolation was provided and assigning a value of zero PCBs, or deleting those points from the interpolation.

Figure 6-84 shows the sediment PCB frequency distribution for Little Lake Butte des Morts. In addition, all of the sediment effects concentrations from Table 6-129 are plotted on Figure 6-84. This figure shows that depending upon the interpolation used, between 90 and 100 percent of the surface sediment concentrations of PCBs exceed the TEC. Alternatively, this could be expressed as

a 90 to 100 percent probability of a benthic infaunal organism in Little Lake Butte des Morts encountering PCBs in concentrations that exceed the TRV used in this BLRA. When evaluated against the consensus-based effects concentration, there is a 70 to 80 percent probability of encountering PCB levels that exceed the MEC, and between a 40 and 50 percent chance of encountering sediment PCB levels that exceed the EEC. Finally, within Little Lake Butte des Morts, at least 20 percent of the sediment PCB levels exceed 5,300 μg/kg, the Severe Effect Level (SEL). These data collectively demonstrate that there is a high probability (70 to 80 percent) that PCBs are widely distributed throughout the reach at sufficiently high concentrations to moderately effect benthic infaunal populations, and at least a 40 to 50 percent probability of encountering PCB concentrations associated with extreme effects.

The sediment PCB distribution of the Appleton to Little Rapids Reach is shown on Figure 6-85. Here, the distinction between the two applications of the interpolation (i.e., I_d and I_0) are clearly visible. Assuming that the non-interpolated grid values are equal to zero results in a much lower estimate of PCB distribution within the reach. Based on the I_0 interpolation, only about 10 percent of the area within the reach has sediment PCB concentrations that exceed the ARCS SEC, and only approximately 5 percent exceed the MEC. When evaluated based upon the I_d interpolation, those same percentages are approximately 65 and 35 percent, respectively. It has been noted previously that within this reach, there are long sections of fast-flowing water that scour the river bottom, where deposition of sediment PCBs would not occur. Thus, the more applicable interpolation for this reach is likely the I_0. For this reach, the probability of infaunal organisms encountering levels of PCBs associated with toxic effects are low (5 to 10 percent).

The Little Rapids to De Pere Reach is shown on Figure 6-86. As noted above, there is effectively no difference between the I_0 and I_d estimates of PCB concentrations in this reach. Based upon the ARCS SEC, the frequency of exceedance is greater than 90 percent. The frequency of exceedance of the ARCS NEC (high) is approximately 80 percent, approximately 60 percent exceed the consensus-based MEC, approximately 30 percent exceed the consensus-based EEC, and approximately 13 to 15 percent exceed the OMOE Severe Effect Level. Thus, there is a high probability (80 percent) that PCBs are widely distributed throughout the reach at sufficiently high concentrations to moderately impact benthic infaunal populations, and at least a 30 percent probability of encountering sediment concentrations associated with extreme effects.

De Pere to Green Bay (Zone 1) is shown on Figure 6-87. There is effectively no difference between the I_0 and I_d estimates of PCB concentrations in this reach.

Based upon the ARCS SEC, the frequency of exceedance is nearly 100 percent. The frequency of exceedance of the ARCS NEC (high) is approximately 95 percent, approximately 90 percent exceed the consensus-based MEC, approximately 60 percent exceed the consensus-based EEC, and approximately 13 to 15 percent exceed the OMOE Severe Effect Level. Thus, there is a high probability that PCBs are widely distributed throughout the reach at sufficiently high concentrations (95 percent) to moderately impact benthic infaunal populations, and at least a 60 percent probability of encountering sediments associated with extreme effects.

Green Bay Zone 2 is shown on Figure 6-88. Based upon the ARCS SEC, the frequency of exceedance is greater than 95 percent. The frequency of exceedance of the ARCS NEC (high) is greater than 70 percent, for the consensus-based MEC it is greater than 40 percent, approximately 25 percent exceed the consensus-based EEC, and less than 5 percent exceed the OMOE Severe Effect Level. Thus, there is a high probability that PCBs are widely distributed throughout the reach at sufficiently high concentrations (40 percent) to moderately impact benthic infaunal populations, and at least a 25 percent probability of encountering sediments associated with extreme effects.

Green Bay Zone 3A is shown on Figure 6-89. Based upon the ARCS SEC, the frequency of exceedance is greater than 85 percent. The frequency of exceedance of the ARCS NEC (high) is greater than 45 percent, for the consensus-based MEC it is approximately 30 percent, and the concentrations in this zone do not exceed either the consensus-based EEC or the OMOE Severe Effect Level. Thus, relative to the other reaches discussed, there is a moderate probability of encountering PCBs at sufficiently high concentrations (30 percent) to moderately impact benthic infaunal populations, but a 0 percent probability of encountering sediments associated with extreme effects.

Green Bay Zone 3B is shown on Figure 6-90. Based upon the ARCS SEC, the frequency of exceedance is greater than 95 percent. The frequency of exceedance of the ARCS NEC (high) is greater than 90 percent, for the consensus-based MEC it is greater than 60 percent, and the concentrations in this zone do not exceed either the consensus-based EEC or the OMOE Severe Effect Level. Thus, there is a high probability that PCBs are widely distributed throughout the reach at sufficiently high concentrations (60 percent) to moderately impact benthic infaunal populations, but a 0 percent probability of encountering sediments associated with extreme effects.

Green Bay Zone 4 is shown on Figure 6-91. Based upon the ARCS SEC, the frequency of exceedance is between 50 percent (I_0) and 60 percent (I_d). There

were no exceedances of the ARCS NEC (high), the consensus-based MEC, the consensus-based EEC or the OMOE Severe Effect Level. Thus, there is only a very a low probability that PCBs are widely distributed throughout the reach at sufficiently high concentrations to impact benthic infaunal populations.

6.7 Risk Management Integration of the Ecological Risk Assessment

This section provides an integration of the presented information in the ERA by reach and zone. This section brings together the risk characterization (Section 6.5.3), the available population information (Section 6.5.4), and the uncertainties (Section 6.6) in a way that can be used by managers to make risk decisions. This risk assessment fulfills the NRC (2001) recommendation that sites be evaluated using a scientific risk-based framework so that different approaches for remediating PCB-contaminated submerged sediments can be compared in terms of the efficacy and human and ecological risks associated with each approach. The BLRA essentially evaluates risk assuming a no action remedial alternative. Relative risks associated with other potential remedial actions are discussed in the Feasibility Study.

Data and supporting information are arranged below in a format designed to answer the specific risk questions for each reach and zone examined.

Little Lake Butte des Morts Reach

The risk summary by area is given in Section 6.5.3; the reader is referred to that section for details. A brief summary of the reach risks are included in the sections below.

Water Column Invertebrates. As discussed in Section 6.5.3, the HQs for mercury and PCBs indicate risk. The HQs are based on limited data sets; six values for mercury and ten for PCBs (see Tables 6-8 and 6-9). Detected maximums were used for the risk calculation. PCBs were only detected in three of ten samples. Site-specific toxicity testing information for the Lower Fox River presented in Section 6.5.4 suggest that pore water toxicity to water column species was observed, but was principally thought to be associated with ammonia.

These results do not preclude low-level risks to individuals or specific species within the community, but do suggest that adverse alterations to the functioning water column invertebrate communities should not be expected.

Benthic Invertebrates. HQ risks are indicated for lead, mercury, 2,3,7,8-TCDD, PCBs, DDD, and DDT. Total PCB HQs are 50 times greater than any other COC.

Benthic infaunal populations within Little Lake Butte des Morts, while improved over conditions in the 1960s, remain generally low in total numbers and diversity. While the numbers and species present appear to be sufficient to support the resident fish population, generally the taxa present represent "pollution-tolerant" species. While bedded sediment bioassays conducted in other parts of the river suggest that ammonia may play a role in suppressed infaunal populations, this was found to account for only part of the toxicity observed.

A quantitative analysis of uncertainty in Section 6.6.8 indicates that there is a high probability of benthic organisms encountering PCBs that exceed toxic levels. Over 90 percent of the surface sediment area in this reach exceeds the threshold effects concentration, while at least 40 percent of surface sediment exceeds concentrations associated with extreme effects.

These data, taken in total, support the premise that site contaminants in sediment are sufficient to cause adverse alterations to the functioning of benthic invertebrate communities.

Benthic Fish. The risk characterization indicates the total PCB concentrations in benthic fish pose potential risk, while measured coplanar concentrations do not. There is a greater degree of confidence in the larger data set for total PCBs (n = 30) (Table 6-14) than for the coplanar PCBs (n = 3).

There is information that some benthic fish (e.g., carp and bullhead) have maintained self-reproducing populations throughout the river, even during periods when there were active PCB inputs. While recognizing that there may be differences between the two systems, bioassays conducted on cyprinids and Medaka exposed to sediments and pore water from Zone 1, where PCB HQs are higher, failed to demonstrate acute or chronic toxicity. Collections of bullheads from Zone 1, where PCB tissue concentrations in fish are higher, showed a complete absence of cancer-like lesions.

For species for which there are available information, population impacts do not appear to be occurring. Given that these observations are for a limited number of species, there remains potential risk from PCB exposure.

Pelagic Fish. The risk characterization found that there are potential risks to some pelagic fish, but not all. No risks are indicated from coplanar PCBs, although these data sets are limited. Self-sustaining walleye and perch populations are now found throughout the river, suggesting that there are no population level impacts for at least these species. While recognizing that there may be differences between the two systems, liver tumors and pre-tumors were observed on walleye

collected in zones 1 and 2. Walleye tissue PCB concentrations in the Little Lake Butte des Morts Reach are on average approximately six times lower than those from fish collected in zones 1 and 2. It is not known if those same liver lesions exist on fish in the Little Lake Butte des Morts Reach.

For species for which there are available information, population impacts do not appear to be occurring. Given that these observations are for a limited number of species, there remains potential risk from PCB exposure.

Insectivorous Birds. The risk characterization found that levels of PCBs and PCB congeners are at sufficient levels to potentially cause survival or reproductive impairment in insectivorous birds. A field evaluation of reproductive impairment of tree swallows conducted at Deposit A in Little Lake Butte des Morts concluded that there are no discernible effects of PCBs on nesting behavior, clutch size, hatching success, or deformity. While this study is limited in size, it was designed to examine effects of organochlorines on insectivorous birds, increasing the confidence in the conclusion.

The data suggest that levels of site contaminants are not sufficient to cause survival, reproductive impairment, or deformity in populations of insectivorous birds in Little Lake Butte des Morts.

Piscivorous Birds. Modeled dietary intakes for piscivorous birds indicate that some, but not all species, have potential risk from total PCBs and mercury. These conclusions are based upon a limited data set ($n = 2$ for fish tissue concentration), so there is not a high degree of confidence in these HQs.

The status of reproducing tern populations is indeterminate at this time. Increases in the numbers of observed breeding pairs has risen in the region. The field observation data presented suggest that PCBs may not be associated with adverse effects on endpoints such as hatching success and deformities, but toxicity in terns may manifest as sublethal or behavioral changes. Field data for double-crested cormorants clearly show population recovery and implicate DDE and PCBs for past declines. Given that Little Lake Butte des Morts has limited suitable habitat to support nesting sites for some species, which may reduce exposure, it is likely that they use this river reach for resting and foraging.

These data suggest collectively that there may yet be potential risks to survival or reproductive impairment to piscivorous birds from exposure in the Little Lake Butte des Morts Reach to some species.

Carnivorous Birds. The risk characterization found that potential risks to survival and reproduction are indicated. Bald eagles throughout the Lower Fox River and Green Bay are recovering and the data suggest that productivity is better for nests along the Lower Fox River than it is for nests along the border of Green Bay. Eagles are known to forage within Little Lake Butte des Morts and will remain in the area for long periods of the year. The sole eagle nest proximal to this reach on Mud Creek was first successfully occupied in 1994, producing two hatchlings that year, and three the following year.

The modeled, diet-based NOAEC HQs are biased toward a high risk estimate because of the assumption that the eagles consume a 100 percent fish diet, and that 100 percent of the diet is obtained in the Little Lake Butte des Morts Reach. There is no information to determine whether the eagles are foraging exclusively in the Little Lake Butte des Morts Reach, or may be taking a portion of their prey in adjacent, uncontaminated Lake Winnebago.

Given the eagle's special status, the elevated NOAEC HQs based on fish tissue concentrations of PCBs, and the absence of unambiguous site-specific exposure and effect data, carnivorous birds are estimated to be at risk to survival or reproductive impairment or deformity.

Piscivorous Mammals. Modeled dietary intake for piscivorous mammals suggest that PCBs are at sufficient levels to present risk to reproduction and survival.

There is a high degree of certainty associated with the effects of PCBs on piscivorous mammals. The exposure assessment may be biased high by limiting the mink diet to carp only.

Habitat suitable to support mink was identified in the Little Lake Butte des Morts Reach, but there are no capture or population level data for either mink or other piscivorous mammals such as river otter.

Based upon information available, piscivorous mammals are deemed to be at risk from PCB exposure.

Appleton to Little Rapids Reach

Water Column Invertebrates. Only PCBs were identified as causing potential risks to water column invertebrates, but this was based upon an HQ of 1.2 for the NOAEC.

The same pore water bioassays conducted by Ankley *et al.* (1990) on water column invertebrates are relevant here as well. These results do not preclude low-

level risks to individuals in the community, but do suggest that adverse alterations to the functioning water column invertebrate communities are not expected.

Benthic Invertebrates. Concentrations of lead, mercury, and total PCBs in sediments are at levels sufficient to cause adverse impacts to benthic invertebrates.

Within those deposits examined, benthic infaunal populations within this reach remain generally low in total numbers and diversity. Deposit N, which was examined in 1994, has been removed and the benthic data from that site are no longer relevant. While the numbers and species present appear to be sufficient to support the resident fish population, generally the taxa present represent "pollution-tolerant" species.

Sediment isopleths for this reach (Figure 2-3) show that PCB distribution within this reach is not widespread and is confined to relatively limited specific deposits. The quantitative analysis of uncertainty suggests that only between 5 and 10 percent of the area within the reach has sediment total PCB concentrations that exceed the toxic threshold. Thus, while these data support the premise that PCBs are sufficient to cause adverse alterations to the functioning of benthic invertebrate communities, these risks are likely associated with the individual deposits.

Benthic Fish. Potential risks from total PCBs were indicated for benthic fish, based upon the HQ.

Carp and bullheads have maintained self-reproducing populations throughout the river, even during periods when there were active PCB inputs. Bioassays conducted on Medaka exposed to sediments from Zone 1, where PCB HQs are higher, failed to demonstrate acute or chronic toxicity. Collections of bullheads from Zone 1, where PCB tissue concentrations in fish are higher, showed a complete absence of cancer-like lesions.

For species for which there is available information, population impacts do not appear to be occurring. Given that these observations are for a limited number of species, there remain potential risks from PCB exposure.

Pelagic Fish. The risk characterization found that there are potential risks to pelagic fish from total PCBs. No risks are indicated based upon the coplanar PCBs. There is an equivalent degree of confidence in both measurements (n = 3 to 4 for both HQs).

As noted previously, self-sustaining walleye and perch populations are now found throughout the river, suggesting that there are no population level impacts for these species. While recognizing that there may be differences between the two systems, the potential exists that low levels of PCBs may induce liver tumors and pre-tumors, as observed in walleye collected in zones 1 and 2. While walleye tissue PCBs in this reach are somewhat lower than those from fish collected in zones 1 and 2, the sum of dioxin-like congeners is significantly lower.

For species for which there are available information, population impacts do not appear to be occurring. Given that these observations are for a limited number of species, there remains potential risk from PCB exposure.

Insectivorous Birds. Data were not available for the estimation of risk to insectivorous birds.

Piscivorous Birds. Based upon modeled dietary intake of COCs, mercury and total PCBs are estimated to be at sufficient concentrations to cause potential adverse effects on survival, physiology, or reproduction of piscivorous birds.

There are no suitable habitats for terns or cormorants in this reach, although it is likely that the reach is used for resting or foraging. As noted previously, field observation data suggest that PCBs may not be associated with adverse effects on endpoints such as hatching success and deformities, but toxicity in terns may be manifest as sublethal or behavioral changes. Field data demonstrate that cormorant populations in the region have recovered, but that trends for tern populations are not discernible. Given that the Appleton to Little Rapids Reach does not provide suitable habitat to support nesting sites for either terns or cormorants, it is likely that the HQ estimated risks are overestimated and the actual risks would need to be determined based upon some use of the reach for resting and foraging.

These results suggest that there is low survival or reproductive impairment to piscivorous birds from exposure in the Appleton to Little Rapids Reach. However, some sublethal effects may still occur. Therefore, the potential for risk exists.

Carnivorous Birds. Measured concentrations of mercury and PCBs indicate potential risk to bald eagles. However, the confidence in this assessment is low as it is based on only a single sample. Modeled intake also suggests potential risks. The confidence in the modeling estimate is limited in that it assumes 100 percent foraging within the reach and a diet exclusively composed of fish.

Confidence in the risk characterization based on measured tissue concentrations is low based on the fact that measured concentrations of total PCBs is from a single eagle egg collected in 1990, the NOAEC HQ is 45.

Nestling blood samples collected annually from 1991 through 1994 showed persistently elevated levels of PCBs. However, lower levels of dioxin TEQs have been measured in the Kaukauna eagles, which would result in lower levels of toxicity. This is supported further by TEQs calculated for measured fish tissue concentrations of PCB coplanar congeners in this reach, which are an order of magnitude lower than TEQs calculated in lower Green Bay.

One of the oldest and most successful nesting sites within the Lower Fox River is at Kaukauna. The productivity of that nest has been on average two to three young per year since 1988. Given the apparent breeding success at this site, it could be concluded that overall levels of site contaminants may not be sufficient to cause any apparent survival or reproductive impairment. However, there is insufficient information available to determine if eagles are foraging exclusively within this reach, or may be taking their prey from uncontaminated sources such as Lake Winnebago.

Given the eagle's special status, the elevated HQs based on fish tissue mercury and PCBs, and the absence of site-specific exposure data, risks to carnivorous bird survival, reproductive impairment, or deformity are likely low, but cannot be ruled out.

Piscivorous Mammals. Modeled dietary intake for piscivorous mammals suggest that PCBs are at sufficient levels to present risk to reproduction and survival.

There is a high degree of certainty associated with the effects of PCBs on piscivorous mammals. The exposure assessment may be biased high by limiting the mink diet to carp only.

Habitat suitable to support mink was identified in the Appleton to Little Rapids Reach, but there are no capture or population level data for either mink or other piscivorous mammals such as river otter.

Based upon the estimated dietary intakes, site contaminants are estimated to be sufficient to cause survival or reproductive impairment to piscivorous mammals.

Little Rapids to De Pere Reach

Water Column Invertebrates. Mercury is the only chemical that poses risks to functioning water column invertebrate communities in the Little Rapids to De

Pere Reach. However, it should be noted that this risk estimation is based upon only two detections out of three samples for the whole reach. Therefore, the confidence in this interpretation of risk is low. Total PCBs are not at sufficient concentrations to pose risk. Confidence is high in the interpretation of no risk from total PCBs, as there were 97 detections in 98 filtered samples, all with calculated HQs of less than 1.0.

Benthic Invertebrates. The risk characterization determined that persistent risks to infaunal communities exist from levels of lead, mercury, TCDD, total PCBs, DDE, and DDT in sediments. Given the number of samples collected in this reach, there is a high degree of confidence in this finding for all COCs, except TCDD which was based on only two samples.

Benthic infaunal populations within the Little Rapids to De Pere Reach are generally low in total numbers and diversity. While the numbers and species present appear to be sufficient to support the resident fish species, generally the taxa present represent "pollution-tolerant" species. While bedded sediment bioassays conducted in other parts of the river suggest that ammonia may play a role in suppressed infaunal populations, this was found to account for only a part of the toxicity observed.

Sediment isopleths (Figure 2-4) show that elevated surface sediment concentrations are widely distributed throughout the reach. The quantitative uncertainty analysis indicated that there is an 80 percent probability of encountering PCBs in sediments associated with moderate effects, and a 30 percent probability of exceeding an extreme effect level.

These data, taken in total, support the premise that site contaminants in sediment are sufficient to cause adverse alterations to the functioning of benthic invertebrate communities.

Benthic Fish. Risks from PCBs and mercury to fish are indicated for this reach based on the HQs. It should be noted that the estimation of risk from mercury is based on a single carp tissue sample, and thus confidence in this estimation is low. While the calculated NOAEC HQ for total PCBs is based on 20 detections in 20 measured fish (HQ = 7.6), it should be noted that the LOAEC HQ is less than 1.0, and that TEQ HQs calculated from measured dioxins and PCB coplanar congeners are equal to or less than 0.2.

As with the previous two reaches, carp have maintained self-reproducing populations within the Little Rapids to De Pere Reach. Bioassays conducted on cyprinids and Medaka (Call *et al.*, 1991; Ankley *et al.*, 1992) exposed to sediments

and pore water from Zone 1, where PCB HQs are higher, did not demonstrate acute or chronic toxicity from PCB exposure. Collections of bullheads from Zone 1, where PCB tissue concentrations in fish are higher, showed a complete absence of cancer-like lesions.

The potential for risk to benthic fish exists. However, it appears that some less sensitive species are persisting in this reach. The results collectively suggest that levels of site contaminants are not sufficient to cause apparent reproductive or survival impairment to benthic fish. While population impacts do not appear to be occurring, there remains the potential for risk to individuals or sublethal effects.

Pelagic Fish. Potential risks from PCBs and mercury were indicated for pelagic fish in this reach. The TEQ HQs for measured dioxins and coplanar congeners indicate no risk. There are limited data for these fish ($n \leq 4$); therefore, confidence in the risk characterization is low.

As noted before, the presence of self-reproducing populations of walleye and perch throughout the river suggest that there are no population level impacts for these species. Liver tumors and pre-tumors observed on walleye collected in zones 1 and 2 are a potential concern for fish in the Little Rapids to De Pere Reach. While both walleye and perch tissue PCBs in this reach are approximately equal to those observed in zones 1 and 2, the calculated TEQs are an order of magnitude lower than TEQs measured in the same fish in lower Green Bay.

Given that these observations are for a limited number of species, there remains the potential for risk from PCB exposure.

Insectivorous Birds. Data are not available for the estimation of risk to insectivorous birds.

Piscivorous Birds. Modeled dietary intake for all species indicate potential risks for exposure to mercury and total PCBs. These conclusions are based upon a limited data set ($n \leq 3$) and thus there is not a high degree of confidence in these estimated risks. While dietary intake of TEQs was not modeled, it is noted that the measured total TEQs for the modeled prey fish, golden shiner, is an order of magnitude lower than total TEQs in the same fish species from zones 1 and 2.

While there are no tern or cormorant nesting areas in this reach, it is likely that the reach is used for resting or foraging for all three species. Given its proximity to Zone 2 where nesting colonies do exist, at least some exposure to COCs in this reach is expected. As noted previously, field observation data suggest that PCB

toxicity in terns may be manifest as sublethal or behavioral changes. Given that those sublethal or behavioral effects were observed in Green Bay colonies and have been attributed to elevated TEQs, the fact that the TEQs in forage fish species are an order of magnitude lower in the Little Rapids to De Pere Reach suggests that those effects would not be expected from dietary intake of fish in the present reach.

These results taken collectively suggest that there may be potential risks to survival or reproductive impairment to piscivorous birds from exposure in the Little Rapids to De Pere Reach.

Carnivorous Birds. The risk characterization found that potential risks to carnivorous birds from mercury and PCB exposure exist. As noted previously, the limited number of measured endpoints used in the modeling does not impart a high degree of confidence in these estimations.

There are no reported eagle nests or forage sites within this reach, although it remains within the foraging range for both the Kaukauna eagles, as well as the nest on the East River within the Zone 1 basin. Given that foraging occurs in all other reaches of the river, it is likely that at least some fish are taken within this reach. While dietary intake of TEQs was not modeled, it is noted that the measured total TEQs for potential prey species (walleye and carp) are at similar levels to fish collected in the Appleton to Little Rapids Reach, while an order of magnitude lower than those same species collected from zones 1 and 2. Thus, it can be inferred that exposure and risks would be similar to those determined for the Appleton to Little Rapids Reach.

The modeled, diet-based NOAEC HQs are biased toward a high risk estimate because of the assumption that the eagles consumer a 100 percent fish diet, and that 100 percent of the diet is obtained in this reach. There is no information to determine whether the eagles foraging exclusively in this reach may be taking a portion of their prey from the upstream or downstream reaches.

Given the eagle's special status, the elevated NOAEC HQs based on fish tissue concentrations of PCBs, and the absence of unambiguous site-specific exposure and effect data, carnivorous birds are estimated to be at risk to survival or reproductive impairment or deformity.

Piscivorous Mammals. Modeled dietary intake for piscivorous mammals suggest that PCBs are at sufficient levels to present risk to reproduction and survival.

There is a high degree of certainty associated with the effects of PCBs on piscivorous mammals. The exposure assessment may be biased high by limiting the mink diet to carp only.

Habitat suitable to support mink was identified in this reach, but there are no capture or population level data for either mink or other piscivorous mammals such as river otter.

Based upon information available, piscivorous mammals are deemed to be at risk from PCB exposure.

De Pere to Green Bay Reach (Green Bay Zone 1)

Summaries to fish and piscivorous birds in this zone of Green Bay are presented with the results for Green Bay Zone 2, because the areas are not distinct for the purposes of assessing risk to the fish and bird assessment endpoints.

Water Column Invertebrates. The risk characterization found the potential for risk from exposure to PCBs. The confidence in estimating an HQ is high given the number of water samples in this reach ($n \approx 140$). While the HQs for total PCBs indicate risk, the pore water bioassays conducted by Ankley *et al.* (1990) on water column invertebrates from sediments collected in this reach and Zone 2 of Green Bay suggest there are no observed acute or chronic effects that are attributable to PCBs. These results do not preclude low-level risks to individuals in the community, but adverse alterations to the functioning water column invertebrate communities are not expected.

Benthic Invertebrates. Persistent risks exist to benthic infauna from exposure to PCBs, DDD, DDE, arsenic, lead, and mercury. The confidence in these estimates is high based on the number of samples collected for metals (92) and PCBs (290). However, for chlorinated pesticides there is less confidence as there were only two detections in 22 samples.

Benthic infaunal populations within Zone 1 have been well documented over time and generally remain low in total numbers and diversity. While the numbers and species present appear to be sufficient to support the resident fish population, the taxa present represent "pollution-tolerant" species, and the numbers of taxa present are lower than unimpacted reference sites. Bedded sediment bioassays conducted in this river suggest that elevated ammonia levels play a role in suppressed infaunal populations, but this was found to account for only a part of the toxicity observed (Ankley *et al.*, 1992; Call *et al.*, 1991).

The quantitative uncertainty analysis indicates that the threshold effects concentrations are exceeded in 90 percent of the surface sediment area, and a 30 percent probability of encountering sediments that are associated with extreme effects.

These data suggest that site contaminants in sediment are sufficient to cause adverse alterations to the functioning of benthic invertebrate communities.

Benthic Fish. Benthic fish are discussed under Zone 2, below.

Pelagic Fish. Pelagic fish are discussed under Zone 2, below.

Insectivorous Birds. The risk characterization found potential risks from PCB exposures.

Two separate evaluations of reproductive impairment of tree swallows and red-winged blackbirds within Zone 1 concluded that there are no discernible effects of PCBs on nesting behavior, clutch size, hatching success, or deformity in those species. The confidence in these studies is high as there were an adequate number of samples (22) and that the studies were designed to specifically look at the effects of exposure to chlorinated hydrocarbons.

These data suggest that levels of site contaminants are not sufficient to cause survival or reproductive impairment or deformity in populations of insectivorous birds in Zone 1.

Piscivorous Birds. Risks to piscivorous birds are discussed under Zone 2, below.

Carnivorous Birds. Risks to carnivorous birds are discussed under Zone 2, below.

Piscivorous Mammals. Modeled dietary intake for piscivorous mammals suggest that PCBs are at sufficient levels to present risk to reproduction and survival.

There is a high degree of certainty associated with the effects of PCBs on piscivorous mammals. The exposure assessment may be biased high by limiting the mink diet to carp only.

Habitat suitable to support mink was identified in this reach, but there are no capture or population level data for either mink or other piscivorous mammals such as river otter.

Based upon information available, piscivorous mammals are deemed to be at risk from PCB exposure.

Green Bay Zone 2

Risks to fish and piscivorous birds exposed in zones 1 and 2 of Green Bay are presented and discussed here.

Water Column Invertebrates. Based on calculated HQs, only mercury is at concentrations that are posing risk to the functioning of water column invertebrate communities. However, it is noted that mercury was only detected in 2 of 11 samples and that the HQs are calculated from the maximum value measured. Thus, risks from mercury are potentially overestimated.

Benthic Invertebrates. The risk characterization indicated risks to benthic infauna from PCBs and mercury. Confidence in these estimates are very low given that there are only 15 measured values for all of Zone 2. In addition, many of these samples were collected in 1988–1989 as part of the GBMB study. The current sediment concentration is unknown.

The benthic populations of southern Green Bay are dominated by oligochaetes and chironomids with an absence of the more pollution-tolerant species such as gammarid amphipods that are present in the northern parts of the bay. Bioassays conducted on sediments collected in Zone 2 showed toxicity, although at least a portion of that could be attributed to ammonia toxicity.

These data suggest that PCBs and mercury in sediment are sufficient to cause adverse alterations to the functioning of benthic invertebrate communities.

Benthic Fish. The risk characterization found that concentrations of total PCBs and p,p'-DDE are high enough to pose potential risk to benthic fish. However, TEQs based upon dioxins and PCB congeners are less than 1.0. Confidence in these estimations for carp is high; there are a total of 115 samples collected for PCBs and 13 for DDE. Confidence in PCB congeners estimates are also high (14 to 80 detections), but there are only three measurements for dioxins.

There is information that certain benthic fish (carp and bullhead) have maintained self-reproducing populations throughout the river and bay, even during periods when there were active PCB inputs. Bioassays conducted on cyprinids and Medaka exposed to sediments and pore water from Zone 1 failed to demonstrate acute or chronic toxicity that was attributable to the presence of PCBs. Collections of bullheads from within zones 1 and 2 showed a complete absence of cancer-like lesions. For these species, there is ample evidence that

population impacts are not occurring. For species for which there are available information, population impacts do not appear to be occurring. Given that the observations are for a limited number of species, there remains potential risks from PCB exposure.

Pelagic Fish. Concentrations of total PCBs were found in the risk characterization to pose potential risks to all pelagic species evaluated; walleye, perch, alewife, gizzard shad, smelt, and both shiner species. However, when examined on a TEQ basis, only walleye were found to be at potential risk. Confidence in these estimates is high as there is a combined 246 analyses of pelagic fish.

Self-sustaining walleye populations are now found throughout the river, suggesting that there are no impairments to population level impacts for this species. Forage fish communities appear to be sufficient to support the piscivorous fish and bird populations of Green Bay. Sublethal effects in the form of liver tumors and pre-tumors were observed on walleye collected in zones 1 and 2.

While all species of pelagic fish examined do not appear to exhibit population-level effects, pelagic fish as a group may be at risk to sublethal effects from PCBs in zones 1 and 2.

Insectivorous Birds. Concentrations of total PCBs, PCB congeners, and p,p'-DDE are at levels that could cause potential risk.

Field evaluations of reproductive impairment of tree swallows conducted using nest boxes at Kidney Island in 1988, 1994, and 1995 concluded that there were no discernible effects of PCBs on nesting behavior, clutch size, hatching success, or deformity as previously discussed. The studies conducted were specifically to examine risk to insectivorous birds, looked at appropriate population parameters, and had adequate reference sites. Thus, confidence in the interpretation that effects were not observed is high.

These data collectively suggest that insectivorous birds are not at risk.

Piscivorous Birds. The risk characterization found that modeled dietary intake of total PCBs, mercury, and p,p'-DDE are sufficiently high to cause adverse effects to these bird species. There are sufficient whole body and egg measurements (774 and 34, respectively) for pelagial birds as a group to be confident in the risk estimates.

The historic levels of PCBs and DDE clearly impacted these birds at all levels. There is ample field evidence to suggest that the presence of organochlorines in

the diets and tissues of piscivorous birds have had population, reproductive, and deformity impacts. There are multiple lines of evidence in carefully conducted field studies within the Fox River and Green Bay that corroborate the fact that organochlorine residues (PCBs and DDE) are correlated with a range of effects including infertility, embryo mortality, and deformities. Deformities such as crossed bills, edema, unabsorbed yolk sac, and dwarfism are characteristic of abnormalities observed as a result of exposure to polychlorinated hydrocarbons such as DDE and PCBs. Furthermore, there is evidence to suggest that more subtle effects, such as natal regional fidelity, may manifest as a result of exposure to organochlorines.

The cormorant is now more numerous on the Great Lakes than at any time in its previously recorded history. Once on the state's endangered species list, the cormorant was delisted in 1986. Double-crested cormorants on Cat Island in 1995 did not appear to experience any survival effects, reproductive effects, or deformities as a result of increased contaminant levels. Tern populations also appear to be recovering, although this trend has not been fully documented, and the Forster's tern remains on the threatened species list.

For terns, there are indications that risks remain from PCBs and DDE. There are persistent elevated levels of PCBs and DDE at toxic levels associated with reproductive effects. While field data collected by the USGS and USFWS for Zone 2 suggest that PCBs may not be associated with hatching success and deformities, there are no data to corroborate whether threatened tern populations have fully recovered. Persistence of sublethal effects such as site tenacity indicate that some subtle reproductive effects may be manifesting themselves in Green Bay tern populations as a result of exposure to PCB contamination.

For species for which there are available information, population impacts do not appear to be occurring. The weight of evidence support a conclusion that levels of organochlorines in piscivorous birds remain sufficiently high to pose risks to reproduction and deformities.

Carnivorous Birds. The risk characterization found that carnivorous birds foraging in zones 1 and 2 are potentially at risk from exposure to total PCBs, mercury, and p,p'-DDE.

While not specifically modeled or measured for eagles, it is relevant to note that measured TEQs calculated for fish tissue concentrations show elevated levels of PCB coplanar congeners in this reach, which are also present in the piscivorous bird whole body and eggs. The TEQs measured in fish tissue in zones 1 and 2 are

an order of magnitude higher than those observed in the Appleton to Little Rapids Reach, where eagles have successfully nested and bred for over 10 years.

There is a potential high bias introduced by limiting the eagle diet to fish. Eagles are opportunistic foragers and will also feed on small mammals and birds, especially gulls.

The existing population and field data suggest that the reproductive rates of nesting bald eagles in Green Bay, while recovering, are generally depressed relative to inland areas of both Wisconsin and Michigan. Within Zone 1 and Zone 2, there are only two known nests; one just up the East River and a nest at Little Tail Point. Those nests did not exist until 1993–1994, and only a single successful hatch has occurred at the Little Tail Point nest. By contrast, successful nesting has been recorded in both the Appleton to Little Rapids Reach and in Zone 4 since 1988; suggesting a continuing depression on successful breeding in zones 1 and 2.

The weight of evidence collectively support a conclusion that elevated organochlorine levels in prey continue to pose risk to survival and reproduction of carnivorous birds in zones 1 and 2 of Green Bay.

Piscivorous Mammals. Based upon the estimated dietary intakes, total PCBs were estimated to be sufficient to cause survival or reproductive impairment to piscivorous mammals.

There are no measured tissue data within the Fox River database, but two field collections of mink produced conflicting results. One collection in the late 1980s had levels exceeding 5,700 μg/kg in mink liver, while a 1998 collection showed 40.4 μg/kg in mink liver. While this might suggest PCB levels have dropped several orders of magnitude in 10 years, these are only two data points and may not provide an adequate picture of contaminant distribution in mink of the region.

Habitat suitable to support mink was identified in Zone 2, but there are no capture or population level data for either mink or other piscivorous mammals such as river otter.

Based upon these data, piscivorous mammals are judged to be at risk in this zone.

Green Bay Zone 3A

Water Column Invertebrates. Neither mercury nor total PCB concentrations are sufficient to pose risk to water column invertebrate communities, based on

calculated HQs that are all 0.1 or less. Although these were the only two COPCs analyzed in this zone, analyses in the other river reaches and Green Bay zones suggest that these are the COPCs that are the most likely to have HQs of greater than 1.0.

Benthic Invertebrates. Only total PCBs were found to pose risks to benthic invertebrates in Zone 3A. Confidence in these estimates is low given that there are only 15 measured values of PCBs for all of Zone 3A. In addition, many of these samples were collected in 1988–1989 as part of the GBMB study. The differences between current concentrations and those 1989 values is unknown.

Distribution of total PCBs within Zone 3A is such that elevated levels of PCBs exist in the southern part of the zone, with decreasing concentrations in the north (Figure 2-6). This may be consistent with the observations that within Zone 3A communities north of the Peshtigo Reef have greater species richness and total numbers of organisms than in the inner bay. Gammarid amphipods (*Gammarus fasciatus* and *Pontoporeia hoyi*) first appear in abundance in the northern part of the reach and are principally absent from the lower parts. South of Peshtigo Reef, the communities are still dominated principally by oligochaetes and chironomids. While it is likely that PCBs are impacting infaunal populations within the southern part, changes in the physical habitat conditions in the bay north of Peshtigo Reef likely also contribute to the changes in community structure.

These data suggest that contaminants in sediment are sufficient to cause adverse alterations to the functioning of benthic invertebrate communities.

Benthic Fish. Risk from exposure to total PCBs was indicated for benthic fish in the zone. However, this determination is based upon a relatively limited number of samples so confidence in this is low.

Discrepancies on the sublethal carcinogenic effects of COCs in benthic fish from the Lower Fox River (Green Bay Zone 1) are reported by USFWS and EPA personnel (Ankley *et al.*, 1992). It is important to note that Zone 1, where the carcinogenic effects were investigated, had higher PCB concentrations in benthic fish than the benthic fish from Green Bay Zone 3A.

These results collectively suggest that levels of site contaminants are not sufficient to cause apparent reproductive or survival impairment to benthic fish. However, there remains the potential for sublethal adverse effects. Therefore, it is concluded that benthic invertebrate fish have potential risk from total PCBs.

Pelagic Fish. Concentrations of total PCBs in pelagic fish are sufficiently high to potentially be of risk to pelagic fish reproduction or survival. The PCB risk conclusions are not in agreement for total PCBs and PCB congeners. Additionally, when looked at on a species-by-species basis, HQs for rainbow smelt are 1.0 or less, HQs for alewife are between 1.0 and 2.0, HQs for brown trout are between 4.0 and 5.0, and HQs for walleye are between 5.0 and 7.0. In the recently released NRDA evaluation of walleye, USFWS concluded that fish from Green Bay had a significantly higher incidence of liver tumors and pre-tumors. While effects on survival and reproduction in walleye were not investigated as part of the NRDA evaluation, both the HQs and sublethal effects suggest that PCBs are potentially causing risk to pelagic fish reproduction or survival.

Insectivorous Birds. Data were not available for the estimation of risk to insectivorous birds.

Piscivorous Birds. Estimated concentrations of total PCBs and mercury in the diet pose potential risks to piscivorous birds.

Productivity data for these species was previously presented. Double-crested cormorant populations have been increasing in Green Bay while the status of common and Forster's terns remains more uncertain. The dietary TRVs used to estimate risk from COCs were based on survival and reproduction, but do not indicate the likelihood of deformities. NRDA investigations, however, suggest that physical deformities are occurring in piscivorous birds and that, at least for Forster's terns, these deformities are likely caused by PCBs.

These data suggest that for this area piscivorous birds are potentially at risk from concentrations of mercury and total PCBs.

Carnivorous Birds. Carnivorous birds are estimated to be at risk from total PCBs and potentially at risk from dieldrin.

It should be noted that the actual bald eagle data for total PCBs and dieldrin are based on only one sample. Also, estimated concentrations in bald eagles are based only on fish consumption while bald eagles likely also include bird consumption as a minor part of their diet. Population data suggest that bald eagle productivity within Green Bay is still much less than productivity in the Lower Fox River and inland Wisconsin. The USFWS NRDA investigation concluded that reduced hatching success and productivity for bald eagles are highly likely and that these adverse effects are likely the result of exposure to PCBs.

These data suggest that carnivorous birds are at risk from total PCBs and potentially at risk from dieldrin. Given the special status of the bald eagle, they are considered to be at risk from both COCs.

Piscivorous Mammals. Piscivorous mammals were estimated in the risk characterization to be at risk from exposure to total PCBs and potential risk from exposure to dieldrin.

Habitat suitable to support mink was identified in this zone, but there are no capture or population level data for either mink or other piscivorous mammals such as river otter.

Based upon the available information, piscivorous mammals are deemed to be at risk.

Green Bay Zone 3B

Water Column Invertebrates. No risks were identified for these receptors in this zone.

Benthic Invertebrates. The risk characterization found that benthic communities are at risk from exposure to arsenic, lead, mercury, and total PCBs. Confidence in these estimates are low. There are only four measurements for the metals; mercury was detected in only one sample. For total PCBs, the data set consists of 40 samples collected as part of the 1988–1989 GBMB study.

The HQs for PCBs are at least 10 times greater than the HQs for other COCs. Benthic invertebrate community investigations of Green Bay zones 1 and 2 indicate that benthic invertebrate communities are impacted by COCs and ammonia. Therefore, these data suggest that site contaminants in sediment are sufficient to cause adverse alterations to the functioning of benthic invertebrate communities.

Benthic Fish. There are no risks indicated.

The conclusion is based on the collection of only one benthic fish in this zone, but results for other river reaches and Green Bay zones also suggest that levels of site contaminants are not sufficient to cause reproductive or survival impairment to benthic fish.

Pelagic Fish. Potential risks were indicated for exposure to total PCBs and mercury. This includes exposure for walleye and brown trout.

These data suggest that total PCBs are posing risk to pelagic fish, but likely these fish are at risk for sublethal effects.

Insectivorous Birds. Data were not available for the estimation of risk to insectivorous birds.

Piscivorous Birds. Although both the population data and estimated HQs suggest that double-crested cormorants are experiencing less impairment than terns, as a group, piscivorous birds are assumed to be at risk from total PCBs and mercury, and potentially at risk from dieldrin and p,p'-DDE.

Carnivorous Birds. Estimated concentrations of mercury, total PCBs, and p,p'-DDE in the diet of carnivorous birds are sufficient to cause potential adverse effects on survival, physiology, or reproduction. Estimated concentrations of dieldrin are not sufficient to pose risk. There are no measured HQs for carnivorous birds in this area, but these results concur with the results for piscivorous birds. Because of the federal status of the bald eagle, carnivorous birds are assumed to be at risk from these COCs.

Piscivorous Mammals. Modeled concentrations of total PCBs were estimated to pose risk to mink.

Habitat suitable to support mink was identified in this zone, but there are no capture or population level data for either mink or other piscivorous mammals such as river otter.

Based upon the estimated dietary intakes, site contaminants are estimated to be sufficient to cause survival or reproductive impairment to piscivorous mammals.

Green Bay Zone 4

Water Column Invertebrates. Neither mercury nor total PCB concentrations are sufficient to pose risk to water column invertebrate communities, based on calculated HQs that were all 0.1 or less. Mercury was not detected in surface water and results for PCB concentrations are approximately one-third the water concentrations measured in Green Bay Zone 3A. Given that surface water concentrations in Zone 4 are the lowest that have been measured in the bay, it suggests that there is no risk to water column invertebrates in other areas of the bay, and adverse alterations to the functioning water column invertebrate communities are not expected.

Benthic Invertebrates. The risk characterization found that concentrations of total PCBs in the sediment are at sufficient concentrations to cause adverse alterations

to benthic invertebrate communities. Benthic invertebrate community investigations have not been conducted in this zone of Green Bay or in the adjacent Zone 3. It is not clear to what degree results of investigations in Green Bay zones 1 and 2 can be related to existing benthic invertebrate conditions in Green Bay Zone 4. HQs for total PCBs are 3.7 (non-interpolated) and 1.4 (I_d interpolated).

Benthic Fish. Data were not available for the estimation of risk to benthic fish.

Pelagic Fish. Concentrations of total PCBs and p,p'-DDE in pelagic fish are sufficiently high to potentially be of risk to pelagic fish reproduction or survival. Given that the USFWS found sublethal effects in Green Bay walleye, while individual pelagic species may not be at risk, pelagic species as a whole are potentially experiencing adverse risk to survival or reproduction from COCs.

Insectivorous Birds. Data were not available for the estimation of risk to insectivorous birds.

Piscivorous Birds. Estimated concentrations of mercury and total PCBs in the diet of piscivorous birds are sufficient to cause potential adverse effects to survival, physiology, or reproduction. The only species-specific difference noted was regarding risks for mercury; common tern and Forster's tern are at potential risk, but double-crested cormorants are not. There are no measured concentrations for this zone of Green Bay. For this zone it is assumed that mercury and total PCBs are at sufficient concentrations to cause survival or reproductive impairment in piscivorous birds.

Carnivorous Birds. Estimated concentrations of mercury, total PCBs, and p,p'-DDE in the diet of carnivorous birds are at sufficient concentrations to cause potential adverse effects on survival, physiology, or reproduction. There are no measured endpoints for this zone of Green Bay. For this zone it is assumed that mercury, total PCBs, and p,p'-DDE are at sufficient concentrations to cause survival or reproductive impairment in carnivorous birds.

Piscivorous Mammals. Piscivorous mammals are estimated to be at risk in this zone to total PCBs.

Habitat suitable to support mink was identified in this zone, but there are no capture or population level data for either mink or other piscivorous mammals such as river otter.

Based upon the estimated dietary intakes, site contaminants are estimated to be sufficient to cause survival or reproductive impairment to piscivorous mammals.

6.8 Section 6 Figures and Tables

Section 6 figures and tables follow page 6-238 and include:

Figure 6-1 Food Web Model, Lower Fox River - Little Lake Butte des Morts to the De Pere Dam

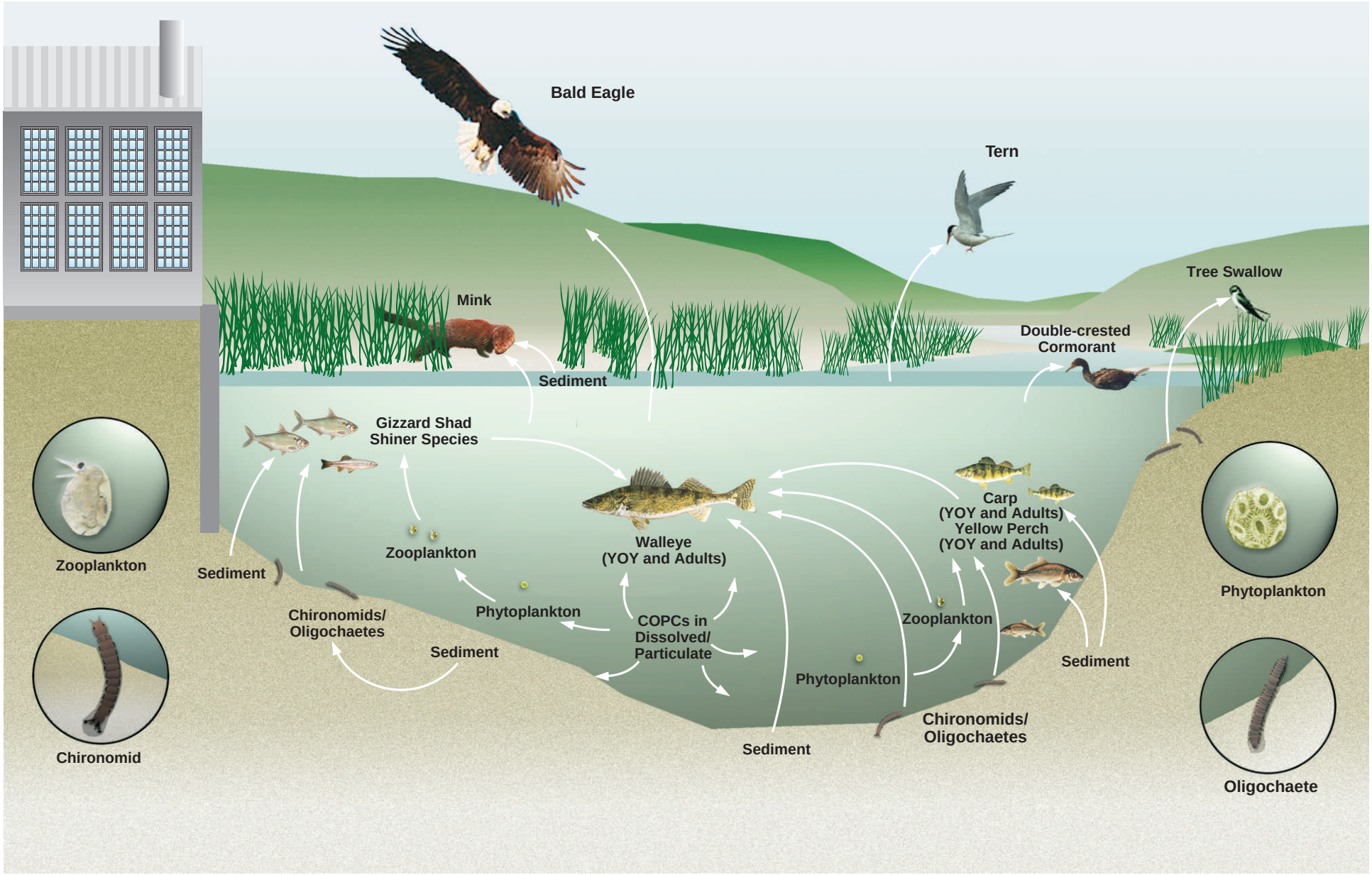

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Figure 6-2 Food Web Model, Green Bay - Zones 1 and 2

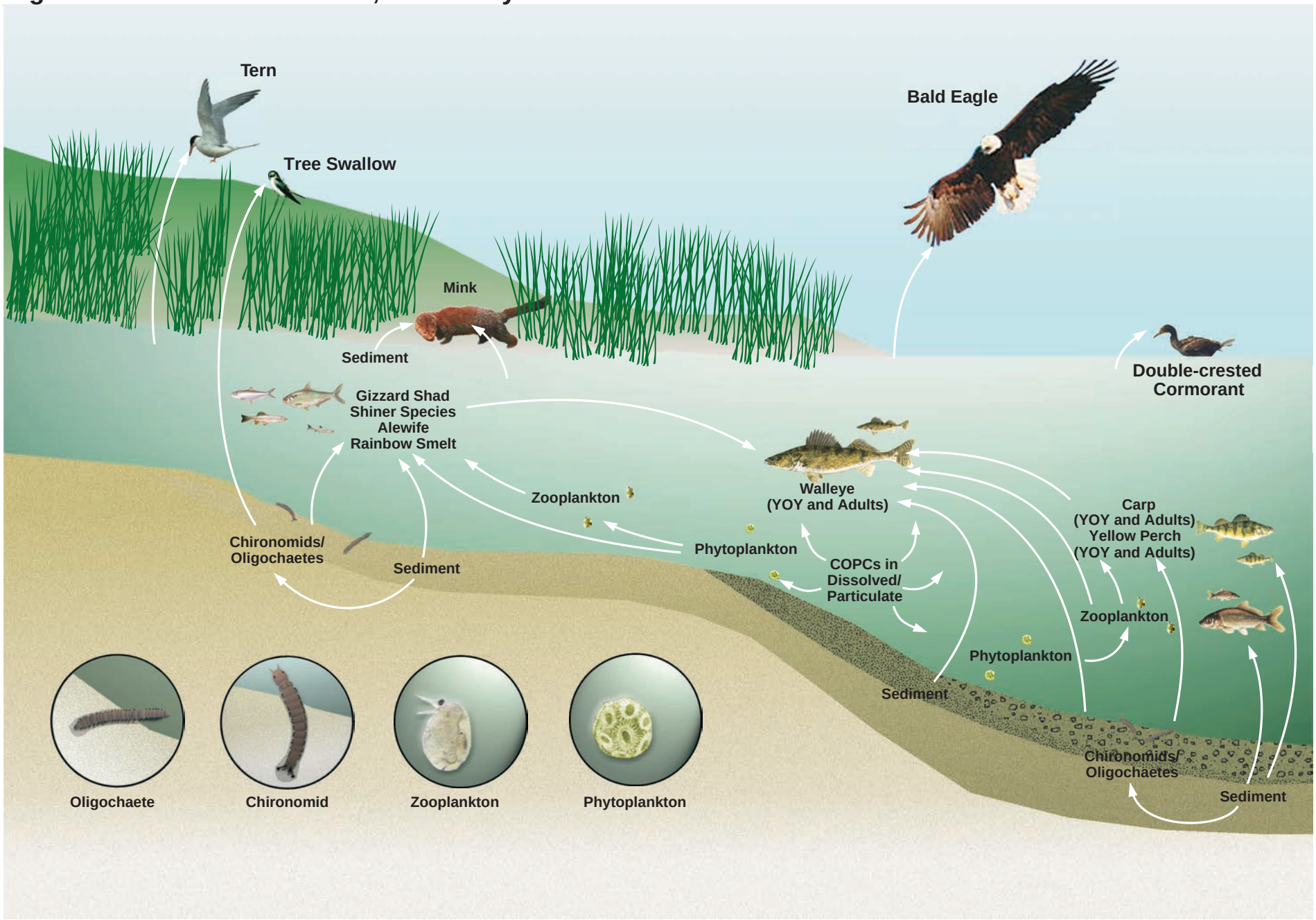

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Figure 6-3 Food Web Model, Green Bay - Zones 3 and 4

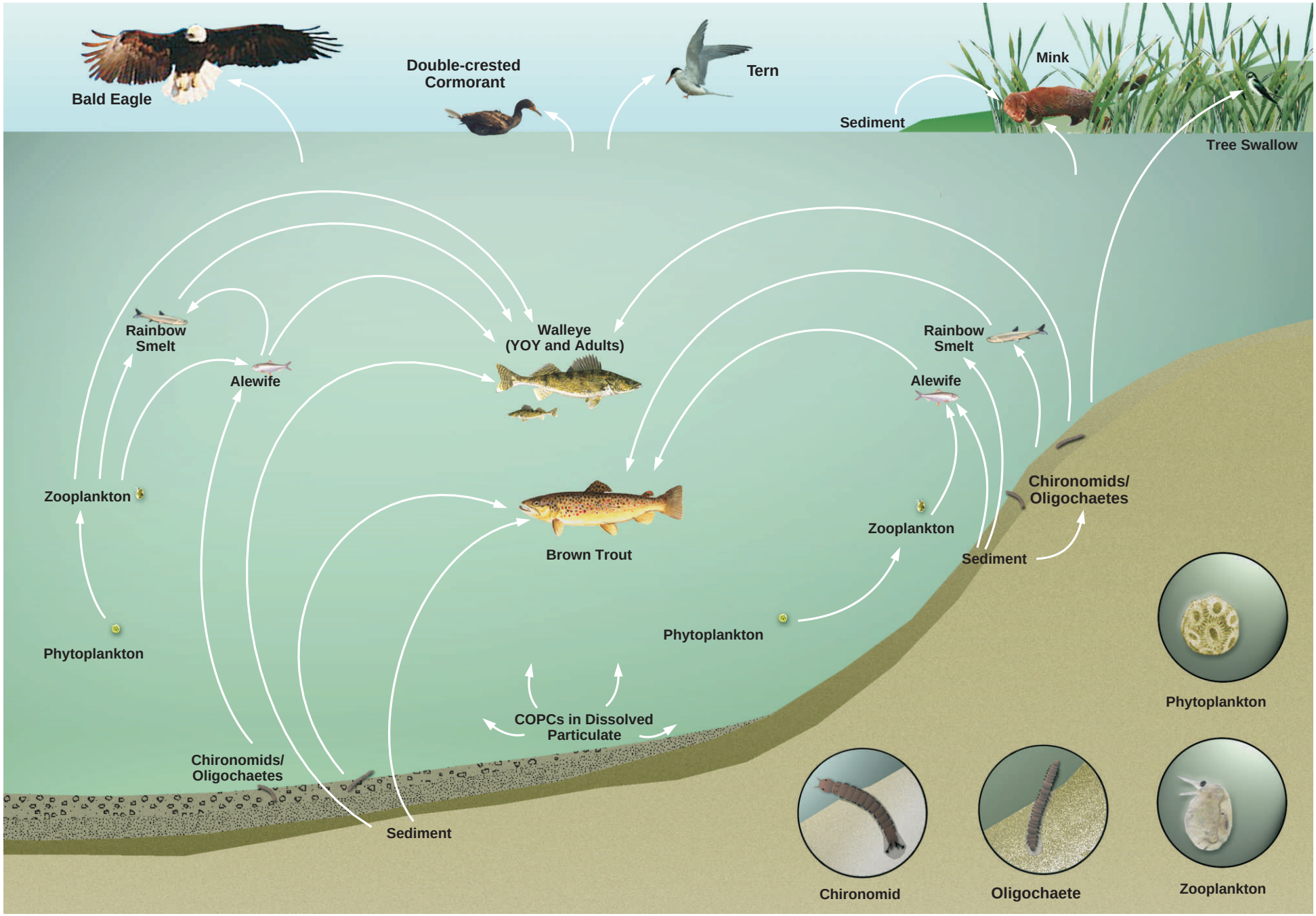

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Figure 6-4 Data for Egg Mortality and TCDD-Eq

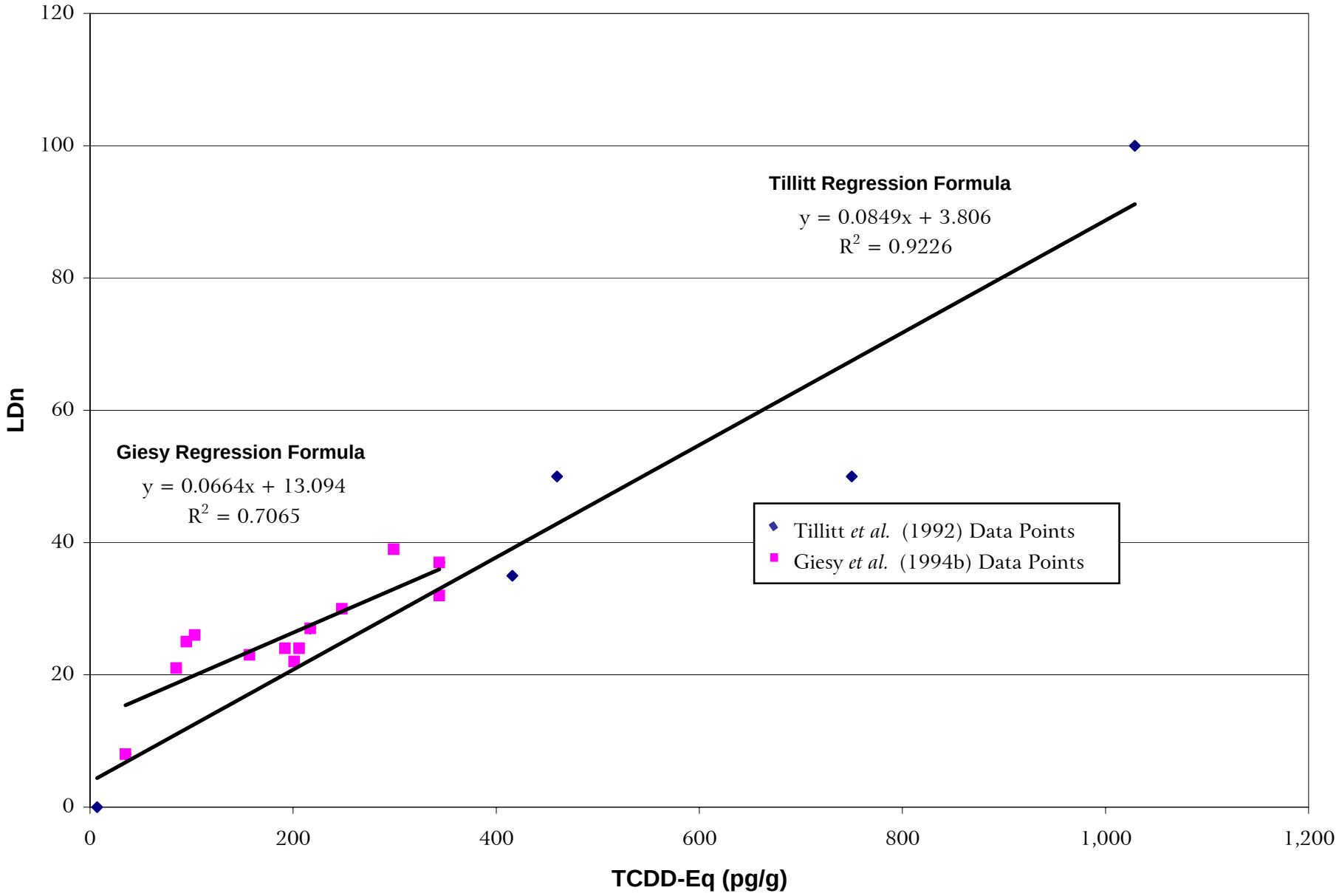


Figure 6-5 Unfiltered Mercury Concentrations in Surface Water

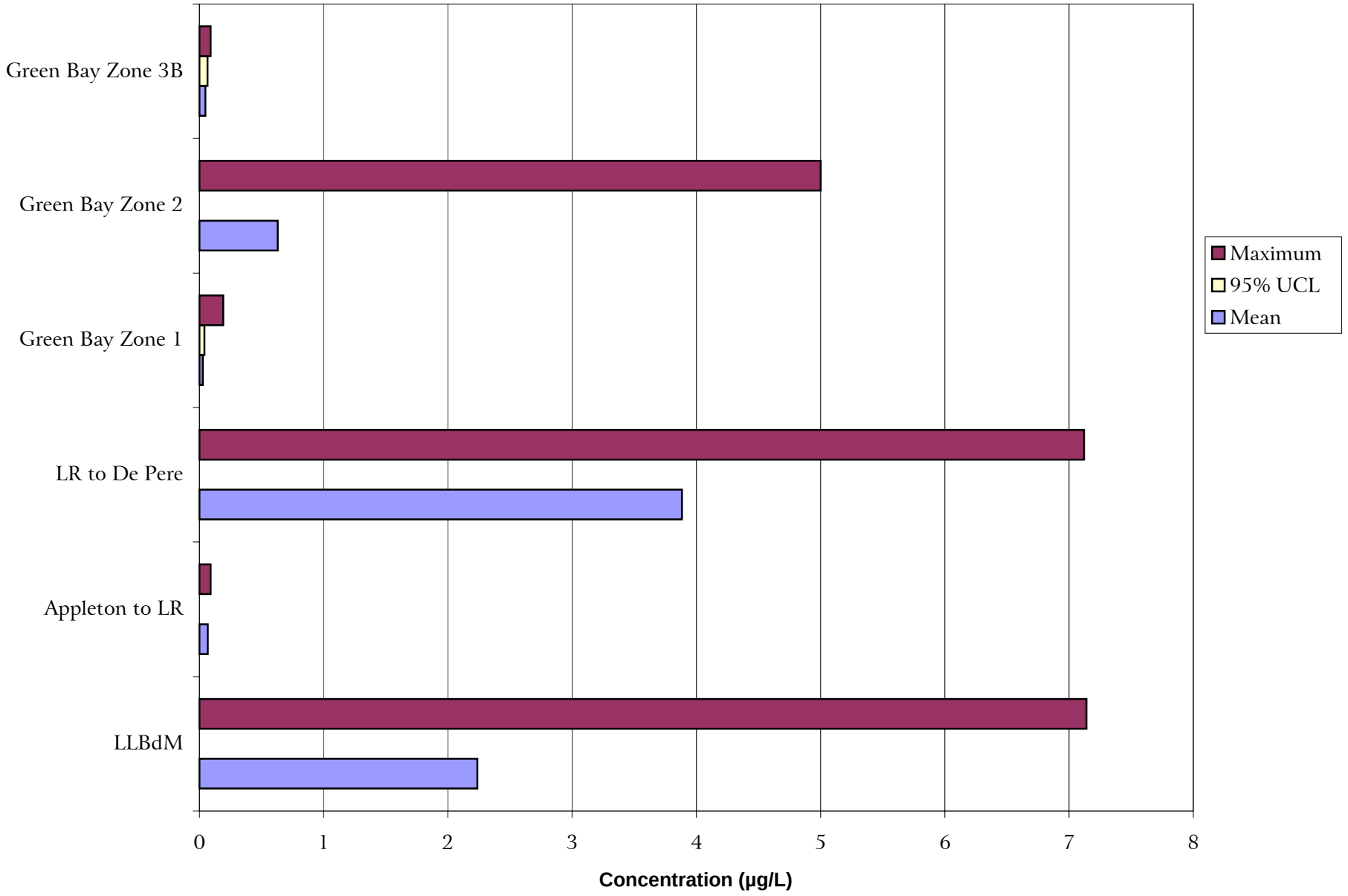


Figure 6-6 Total PCB Concentrations (Filtered + Particulate) in Surface Water

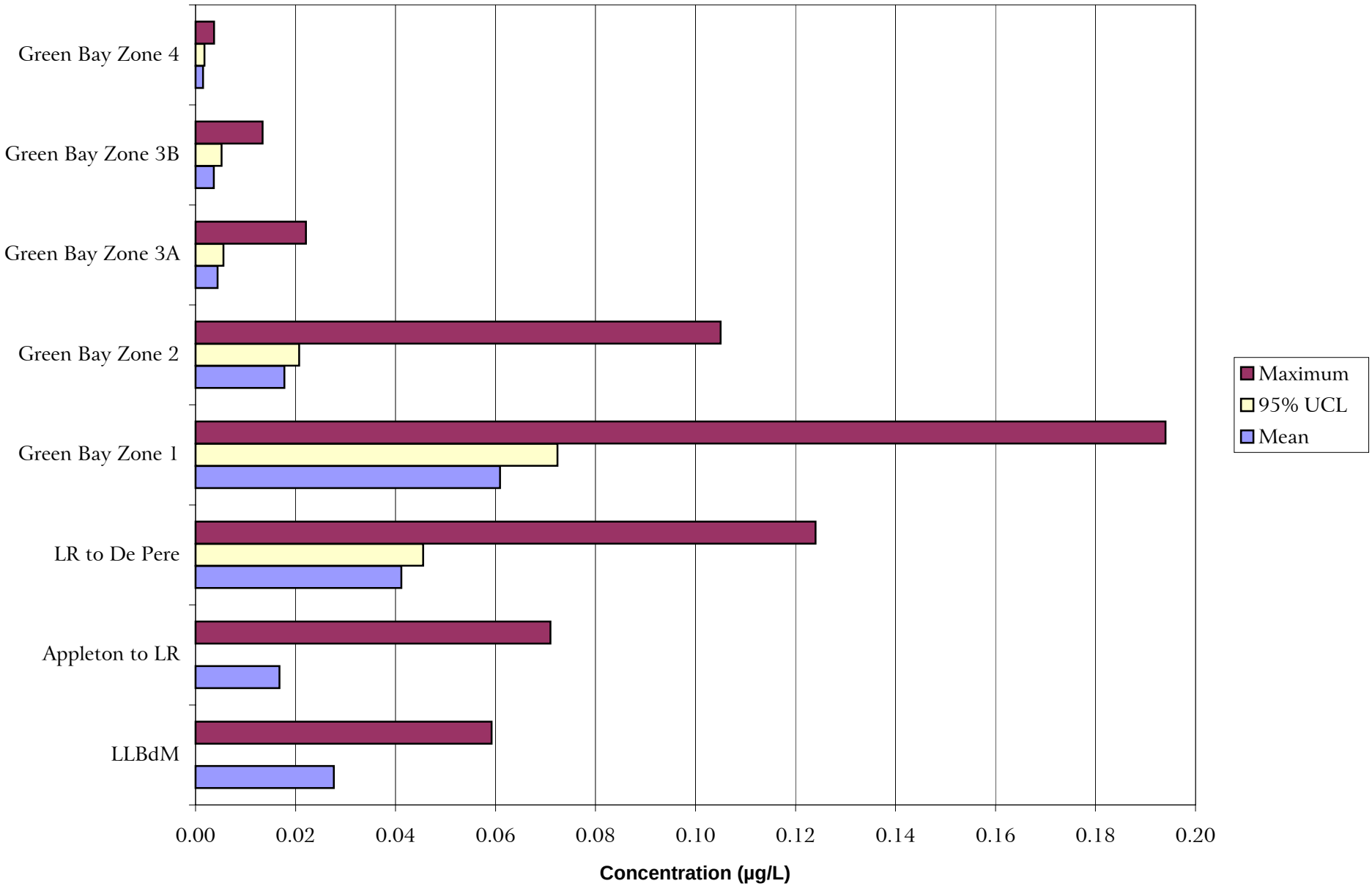


Figure 6-7 Metal Concentrations in Sediments

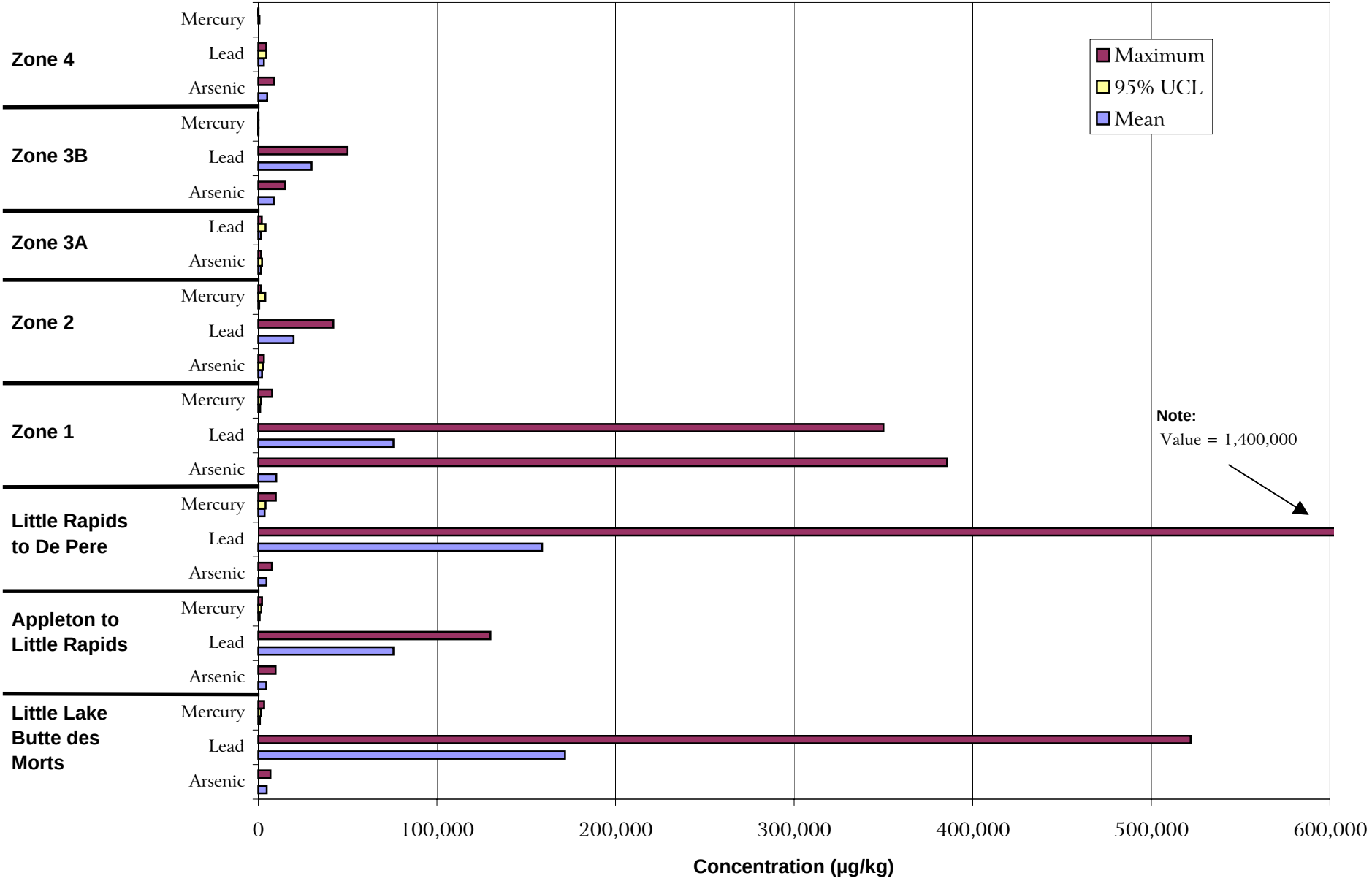


Figure 6-8 Total PCB Concentrations in Sediment

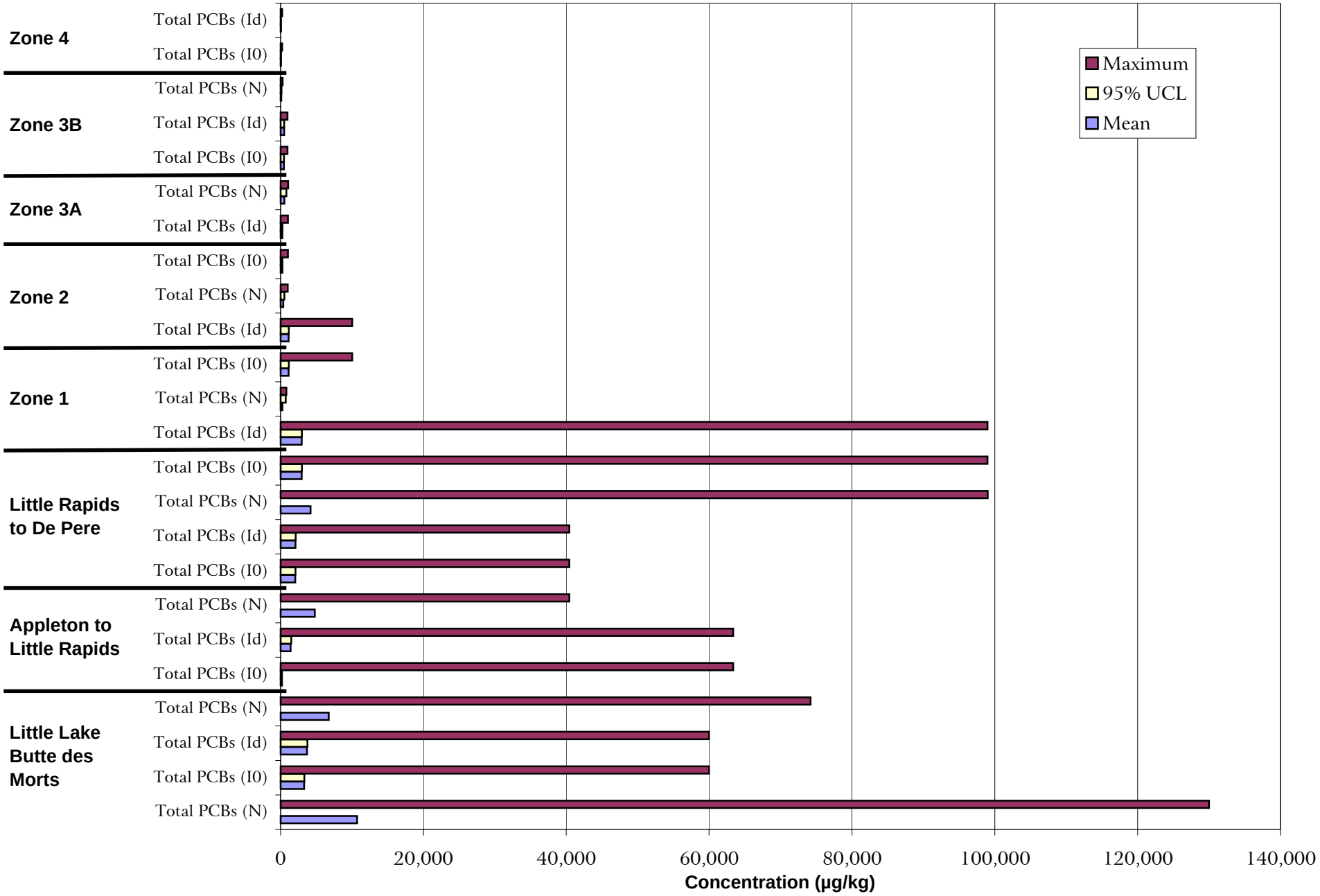


Figure 6-9 Chlorinated Pesticide Concentrations in Sediment

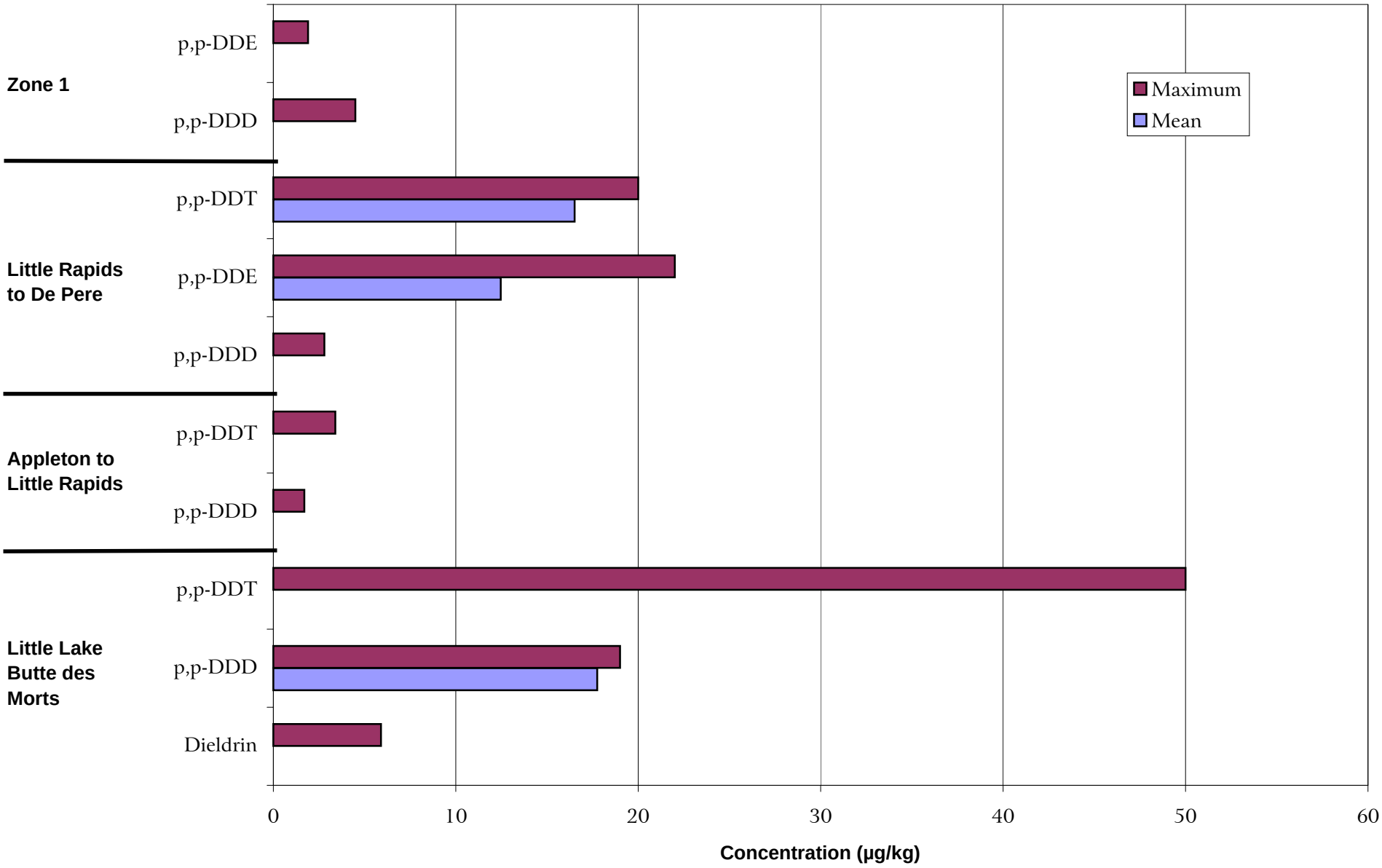


Figure 6-10 Mercury Concentrations in Fish

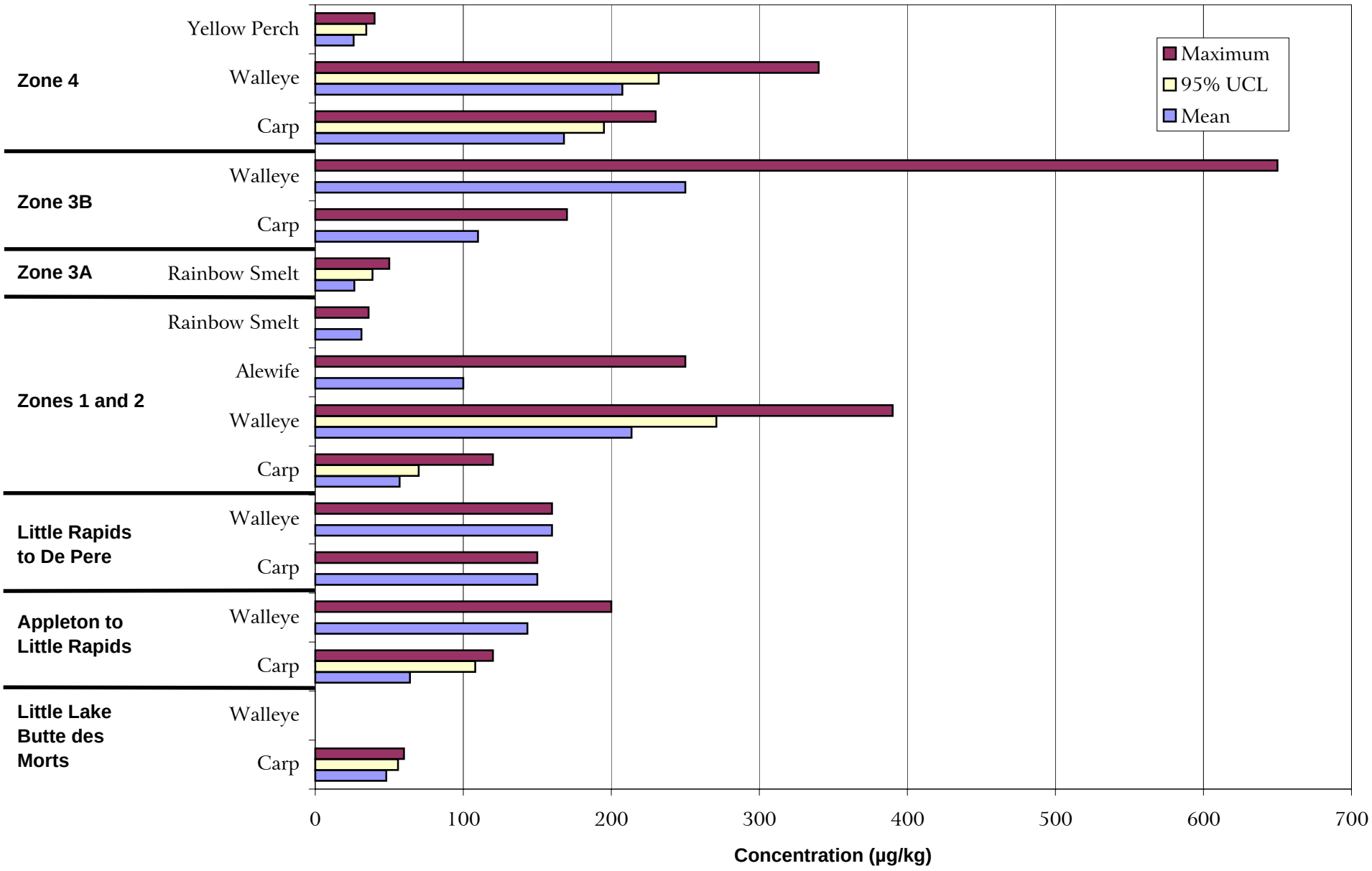


Figure 6-11 Total PCBs Concentrations in Yellow Perch, Carp, and Walleye

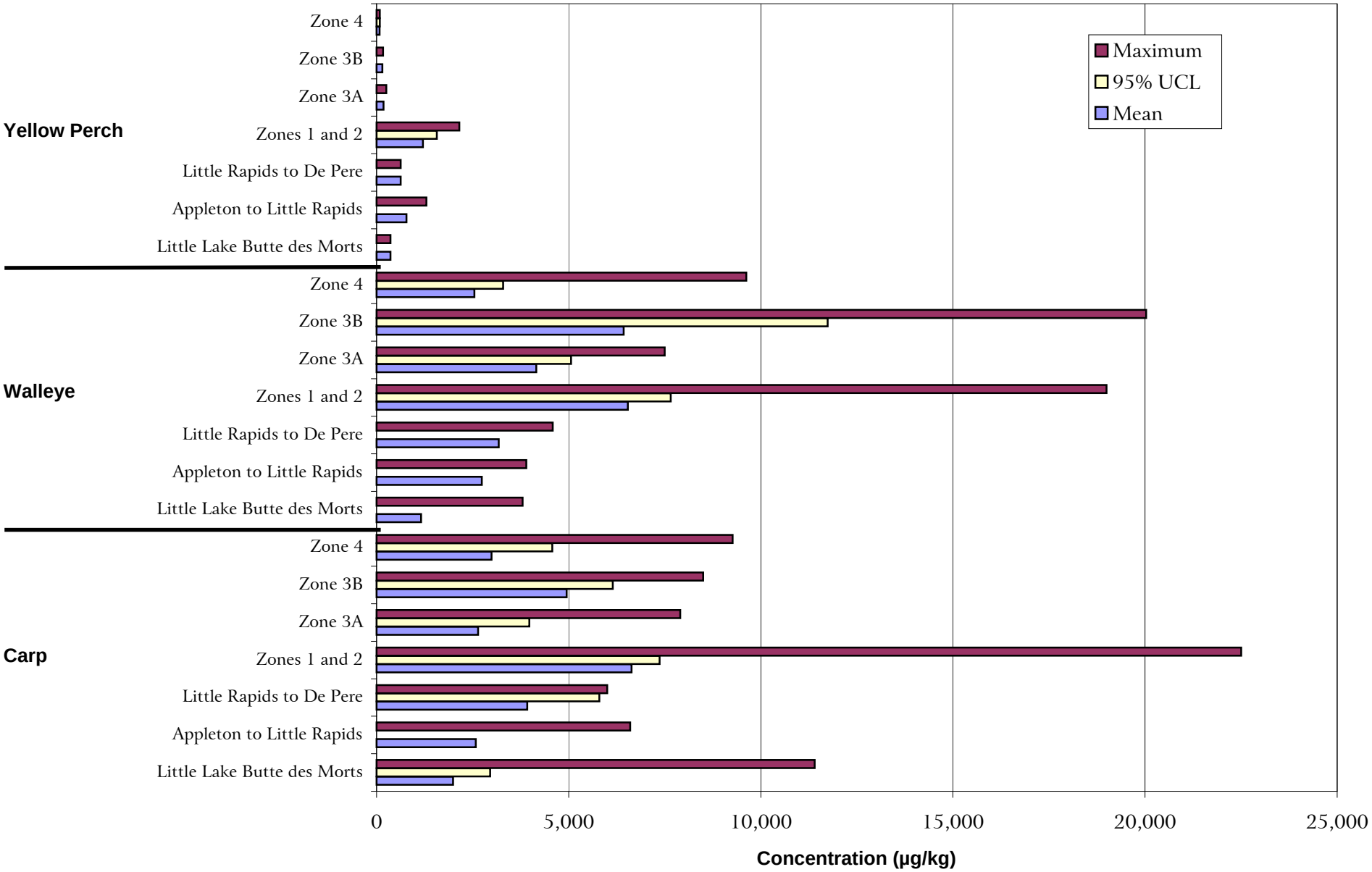


Figure 6-12 Total PCB Concentrations in Forage Fish

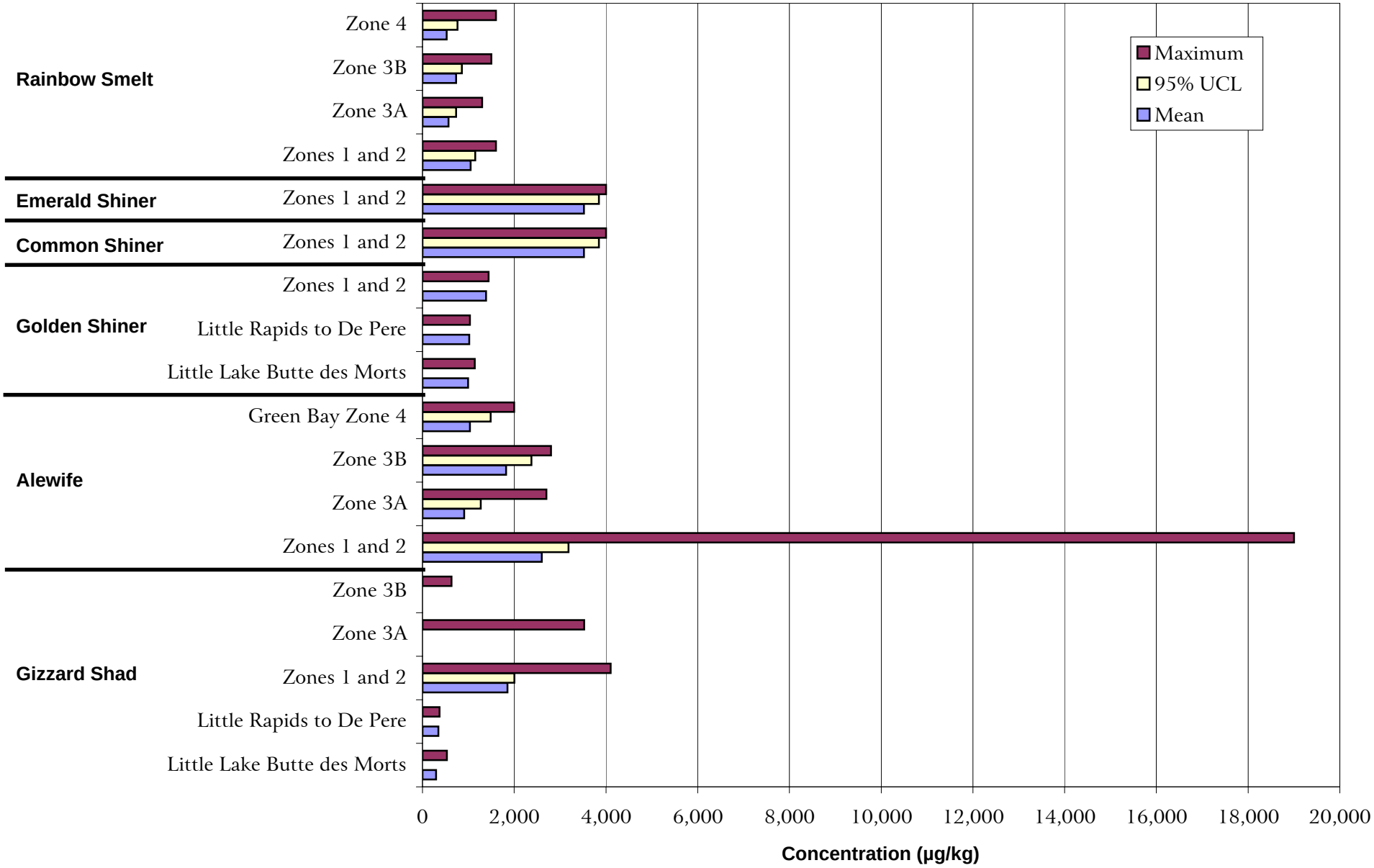


Figure 6-13 Dieldrin Concentrations in Fish

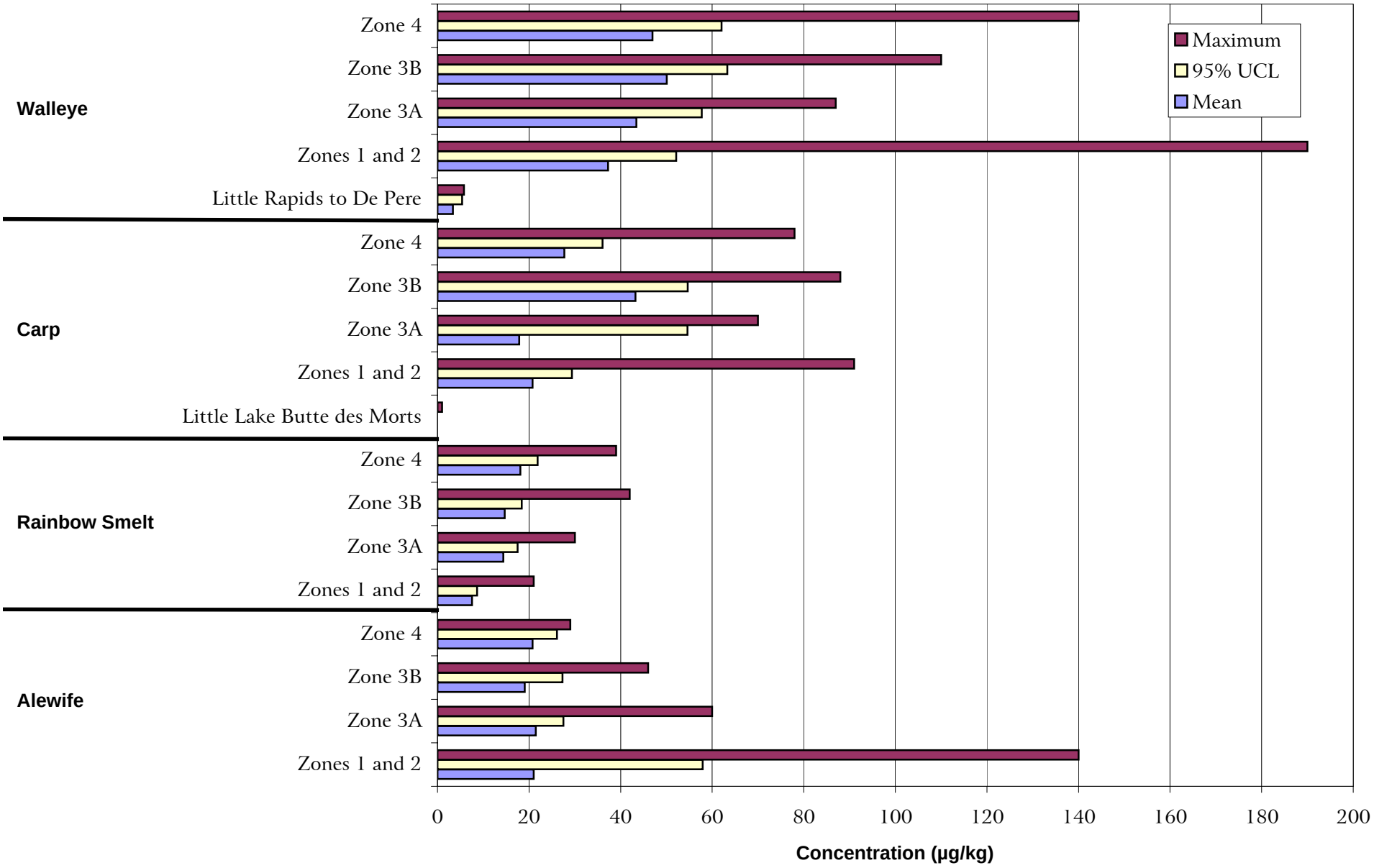


Figure 6-14 p,p'-DDE Concentrations in Fish

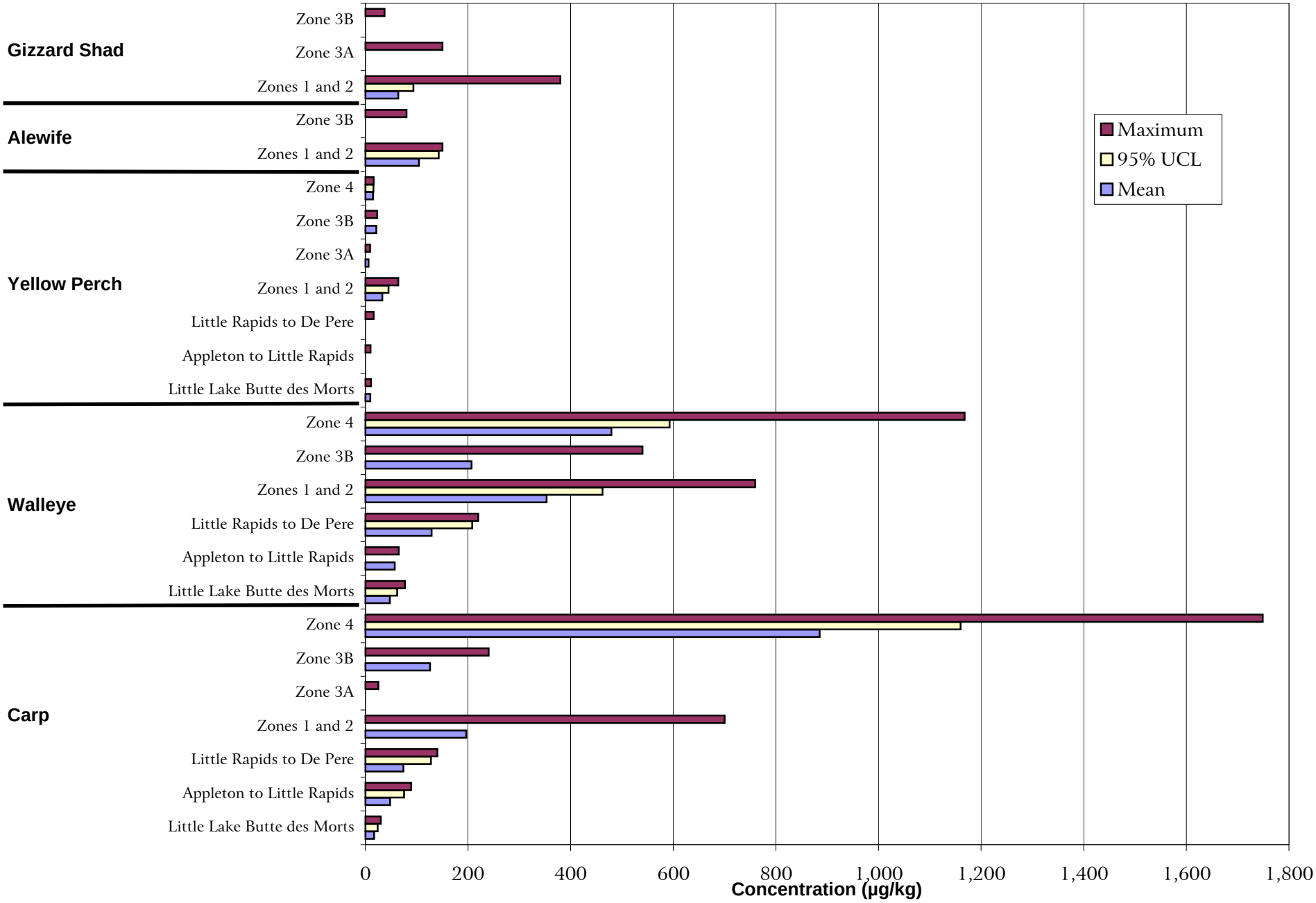


Figure 6-15 Measured Total PCB Concentrations in Birds

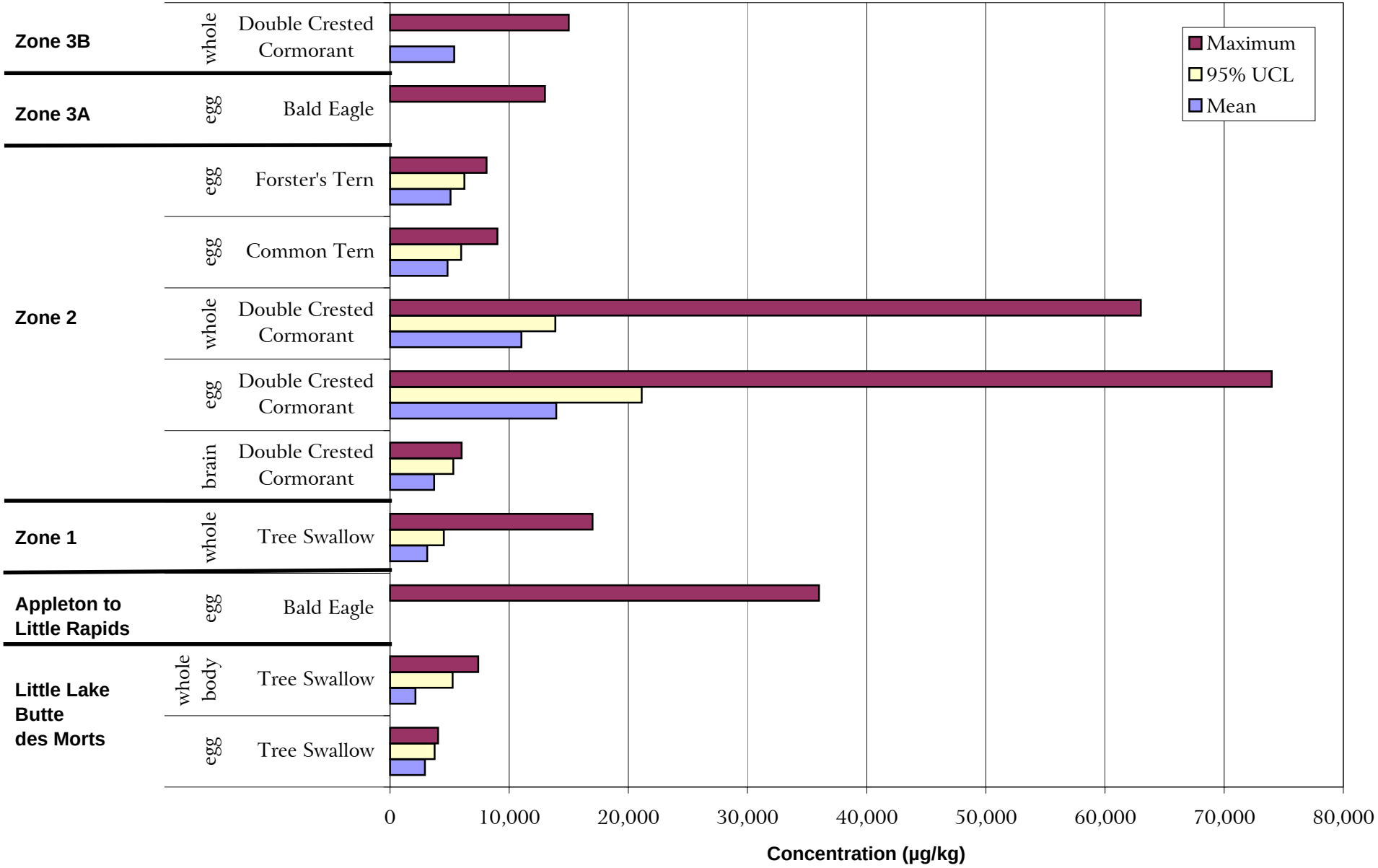


Figure 6-16 Measured Dieldrin Concentrations in Birds

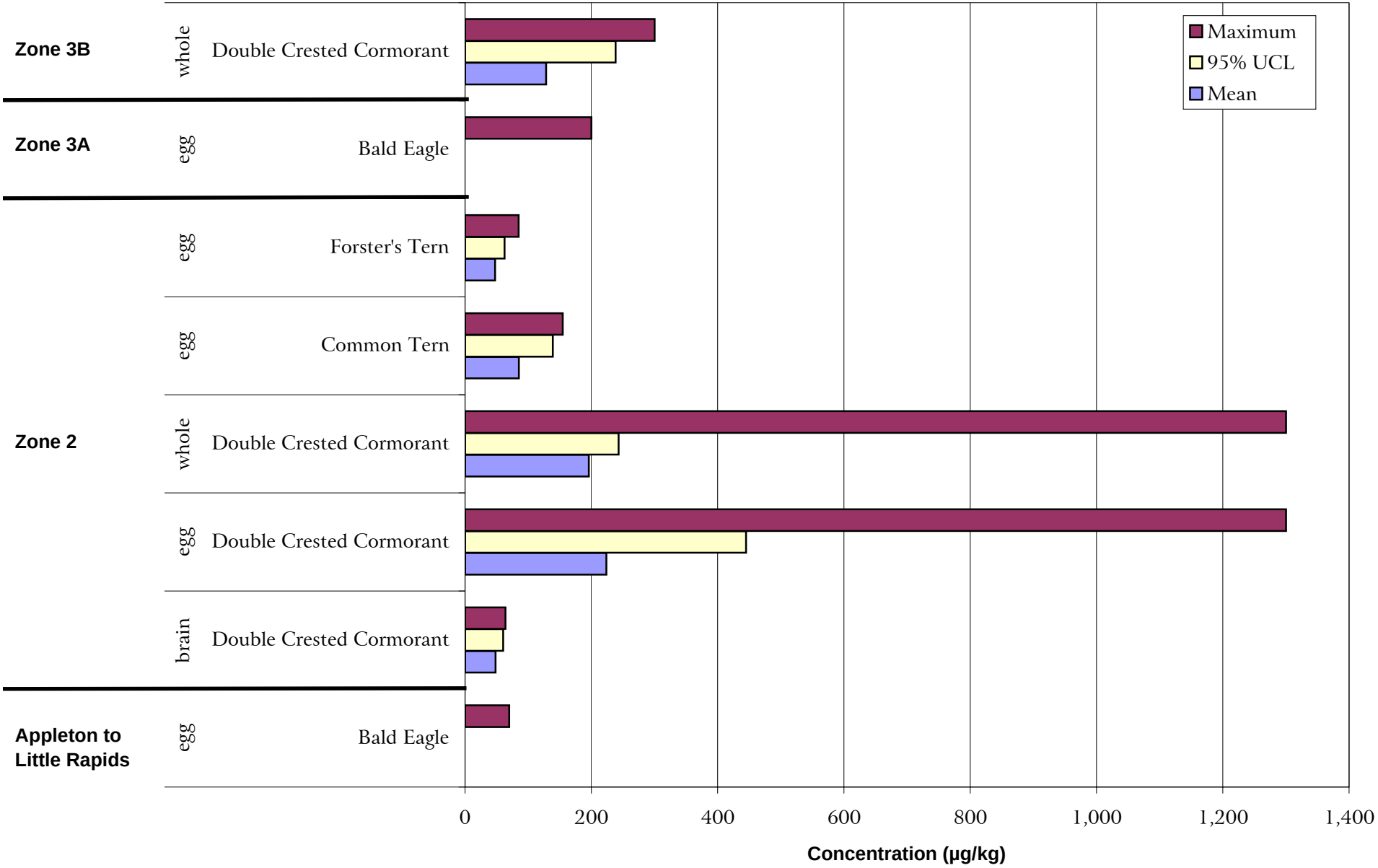


Figure 6-17 Measured p,p'-DDE Concentrations in Birds

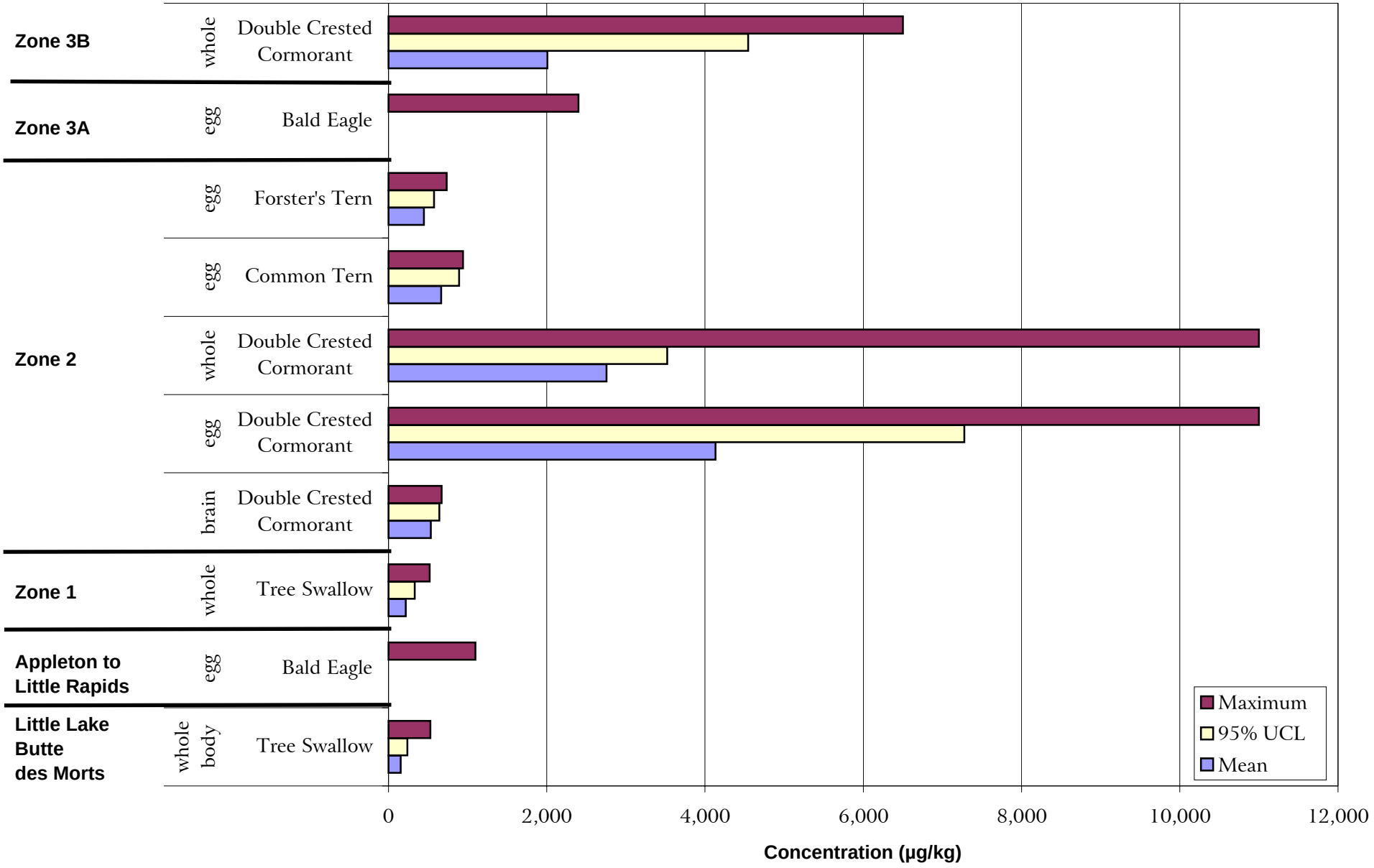
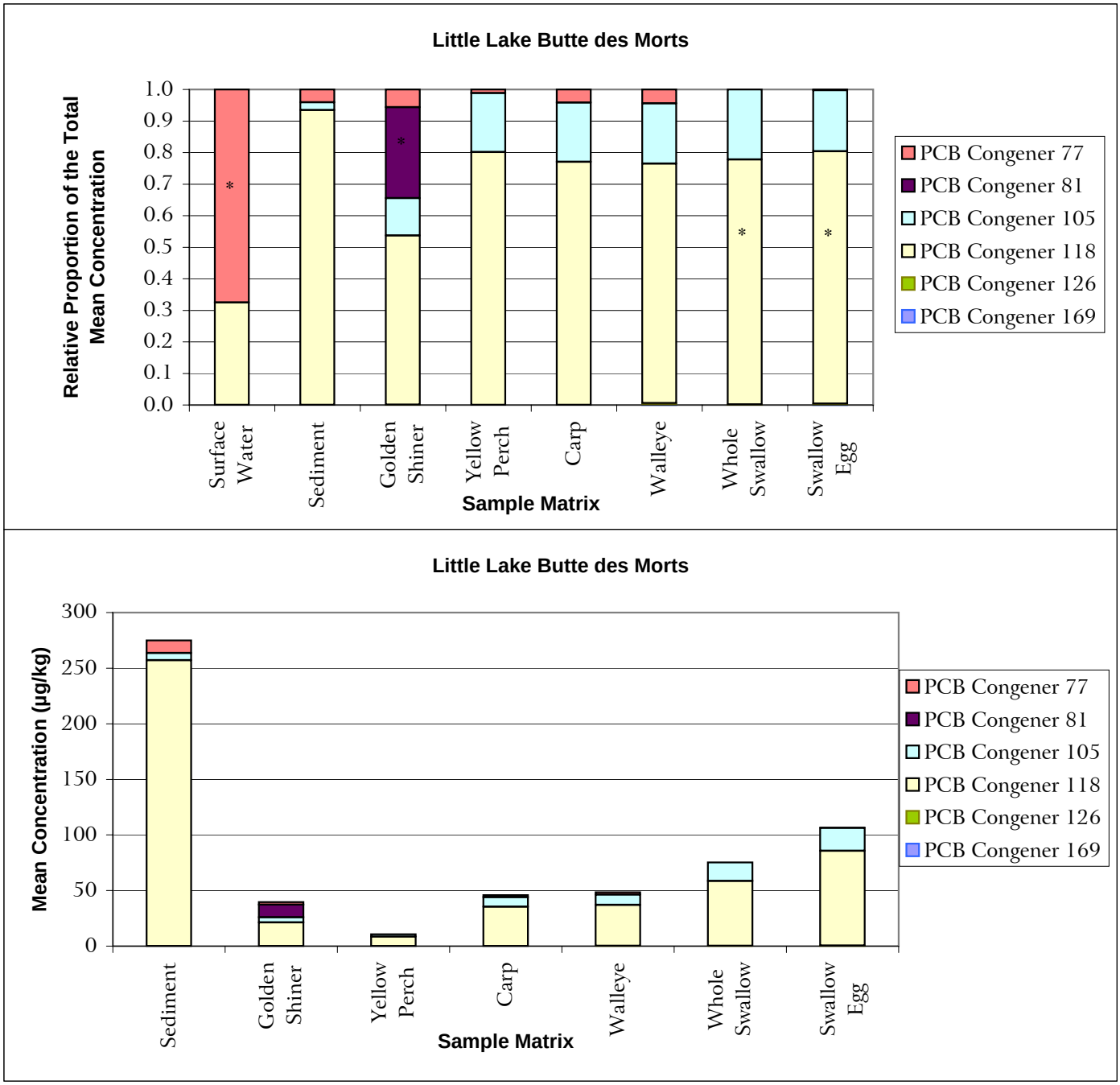
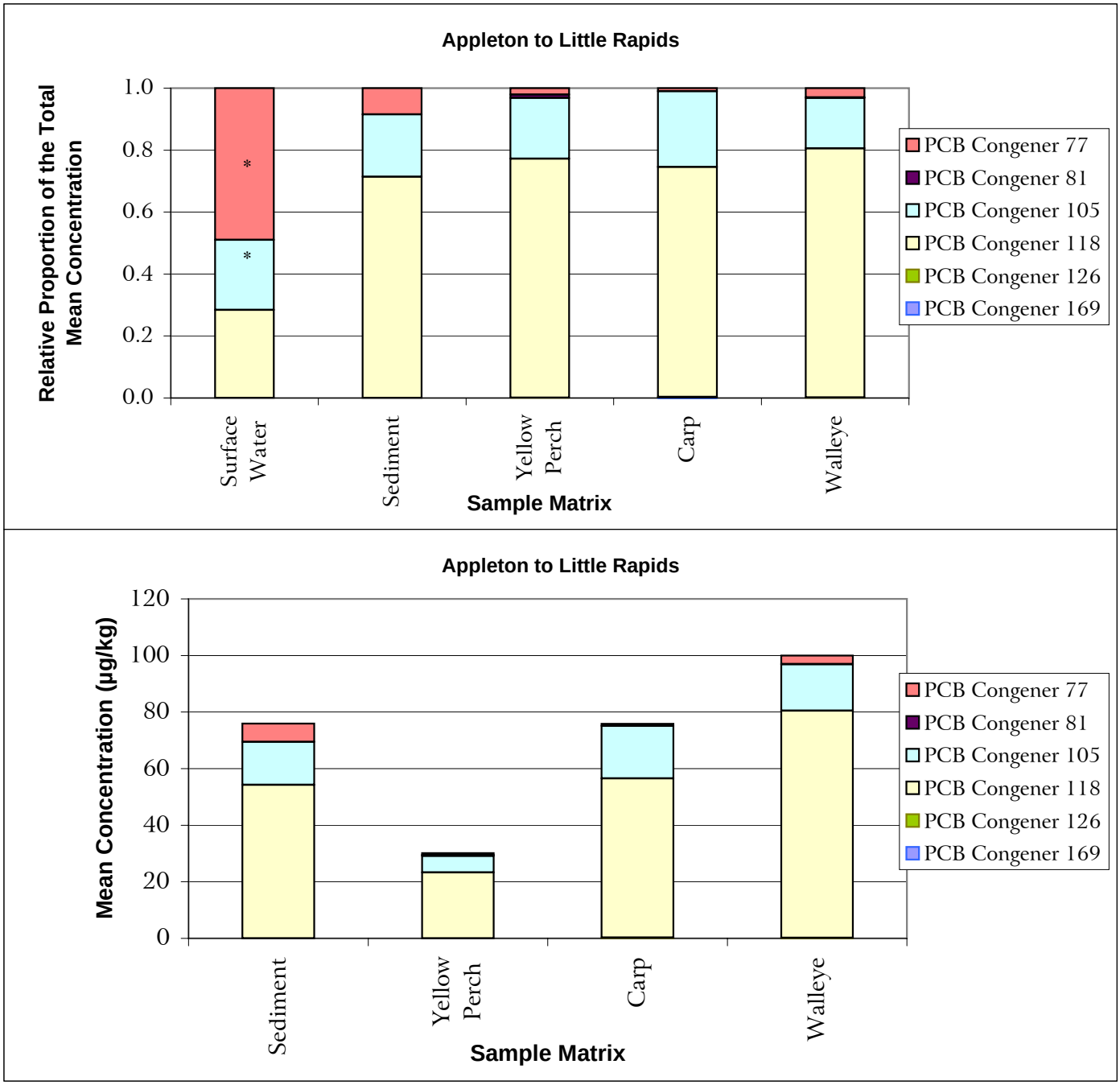


Figure 6-18 PCB Congener Concentrations in Little Lake Butte des Morts Reach



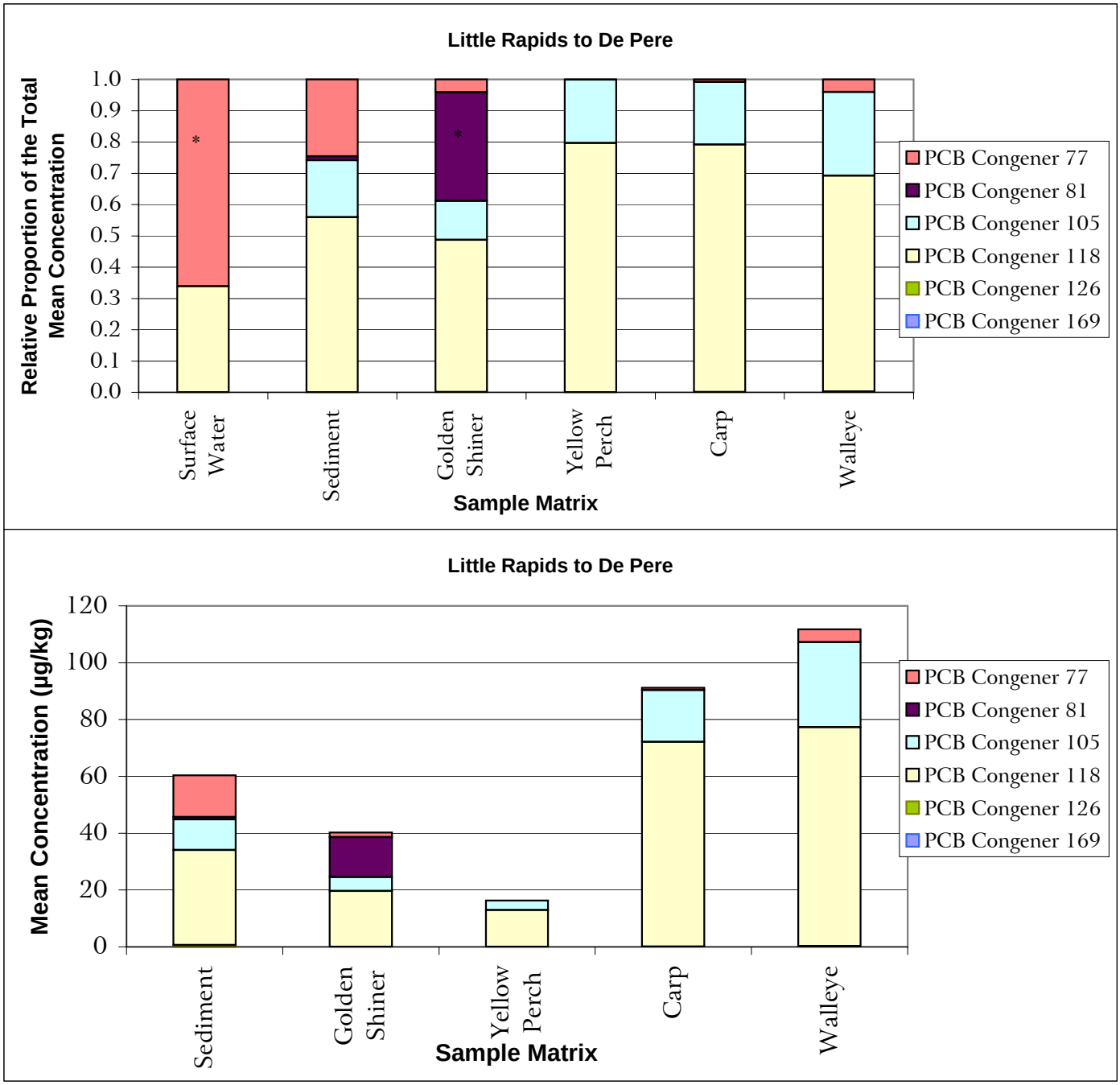
Notes: * Coelution occurred during congener analysis. Whole values used when coelution of congeners occured.
Tissue samples calculated as wet weight.
Sediment sample dry weight corrected.
Congeners 81, 105, 126, and 169 were not anlayzed in surface water.
Congener 81 was not analyzed in whole tree swallow or tree swallow egg.
The mean for congners 81 and 126 could not be evaluated for sediment.
The mean for congner 126 could not be evaluated for carp.

Figure 6-19 PCB Congener Concentrations in Appleton to Little Rapids Reach



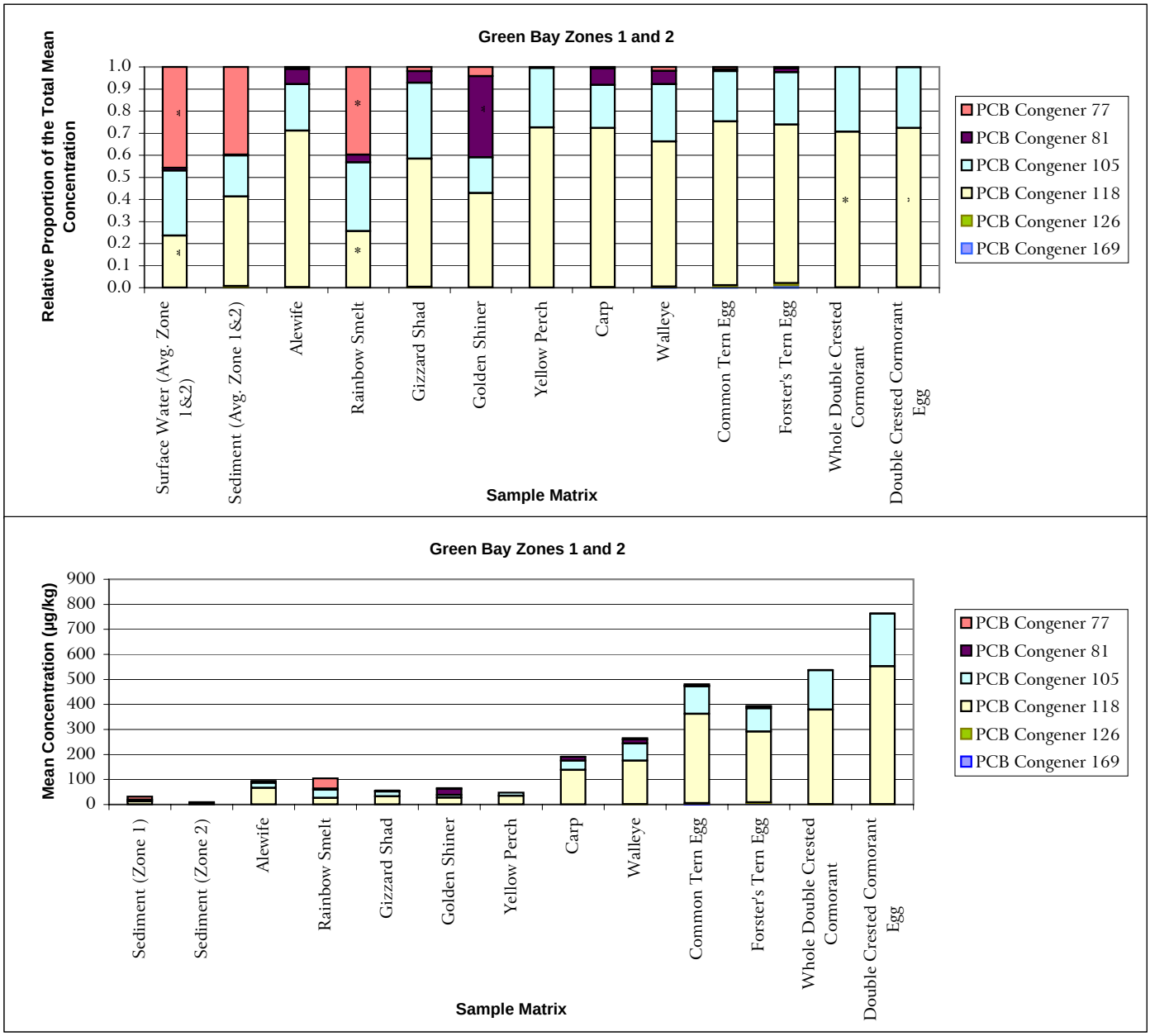
Notes: * Coelution occurred during congener analysis. Whole values used when coelution of congeners occured.
Tissue samples calculated as wet weight.
Sediment sample dry weight corrected.
Congeners 81, 126, and 169 were not anlayzed in surface water.

Figure 6-20 PCB Congener Concentrations in Little Rapids to De Pere Reach



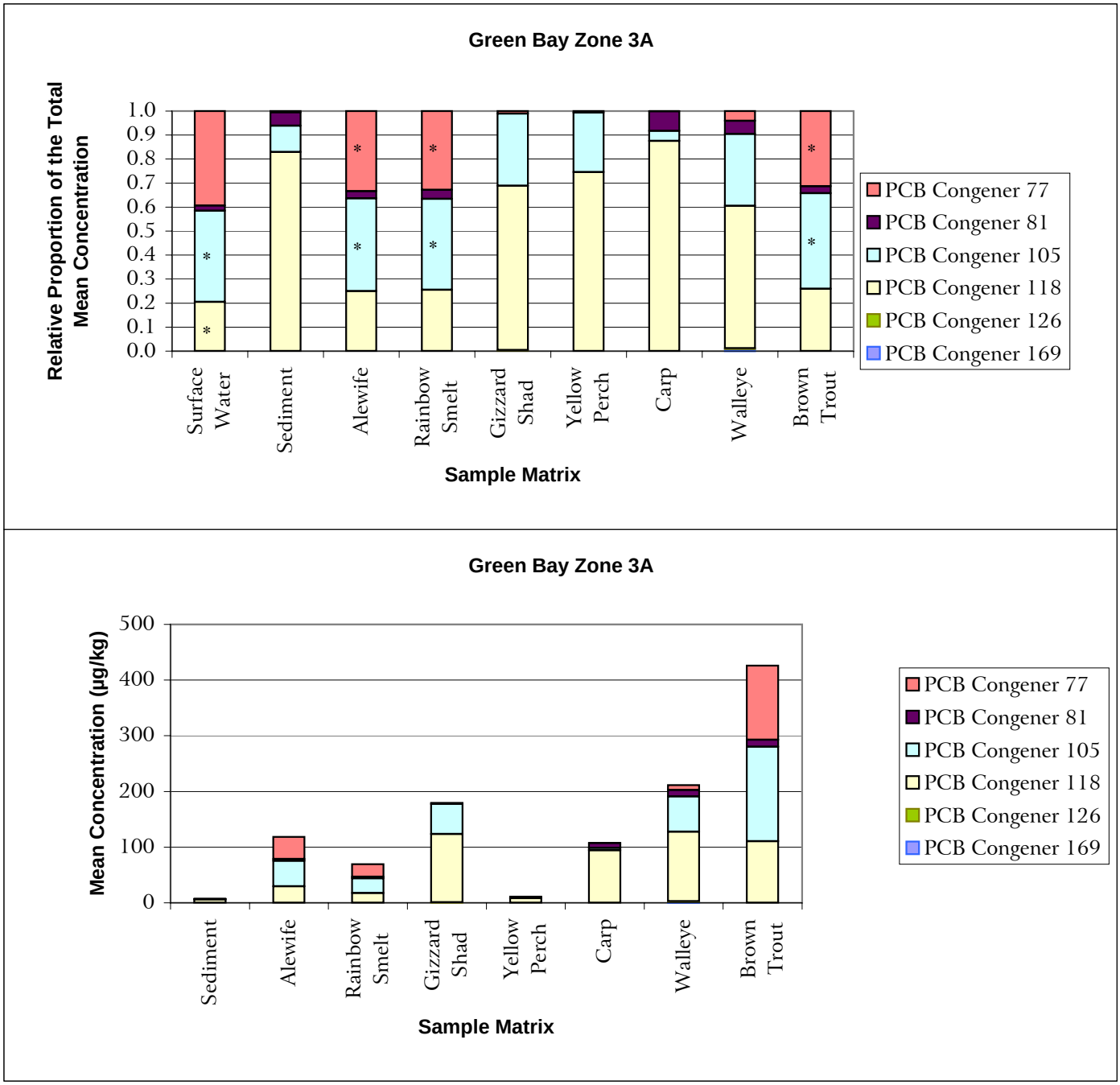
Notes: * Coelution occurred during congener analysis. Whole values used when coelution of congeners occured.
Tissue samples calculated as wet weight.
Sediment sample dry weight corrected.
Congeners 81, 105, 126, and 169 were not anlayzed in surface water.

Figure 6-21 PCB Congener Concentrations in Green Bay Zones 1 and 2



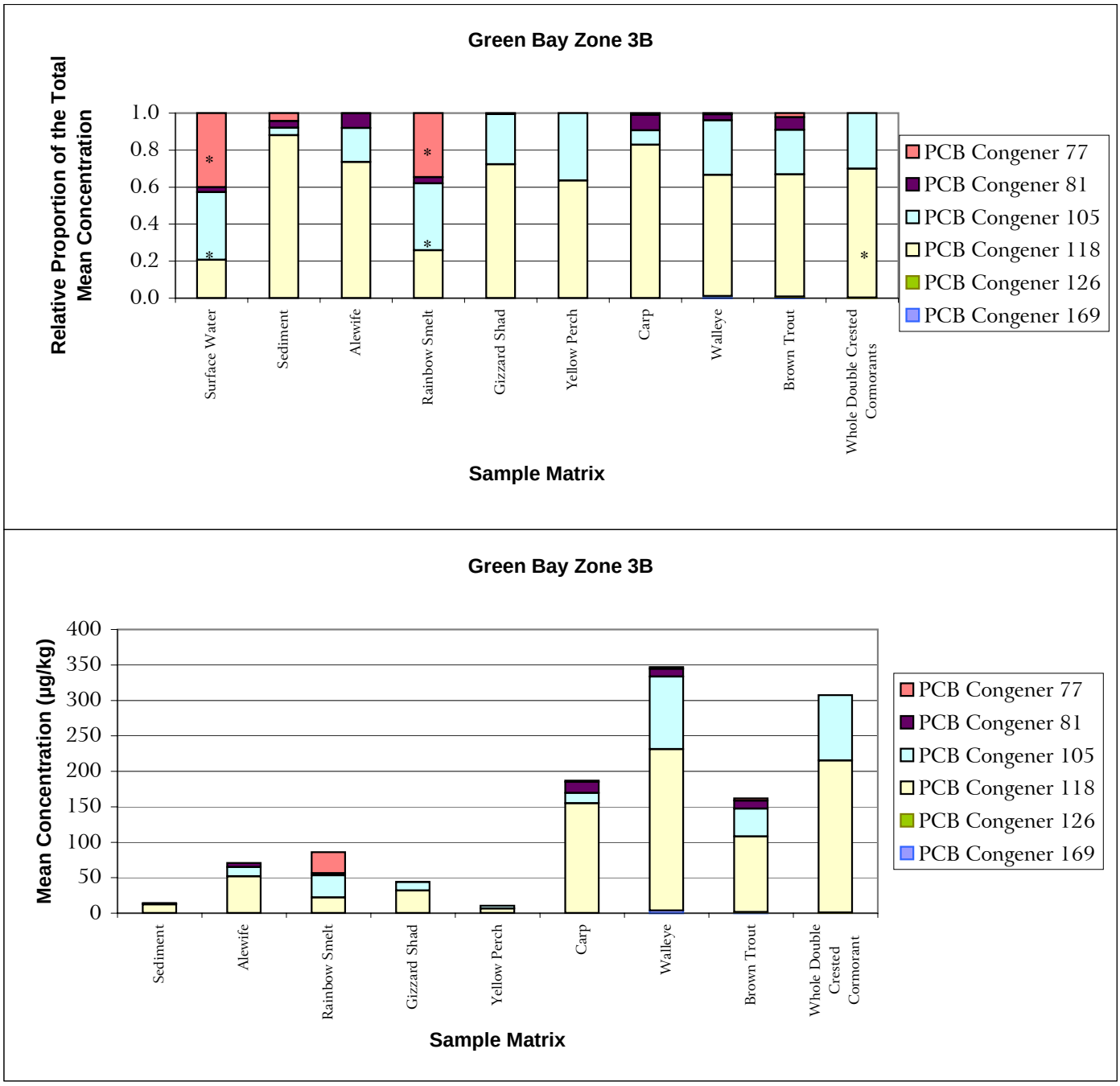
Notes: * Coelution occurred during congener analysis. Whole values used when coelution of congeners occured.
Tissue samples calculated as wet weight.
Sediment sample dry weight corrected.
Congeners 126 and 169 were not anlayzed in surface water or rainbow smelt.
Congener 81 could not be evaluated in sediment (zone 1 only).
Congener 81 was not analyzed in whole double-crested cormorant or double-crested cormorant egg.

Figure 6-22 PCB Congener Concentrations in Green Bay Zone 3A



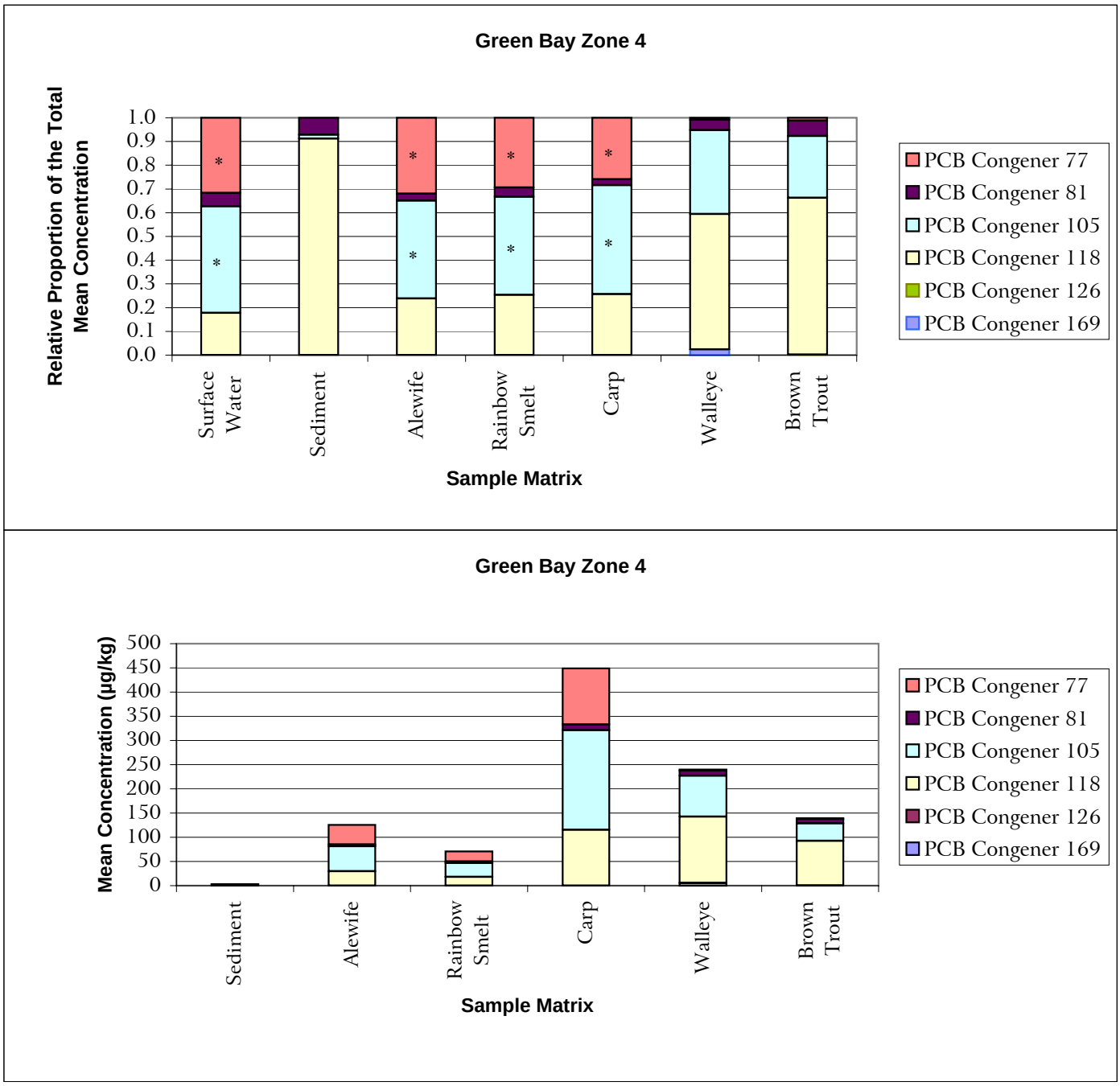
Notes: * Coelution occurred during congener analysis. Whole values used when coelution of congeners occured.
Tissue samples calculated as wet weight.
Sediment sample dry weight corrected.
Congeners 126 and 169 were not anlayzed in surface water, alewife, rainbow smelt, or brown trout.

Figure 6-23 PCB Congener Concentrations in Green Bay Zone 3B



Notes: * Coelution occurred during congener analysis. Whole values used when coelution of congeners occured.
Tissue samples calculated as wet weight.
Sediment sample dry weight corrected.
Congeners 126 and 169 were not anlayzed in surface water or rainbow smelt.
Congener 81 was not anlayzed in whole double-crested cormorant.

Figure 6-24 PCB Congener Concentrations in Green Bay Zone 4



Notes: * Coelution occurred during congener analysis. Whole values used when coelution of congeners occured.
Tissue samples calculated as wet weight.
Sediment sample dry weight corrected.
Congeners 126 and 169 were not anlayzed in surface water, alefiwe, rainbow smelt, or carp.
Congener 169 was not anlayzed in brown trout.
Congener 77 could not be evaluated in sediment.

Figure 6-25 Metal Concentrations in Little Lake Butte des Morts Reach

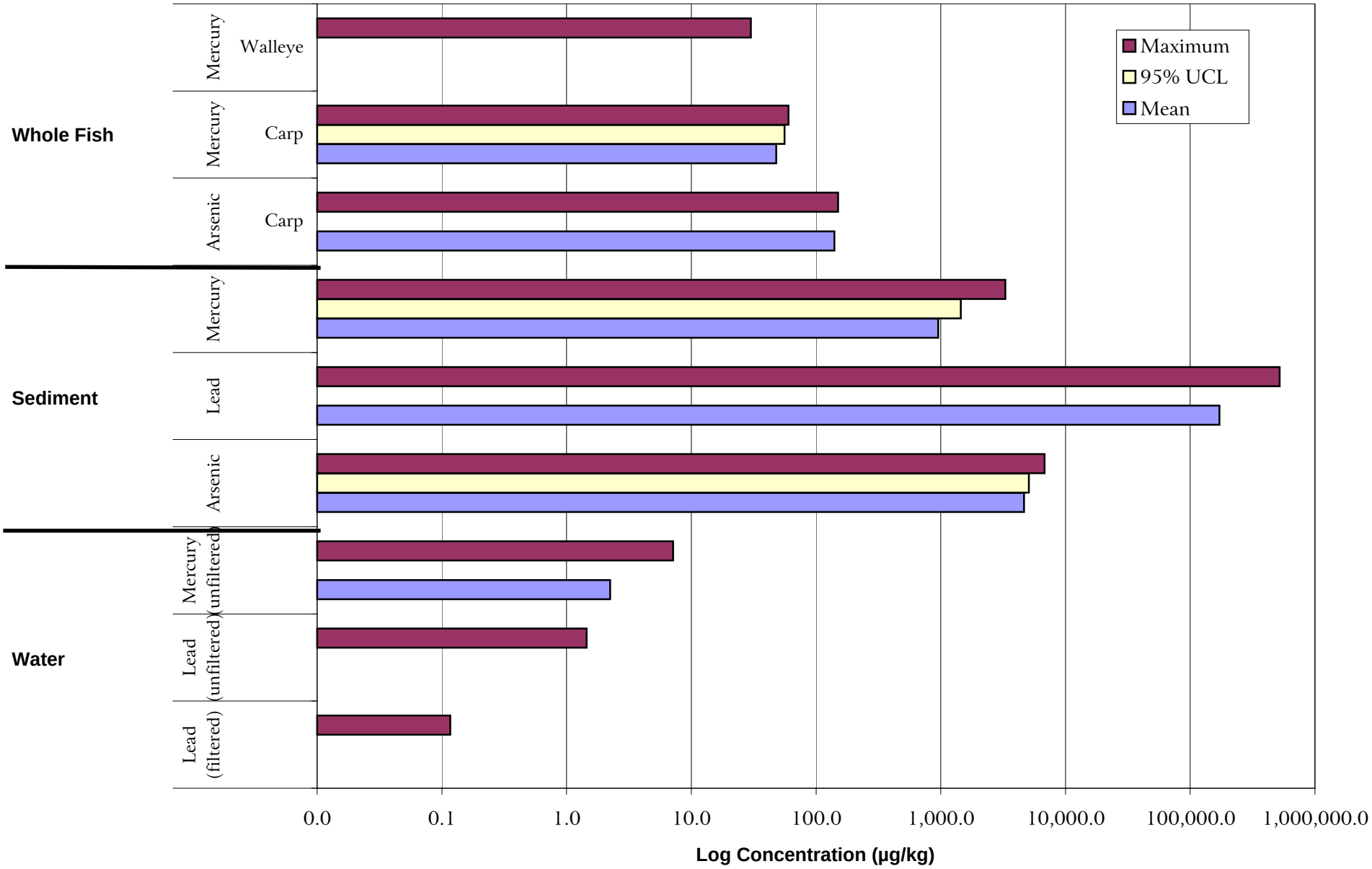


Figure 6-26 Total PCB Concentrations in Little Lake Butte des Morts Reach

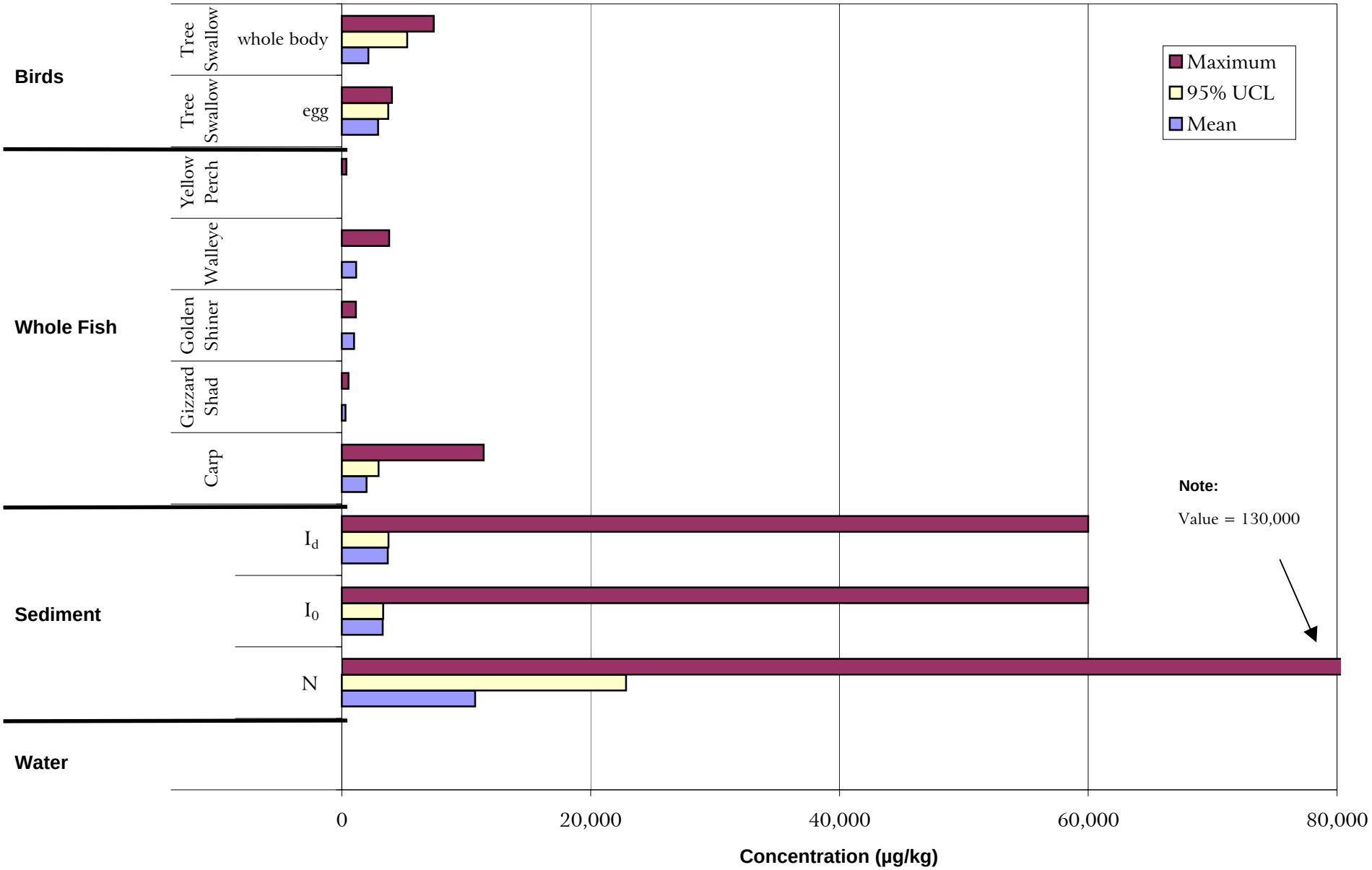


Figure 6-27 Pesticide Concentrations in Little Lake Butte des Morts Reach

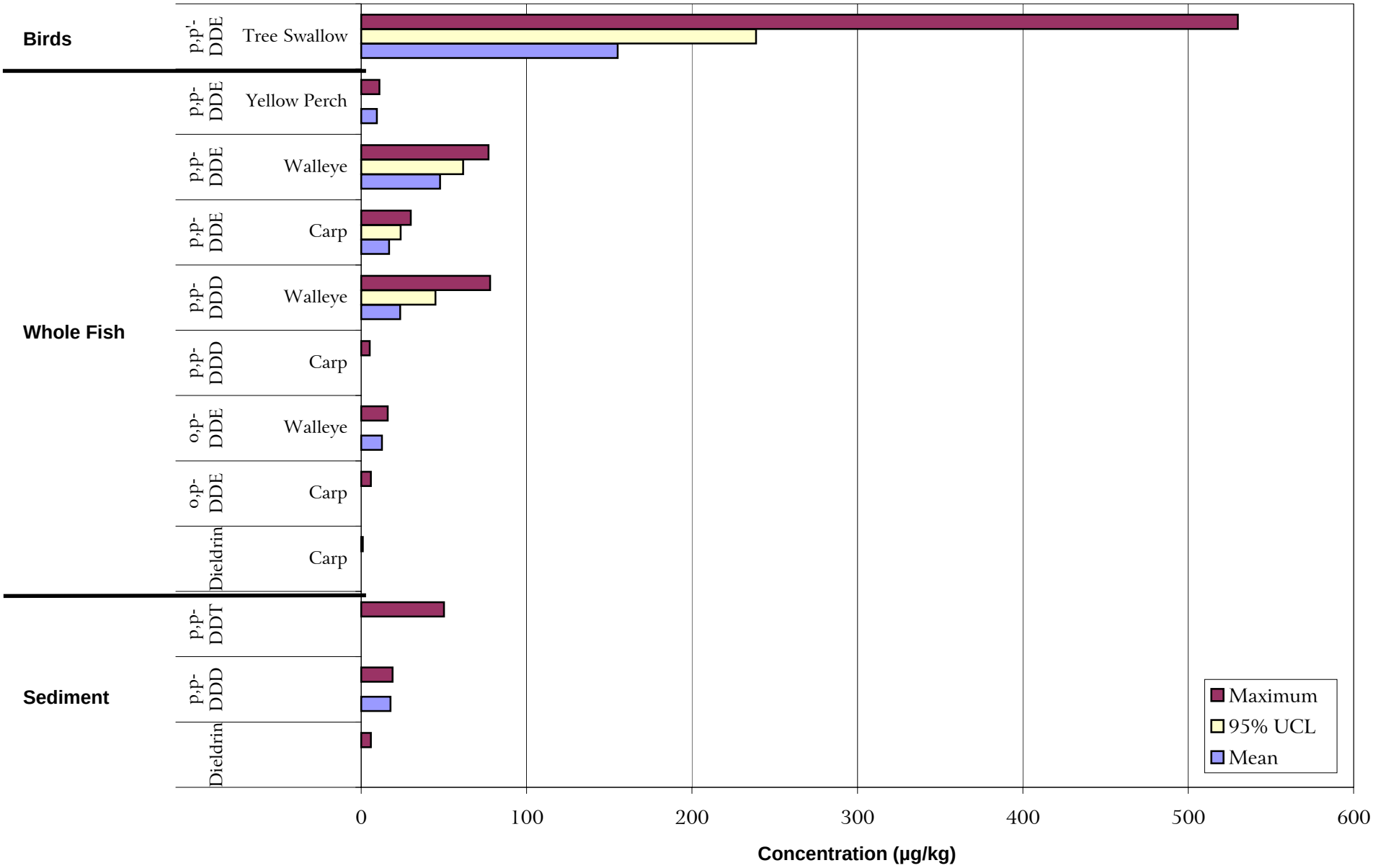


Figure 6-28 Metal Concentrations in Appleton to Little Rapids Reach

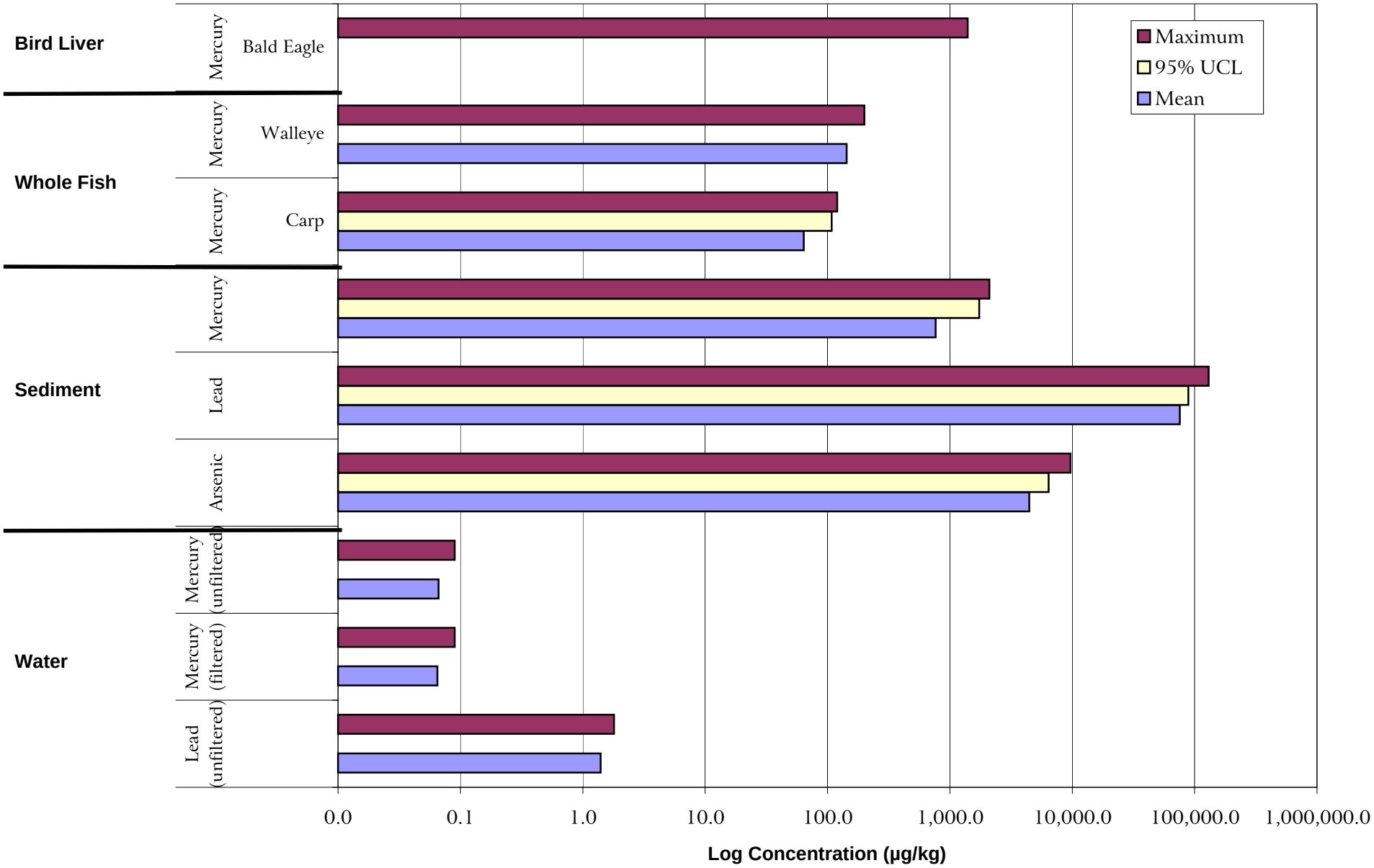


Figure 6-29 Total PCB Concentrations in Appleton to Little Rapids Reach

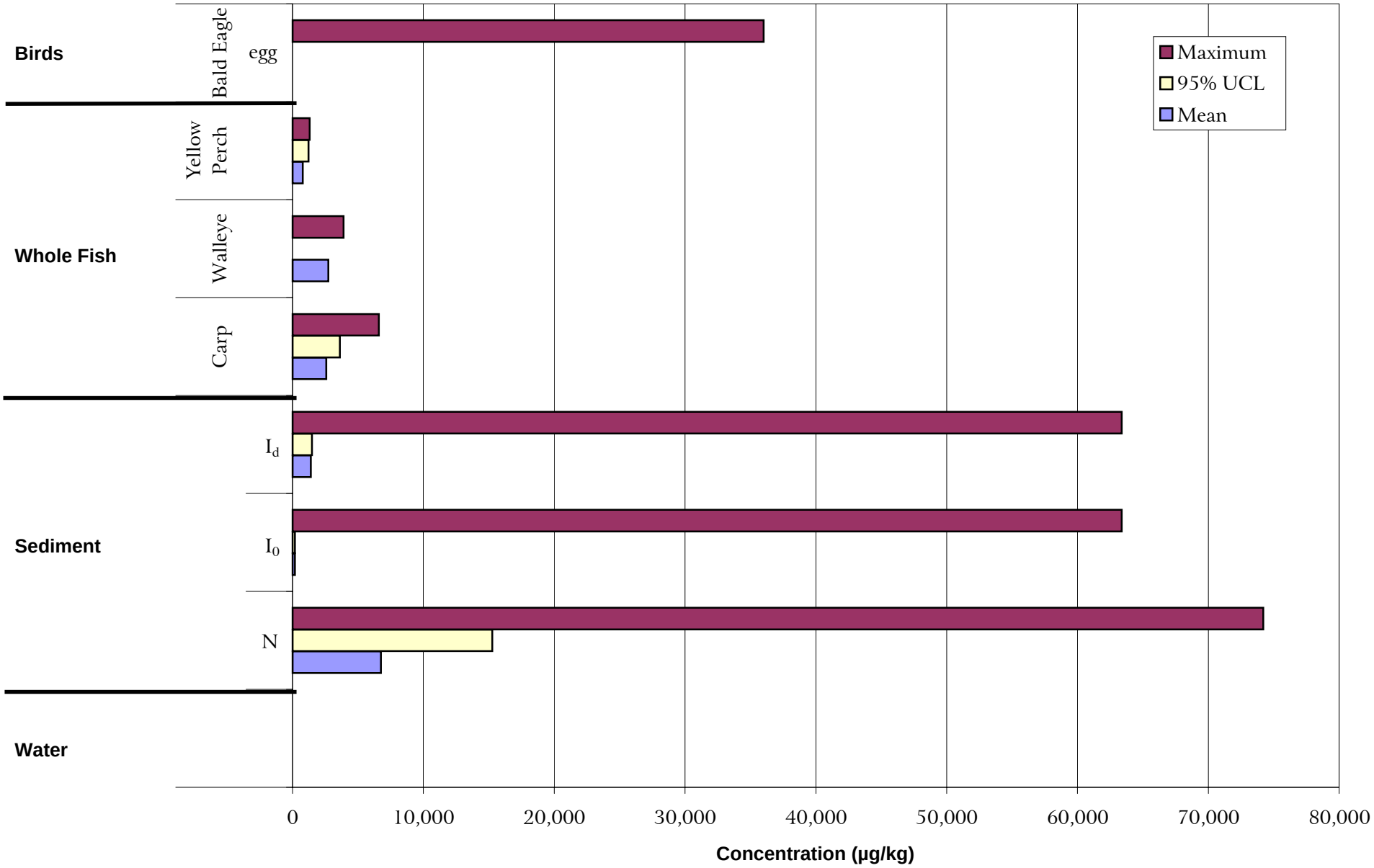


Figure 6-30 Pesticide Concentrations in Appleton to Little Rapids Reach

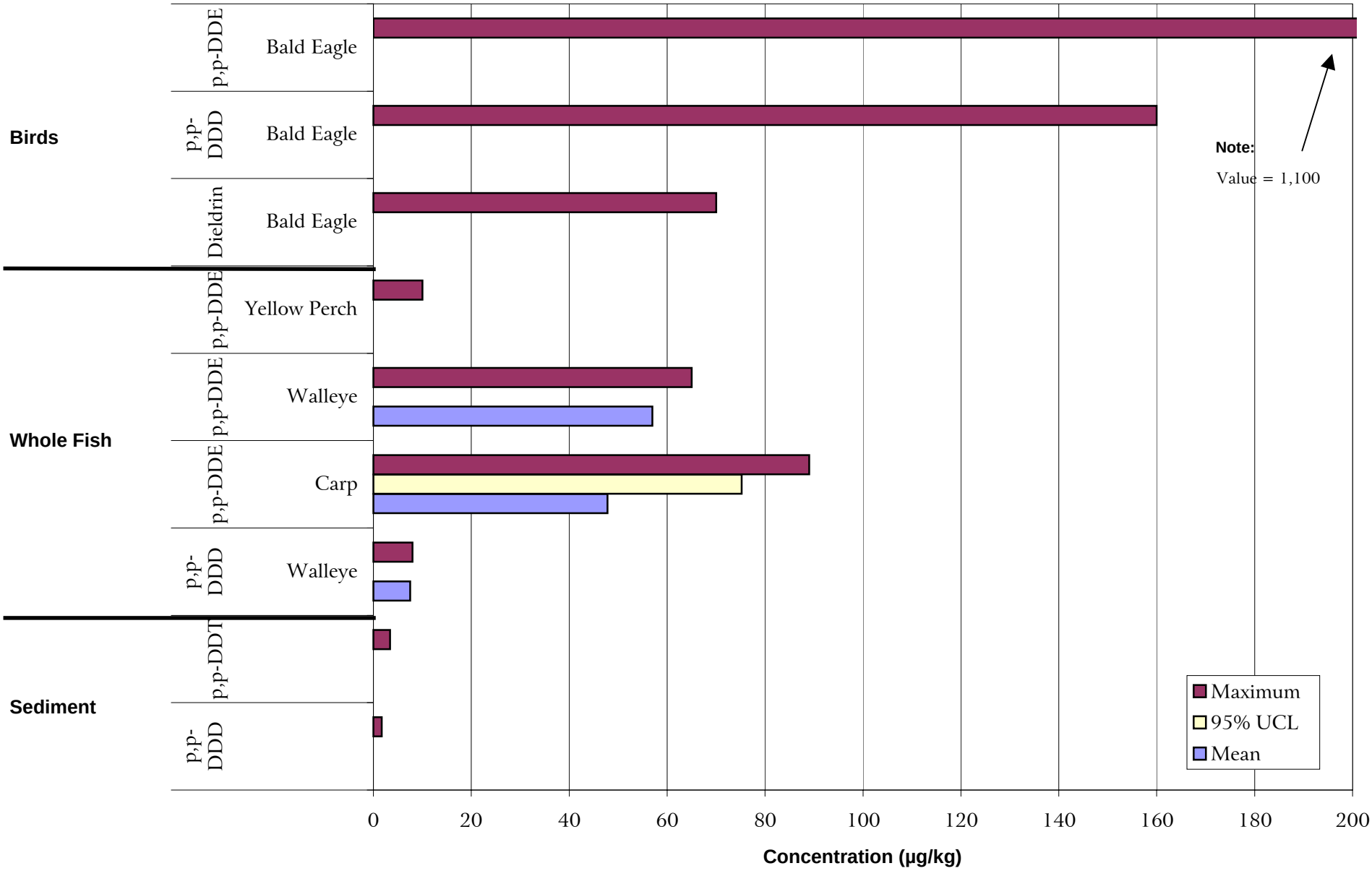


Figure 6-31 Metal Concentrations in Little Rapids to De Pere Reach

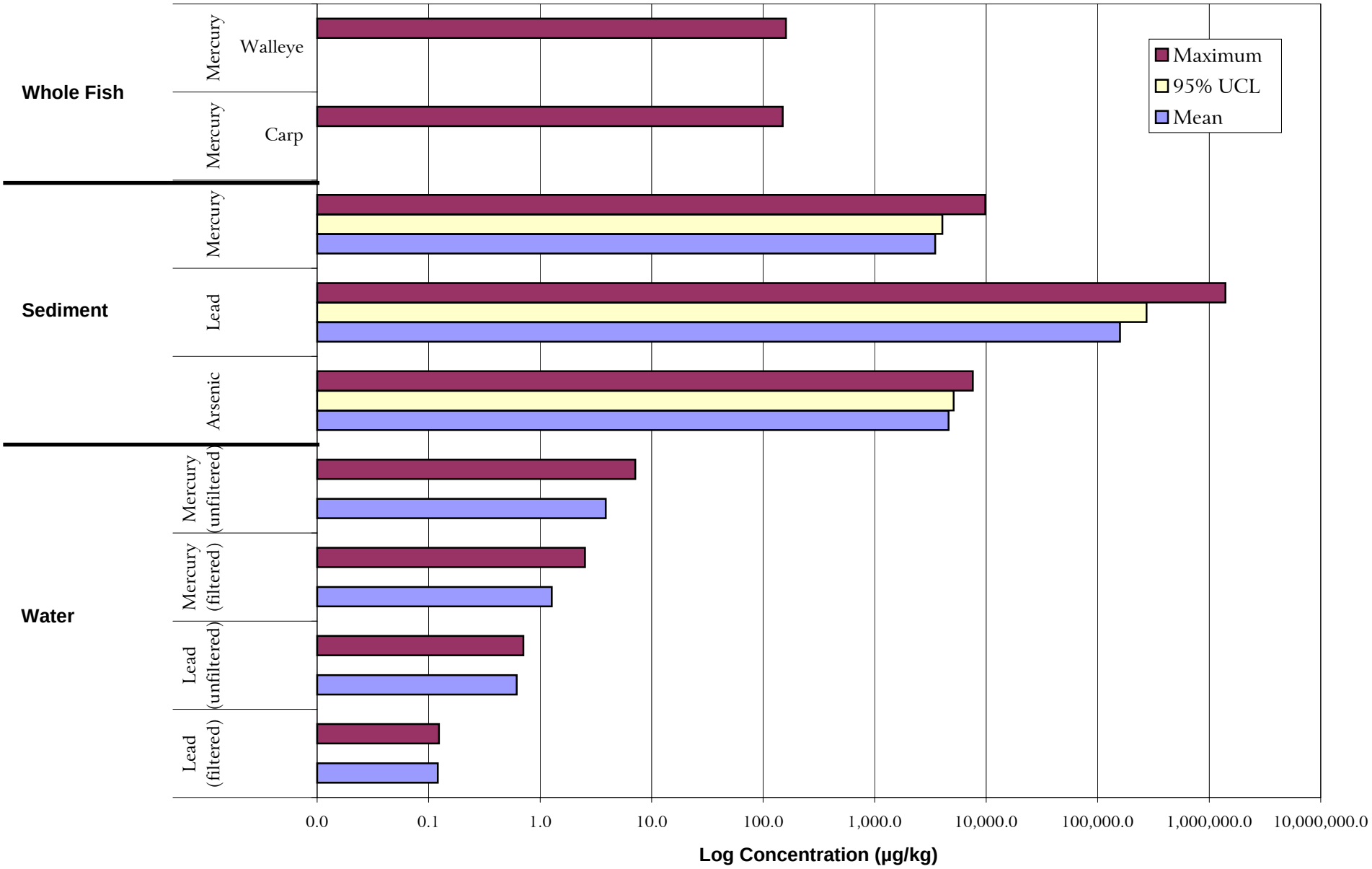


Figure 6-32 Total PCB Concentrations in Little Rapids to De Pere Reach

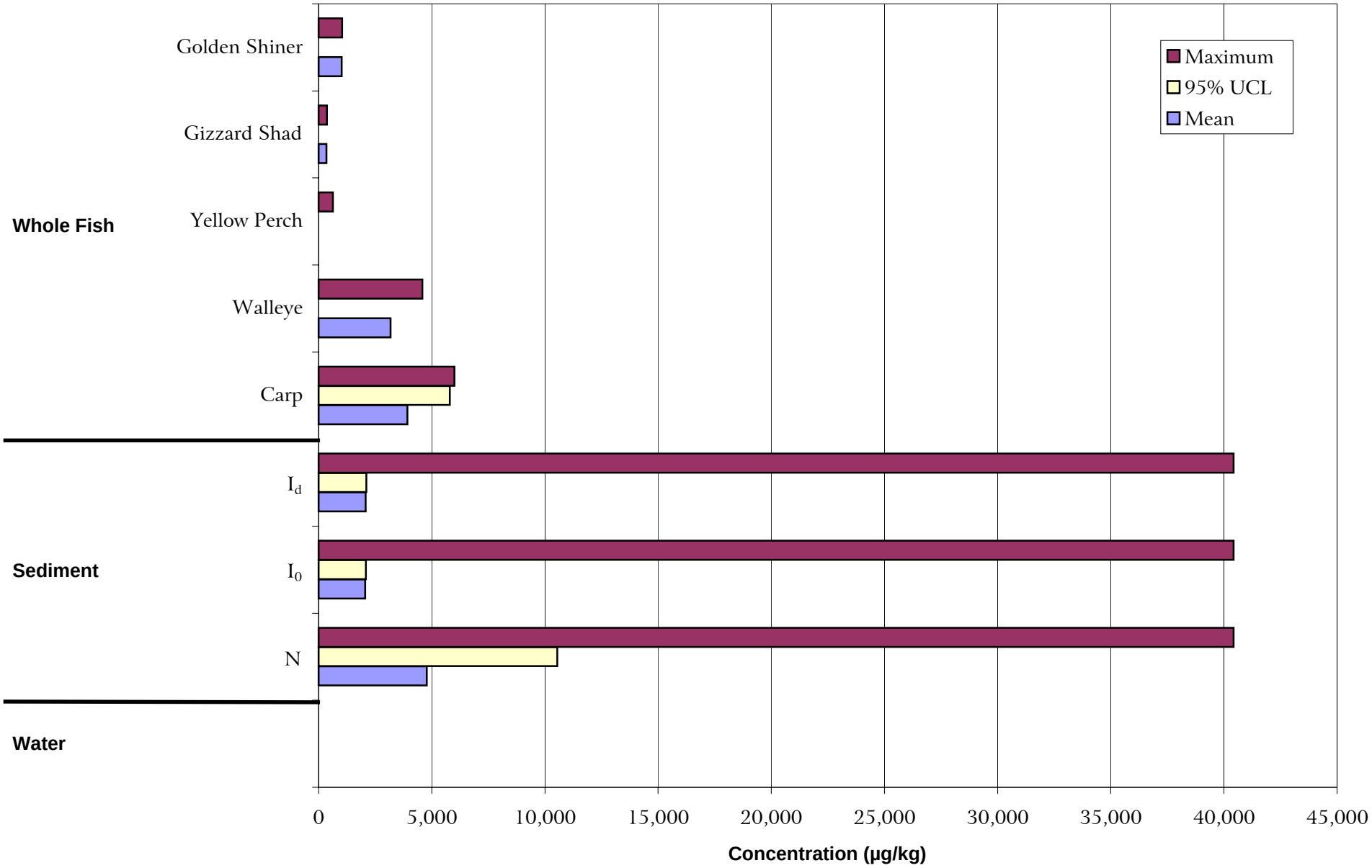


Figure 6-33 Pesticide Concentrations in Little Rapids to De Pere Reach

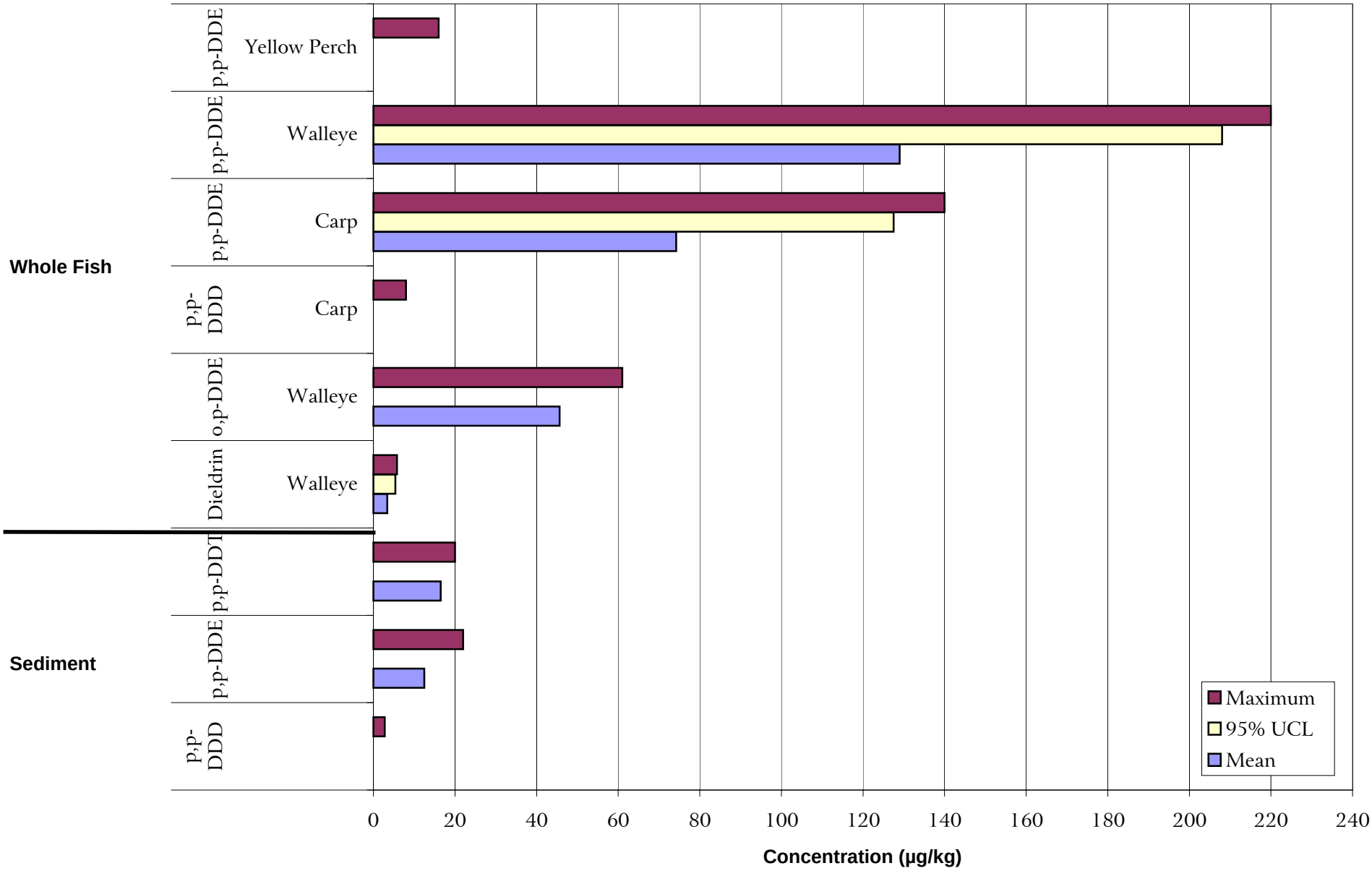


Figure 6-34 Metal Concentrations in Green Bay Zone 1

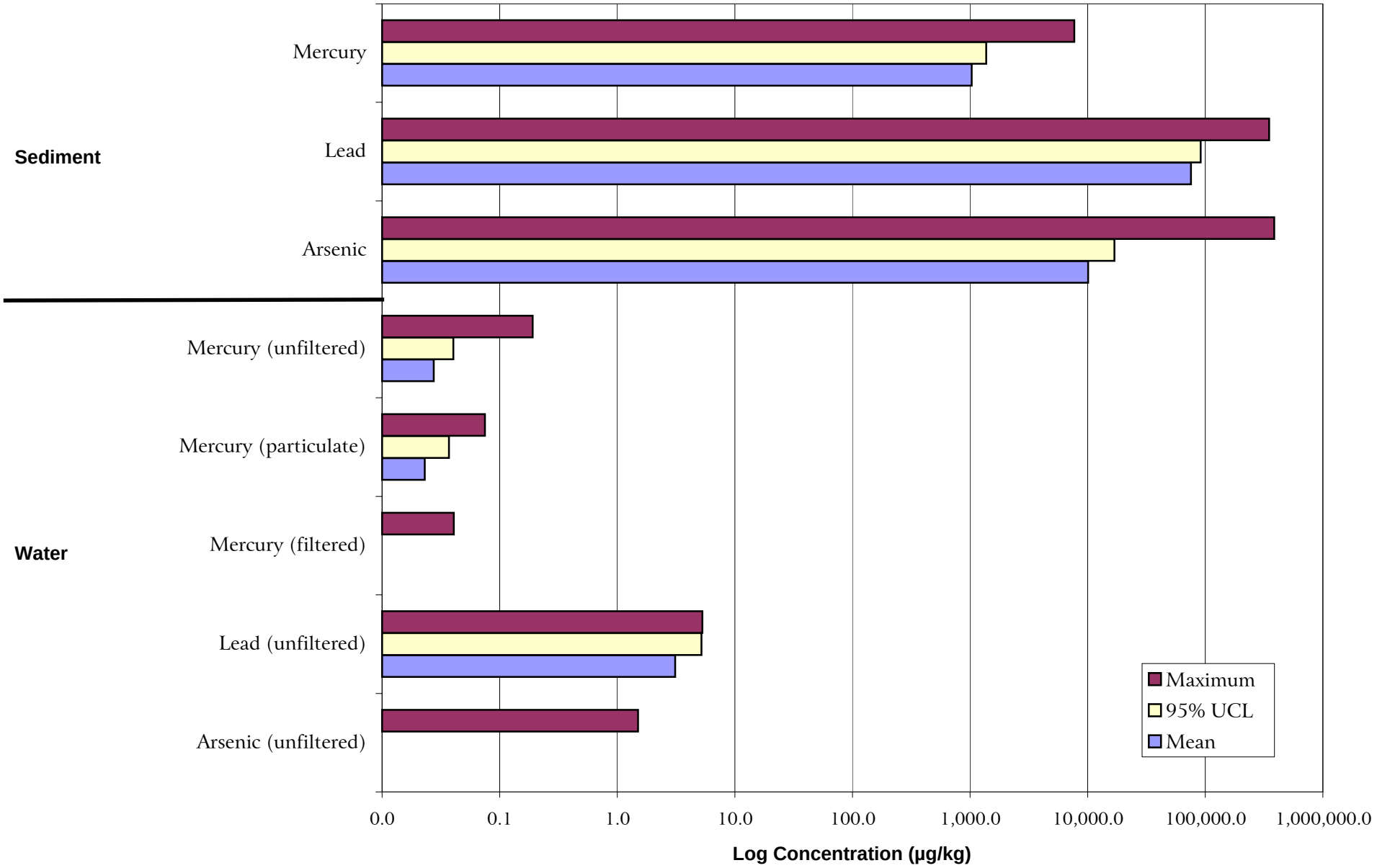


Figure 6-35 Total PCB Concentrations in Green Bay Zone 1

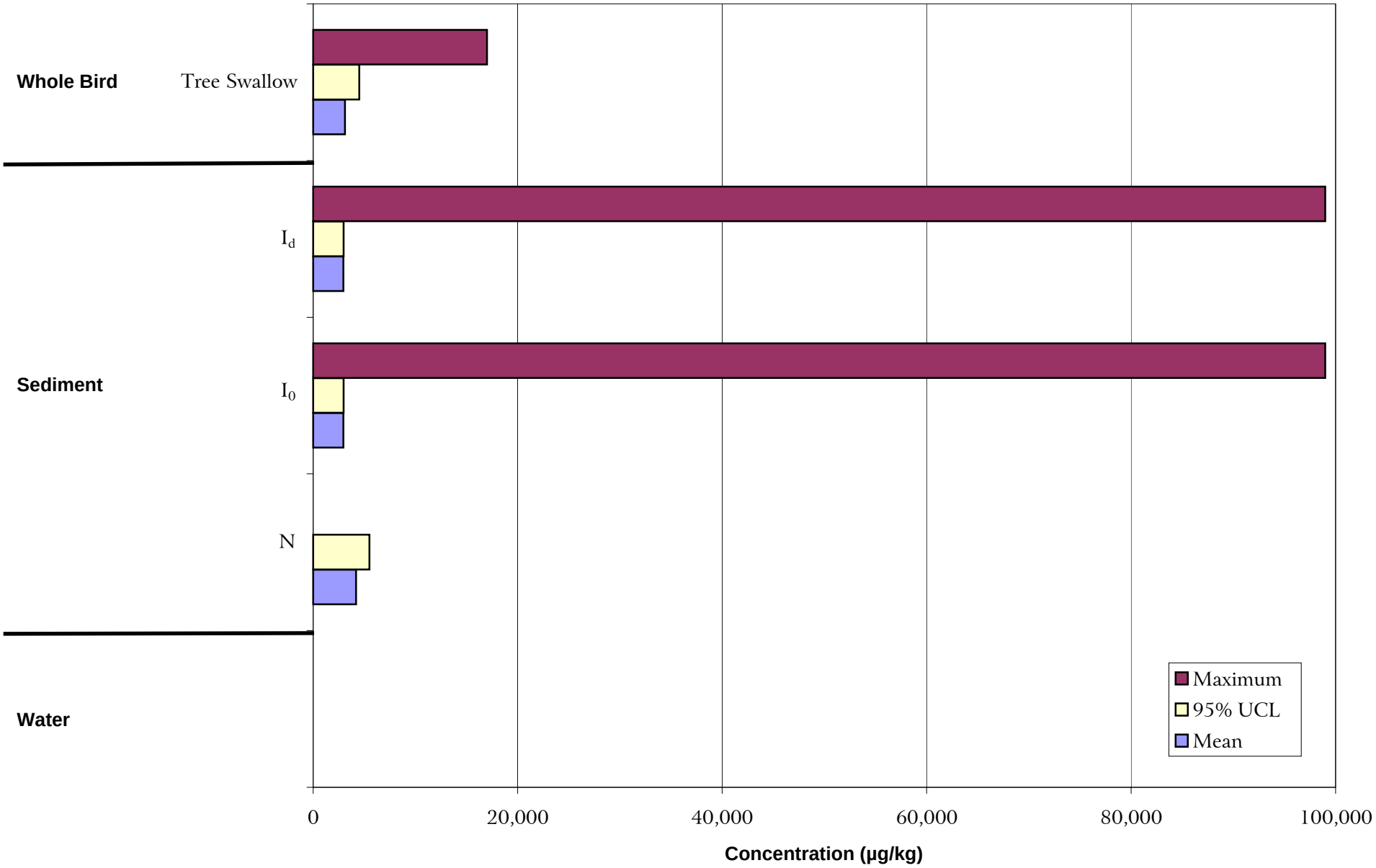


Figure 6-36 Pesticide Concentrations in Green Bay Zone 1

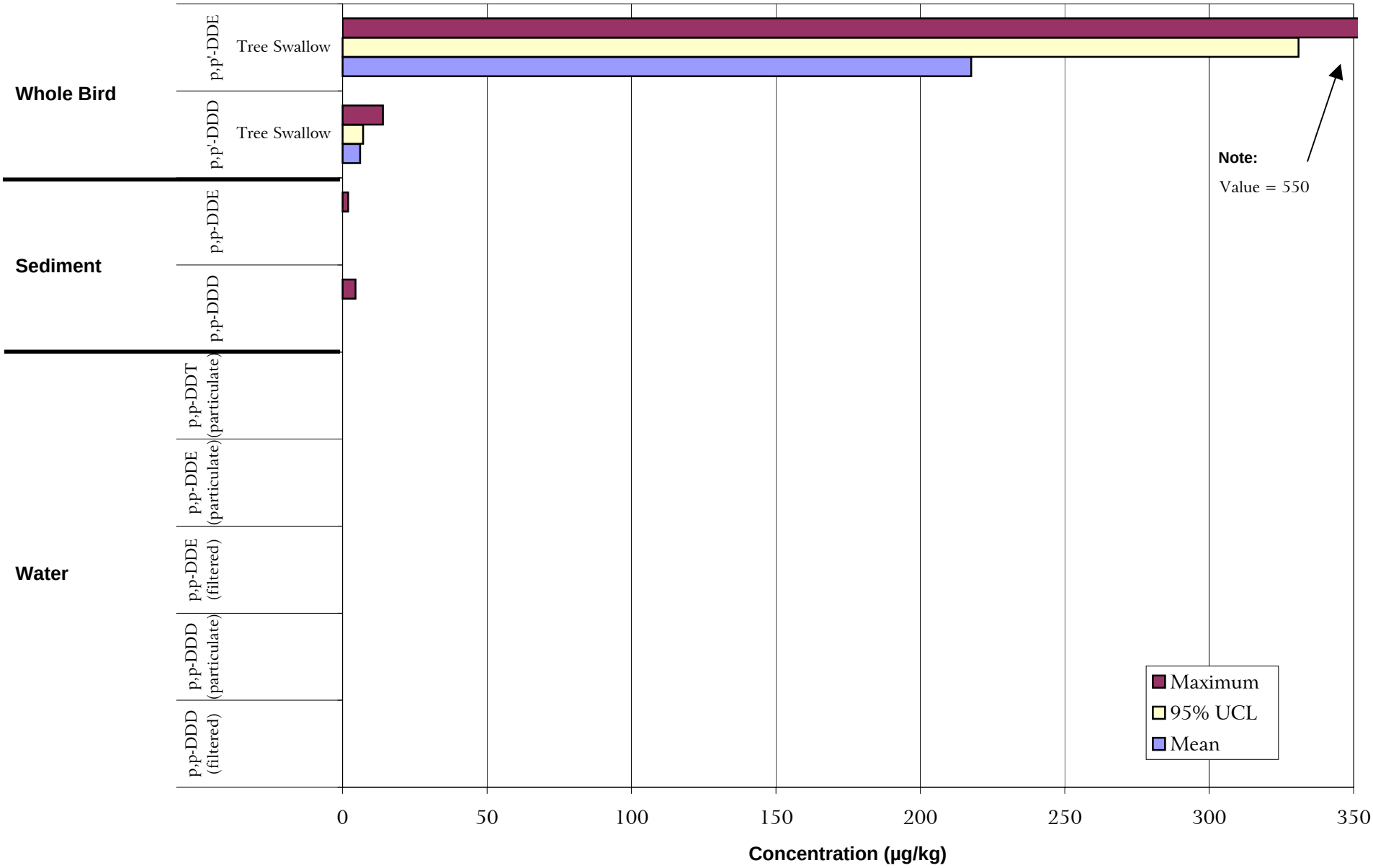


Figure 6-37 Metal Concentrations in Green Bay Zone 2

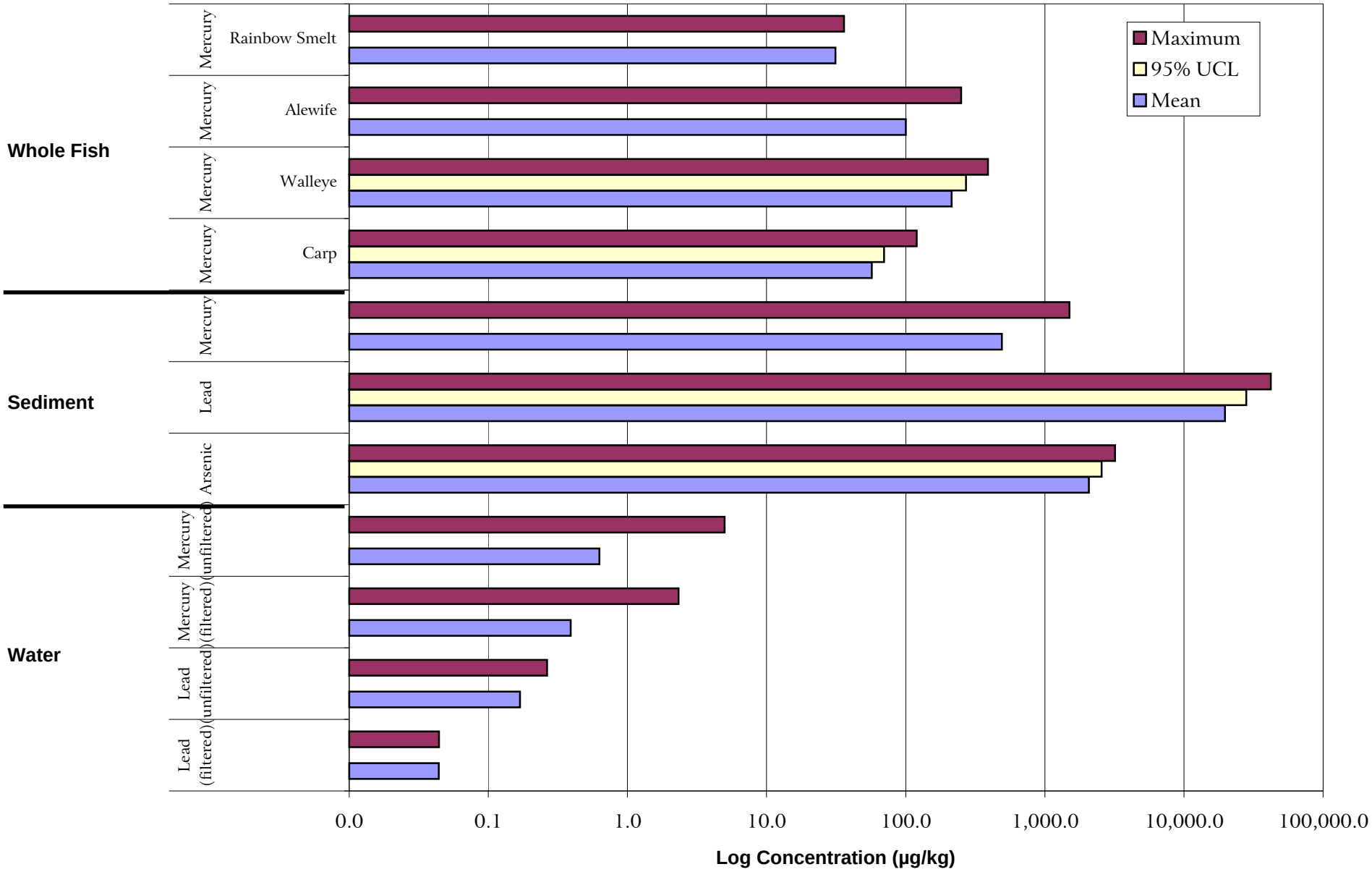


Figure 6-38 Total PCB Concentrations in Green Bay Zone 2

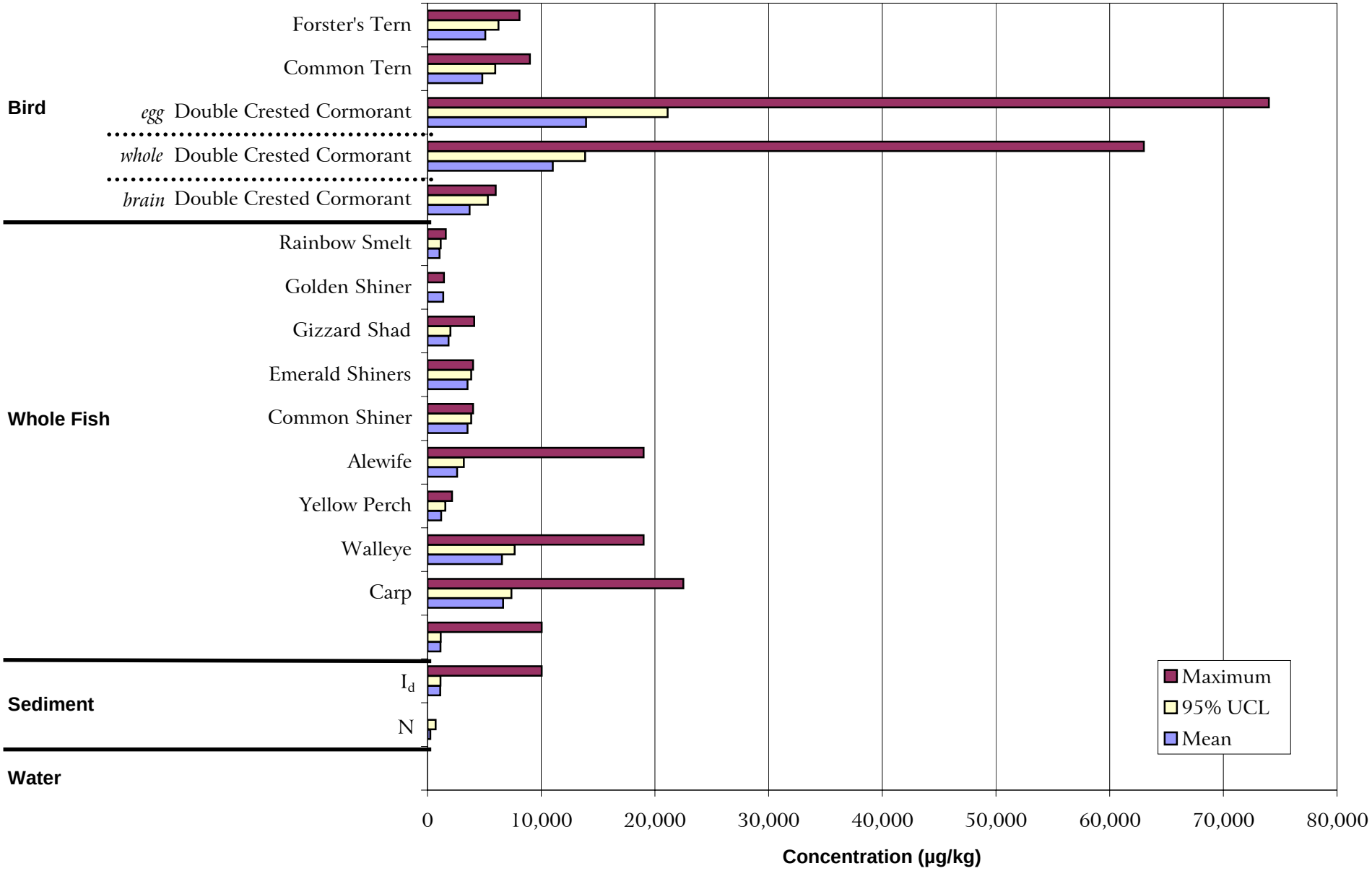


Figure 6-39 Dieldrin Concentrations in Green Bay Zone 2

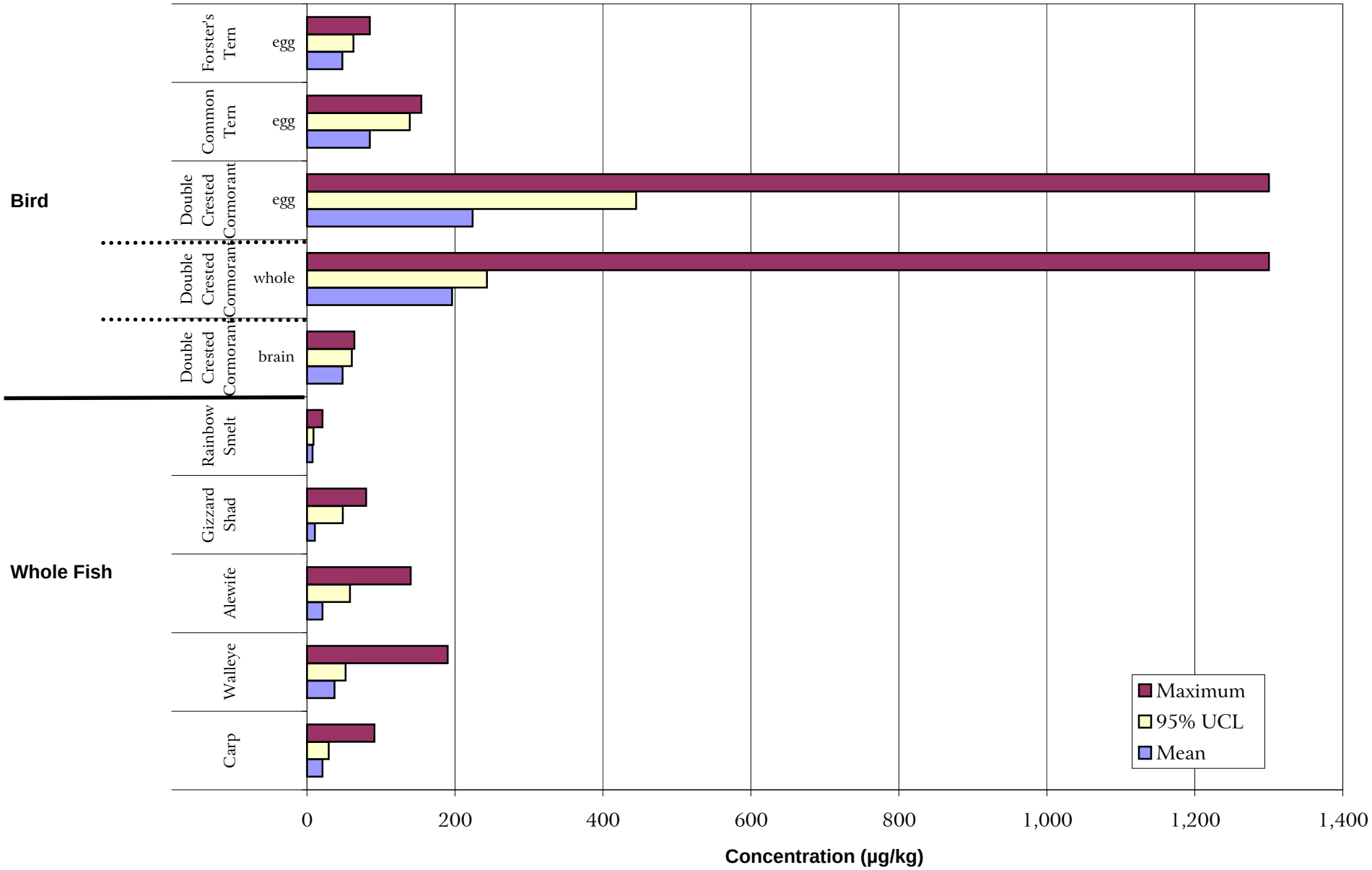


Figure 6-40 DDT Concentrations in Green Bay Zone 2

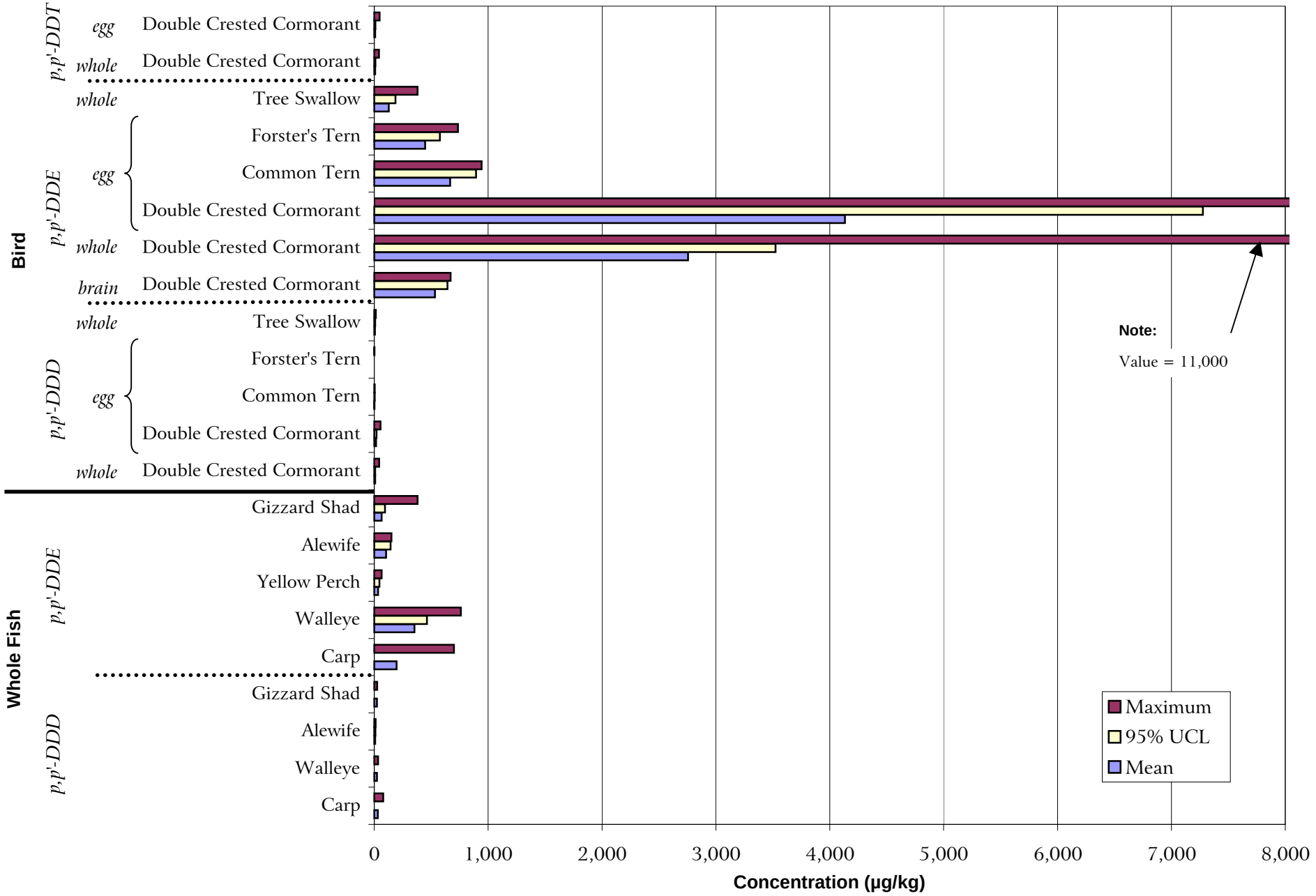


Figure 6-41 Metal Concentrations in Green Bay Zone 3A

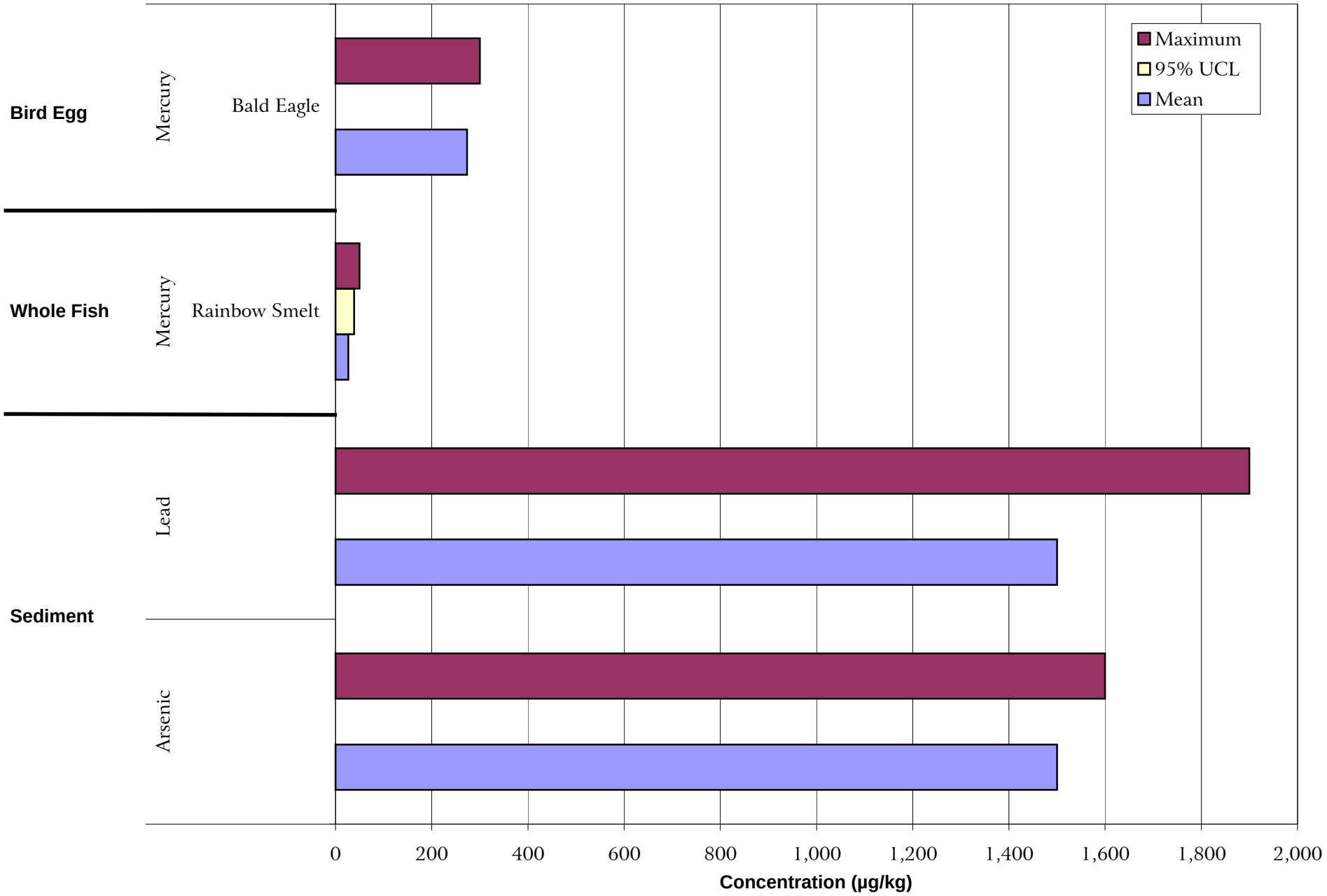


Figure 6-42 Total PCB Concentrations in Green Bay Zone 3A

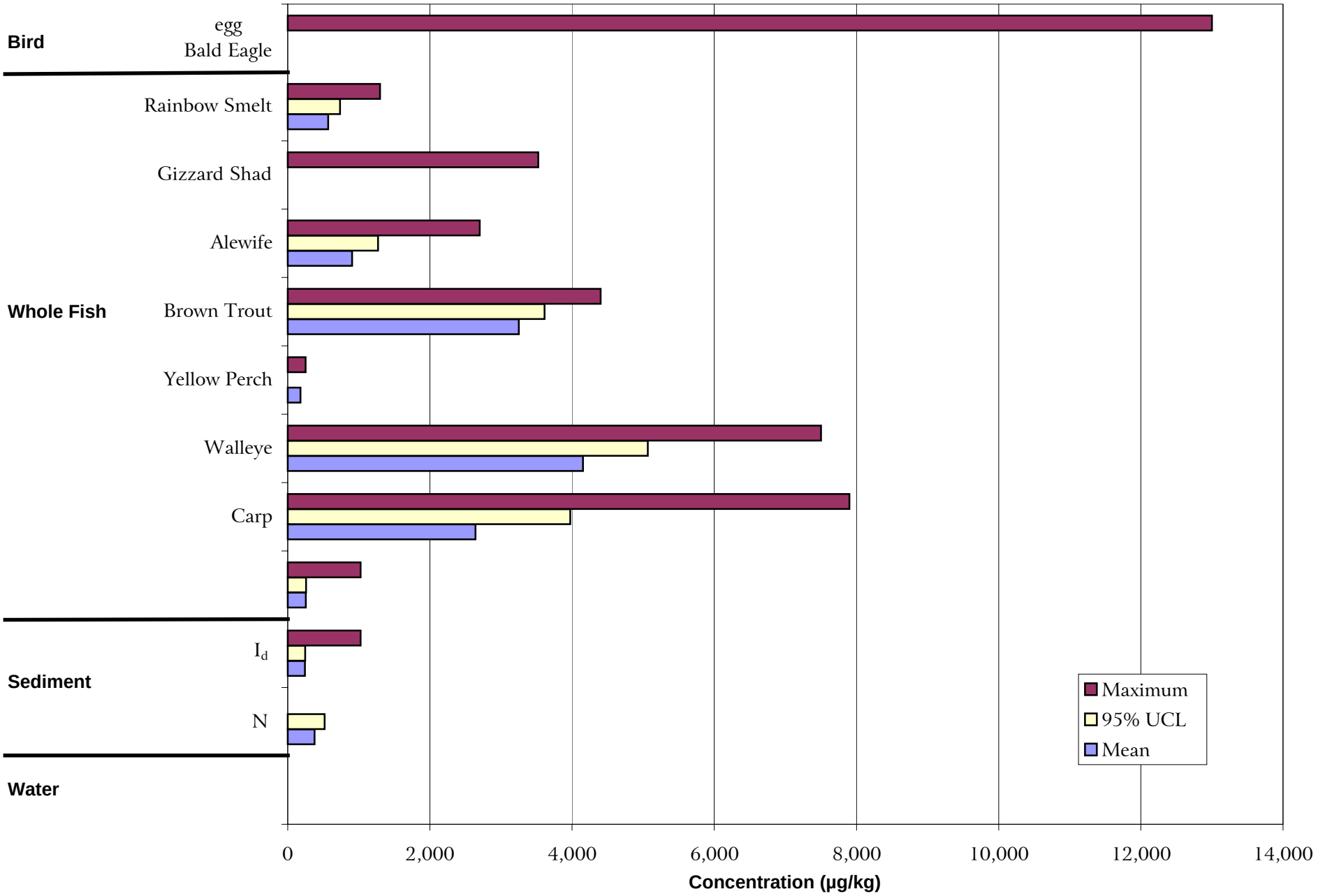


Figure 6-43 Pesticide Concentrations in Green Bay Zone 3A

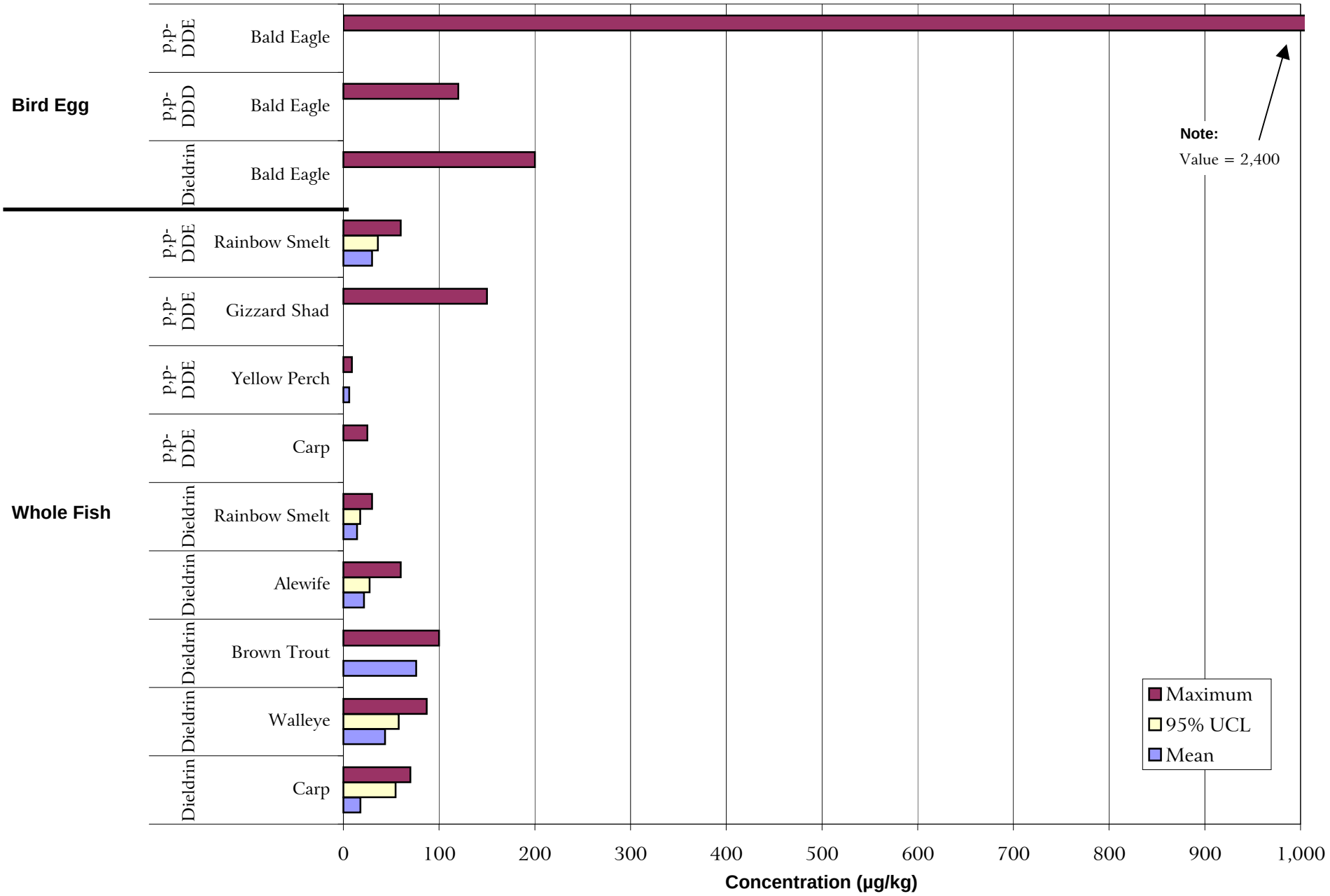


Figure 6-44 Metal Concentrations in Green Bay Zone 3B

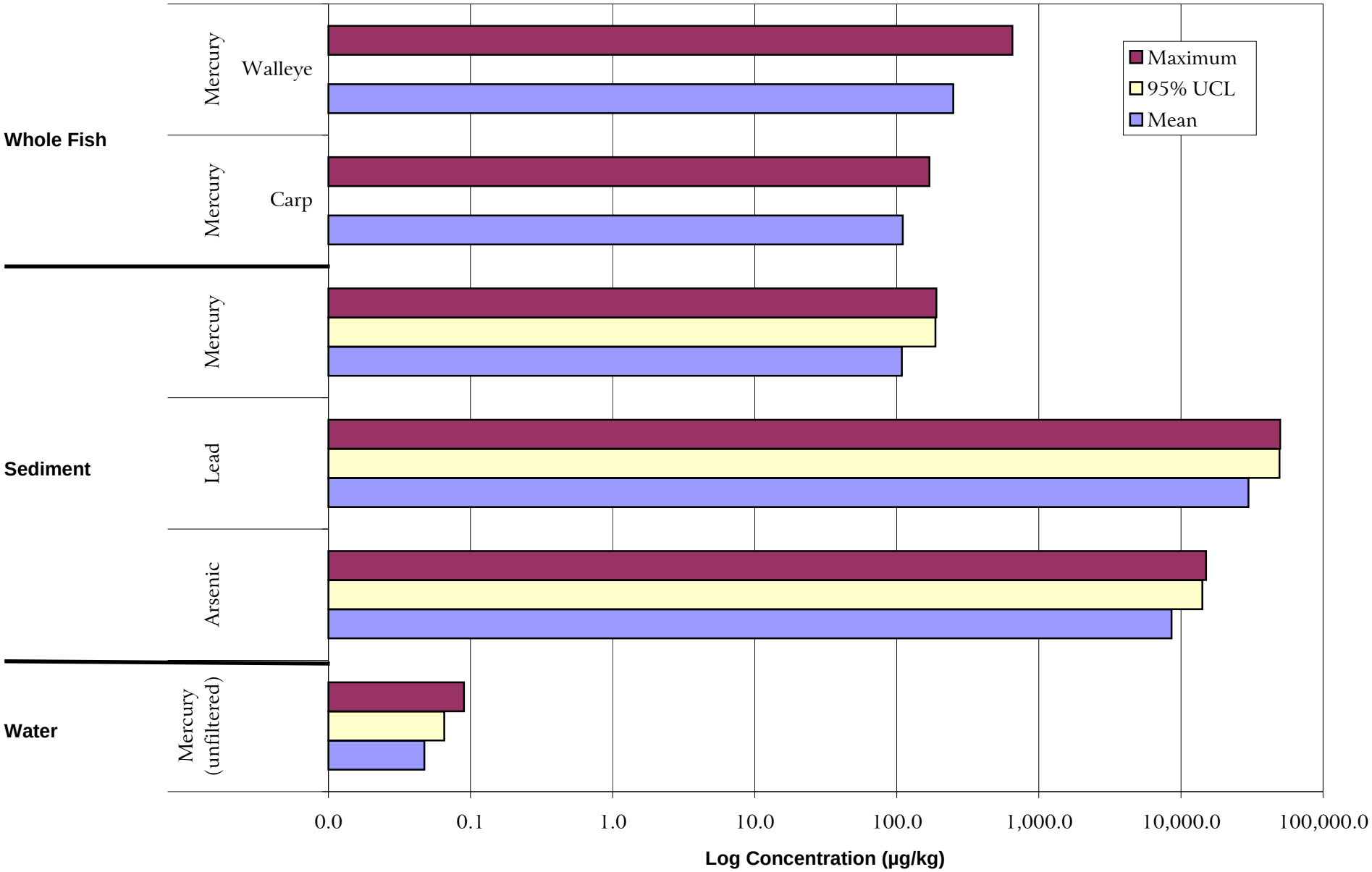


Figure 6-45 Total PCB Concentrations in Green Bay Zone 3B

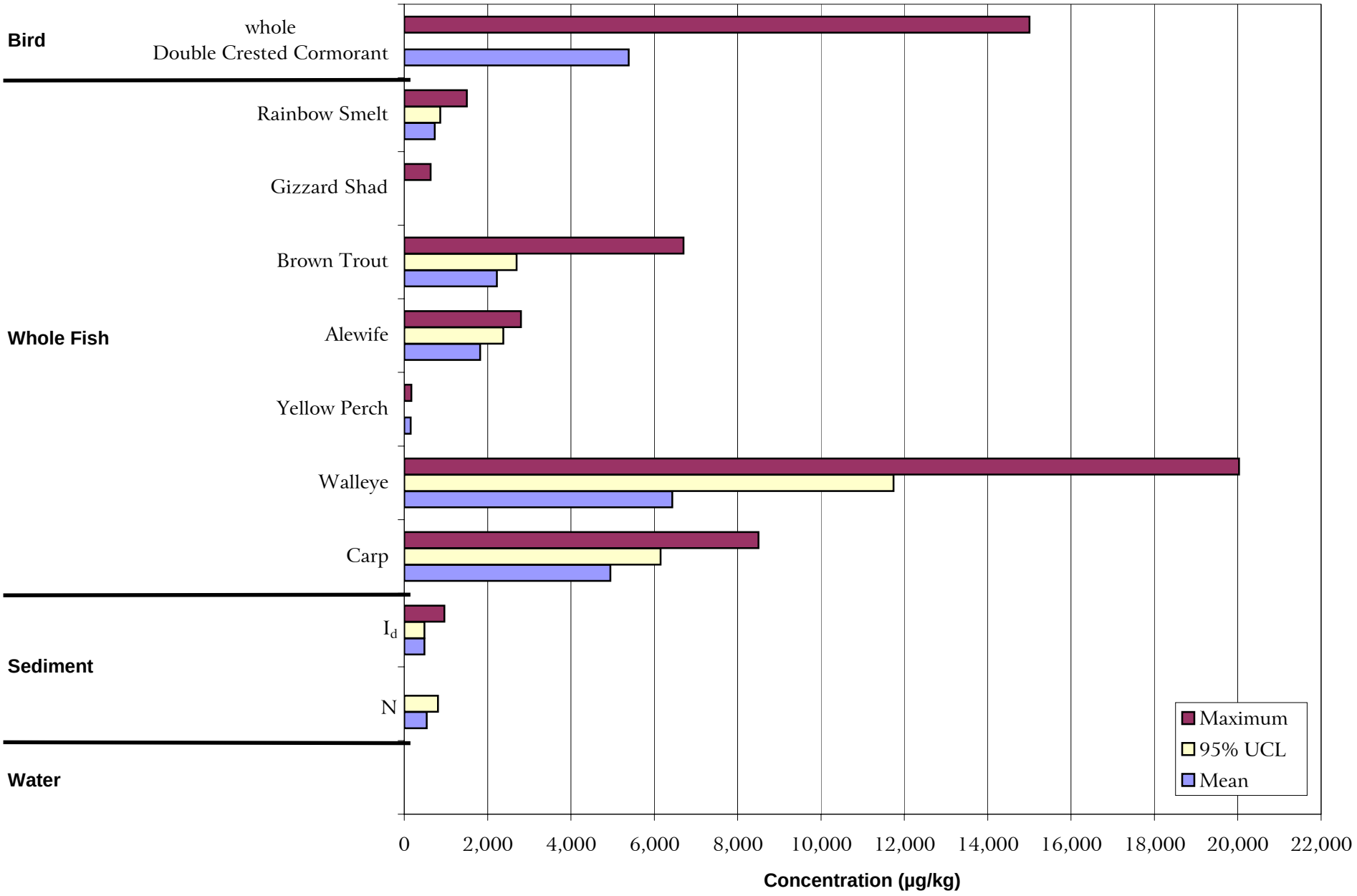


Figure 6-46 Pesticide Concentrations in Green Bay Zone 3B

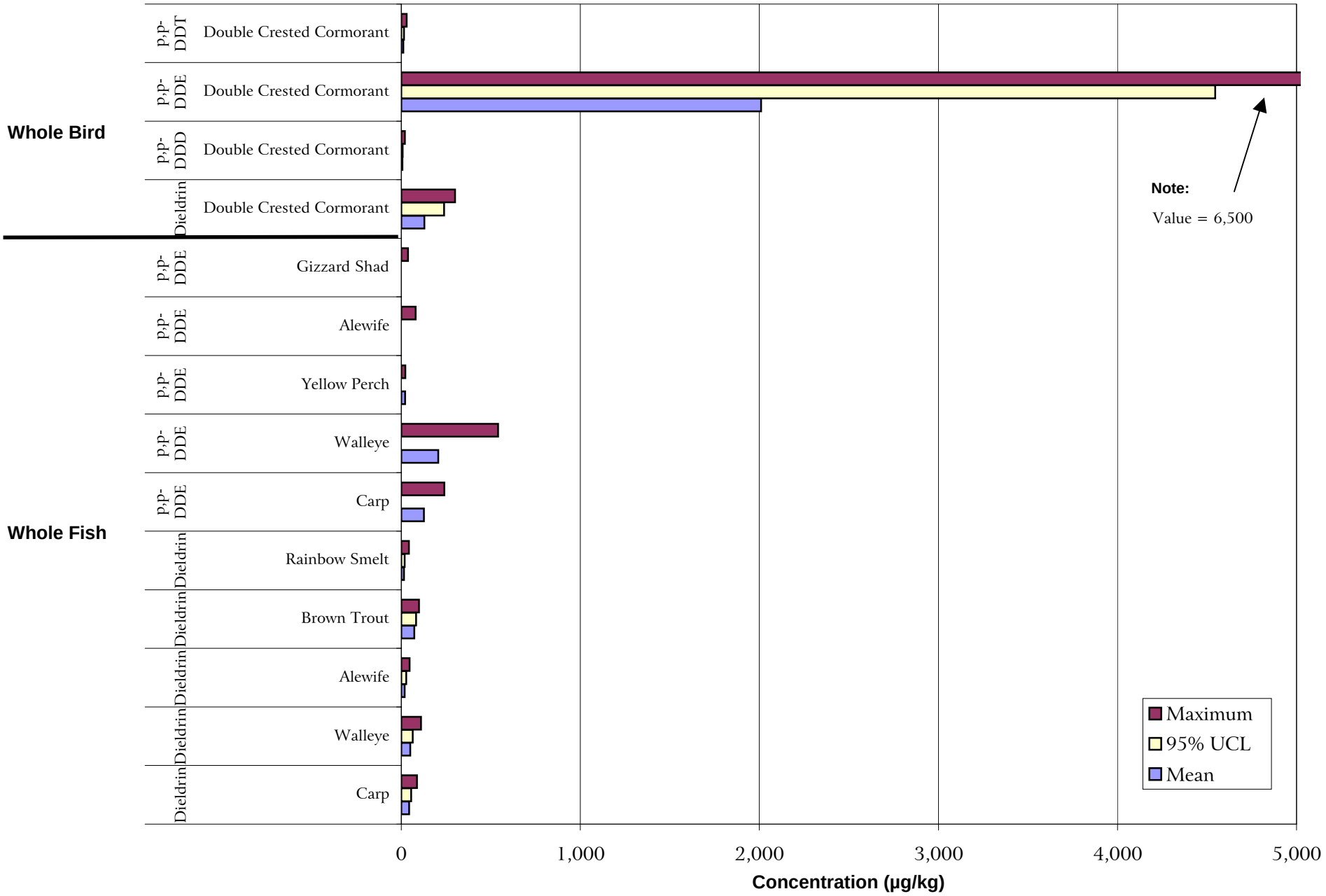


Figure 6-47 Metal Concentrations in Green Bay Zone 4

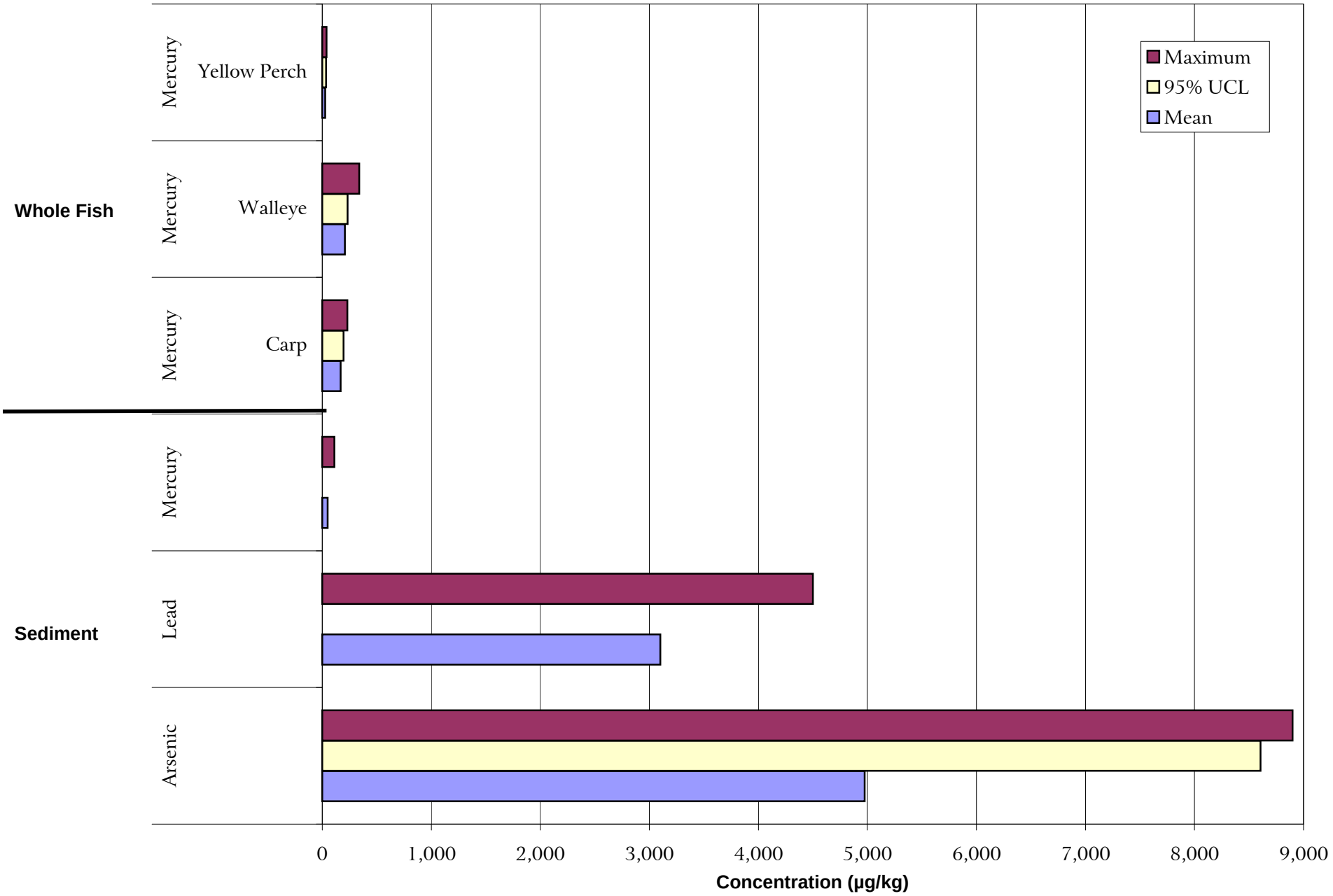


Figure 6-48 Total PCB Concentrations in Green Bay Zone 4

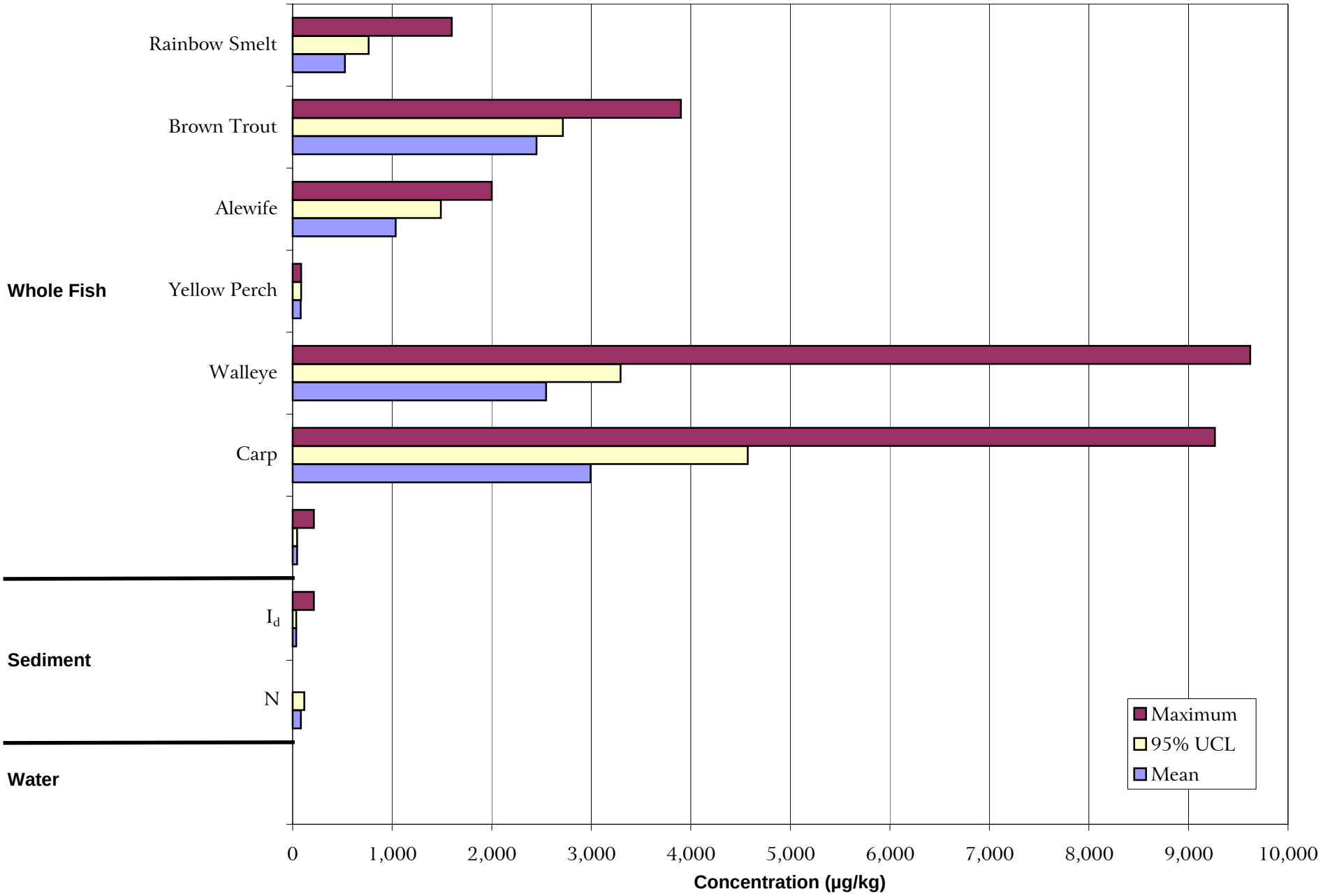


Figure 6-49 Pesticide Concentrations in Green Bay Zone 4

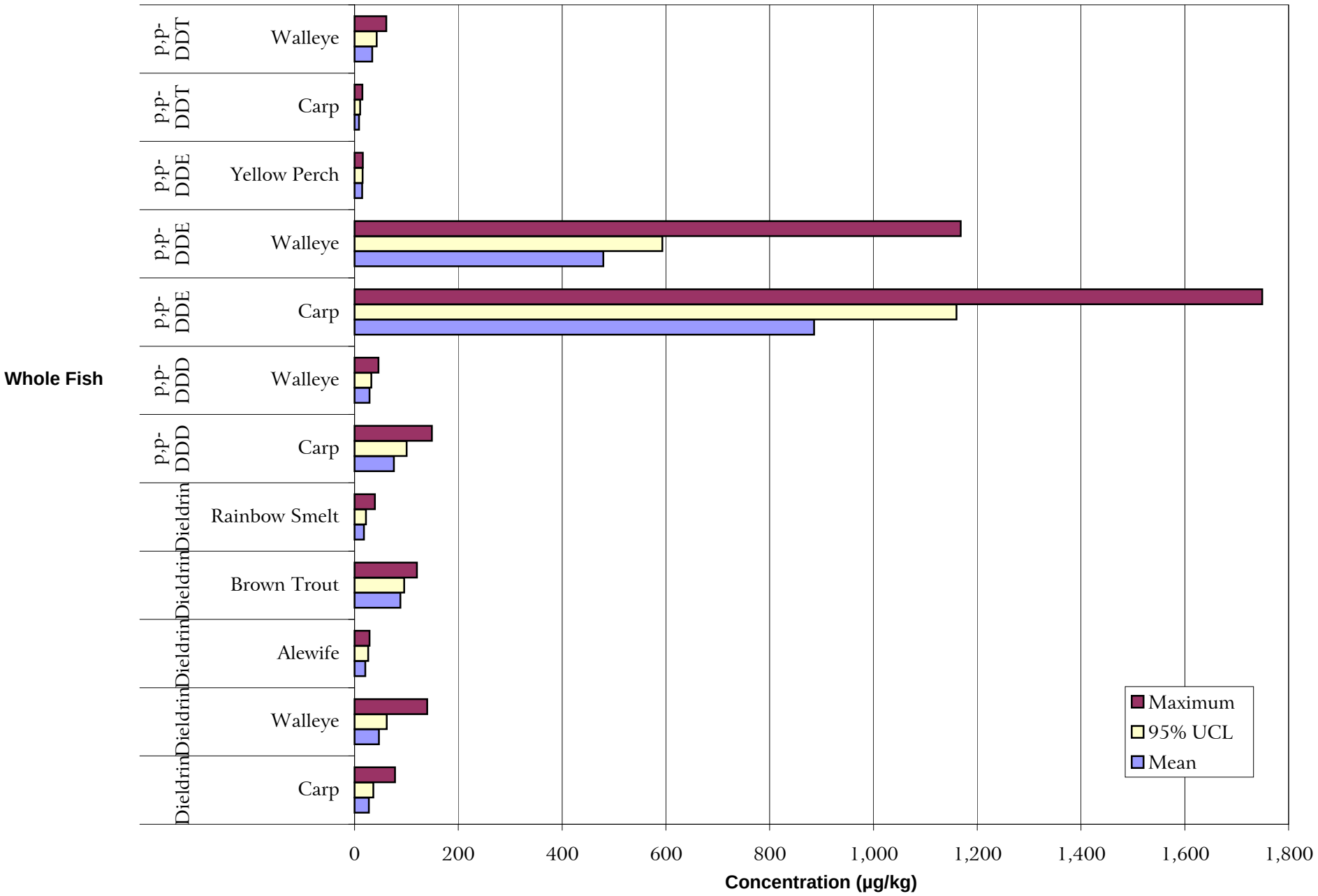


Figure 6-50 Surface Water Hazard Quotients that Exceeded 1.0

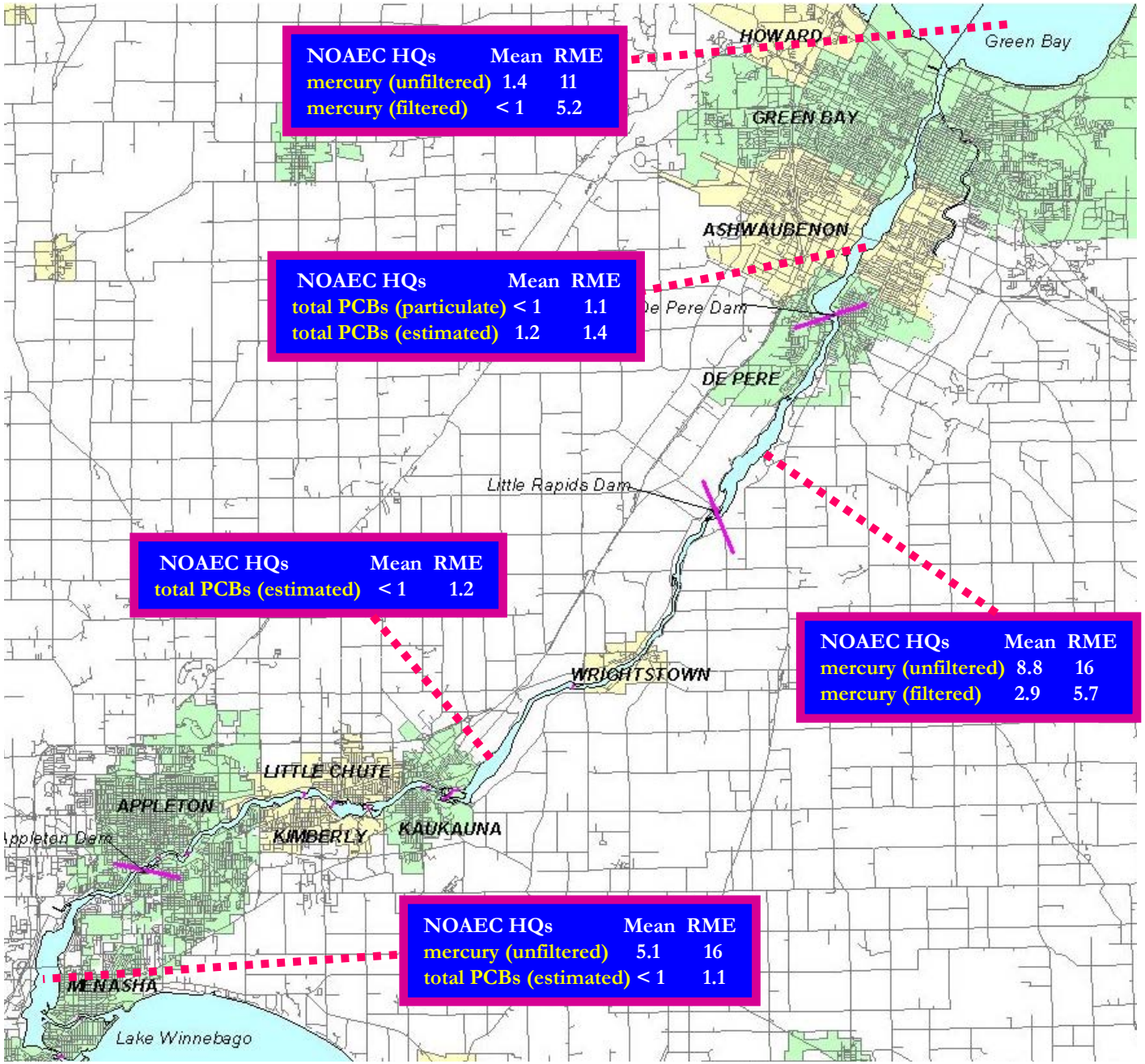


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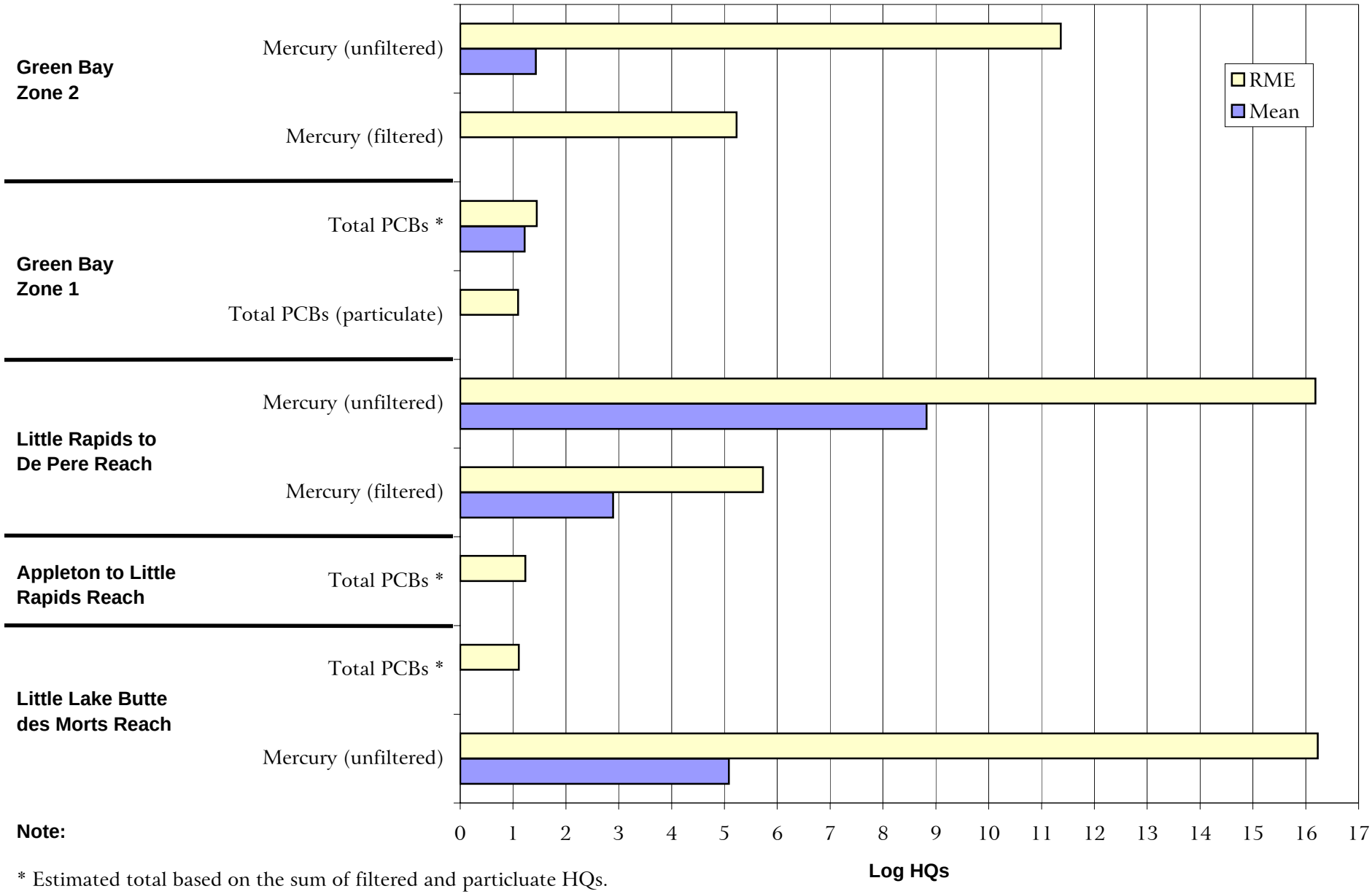
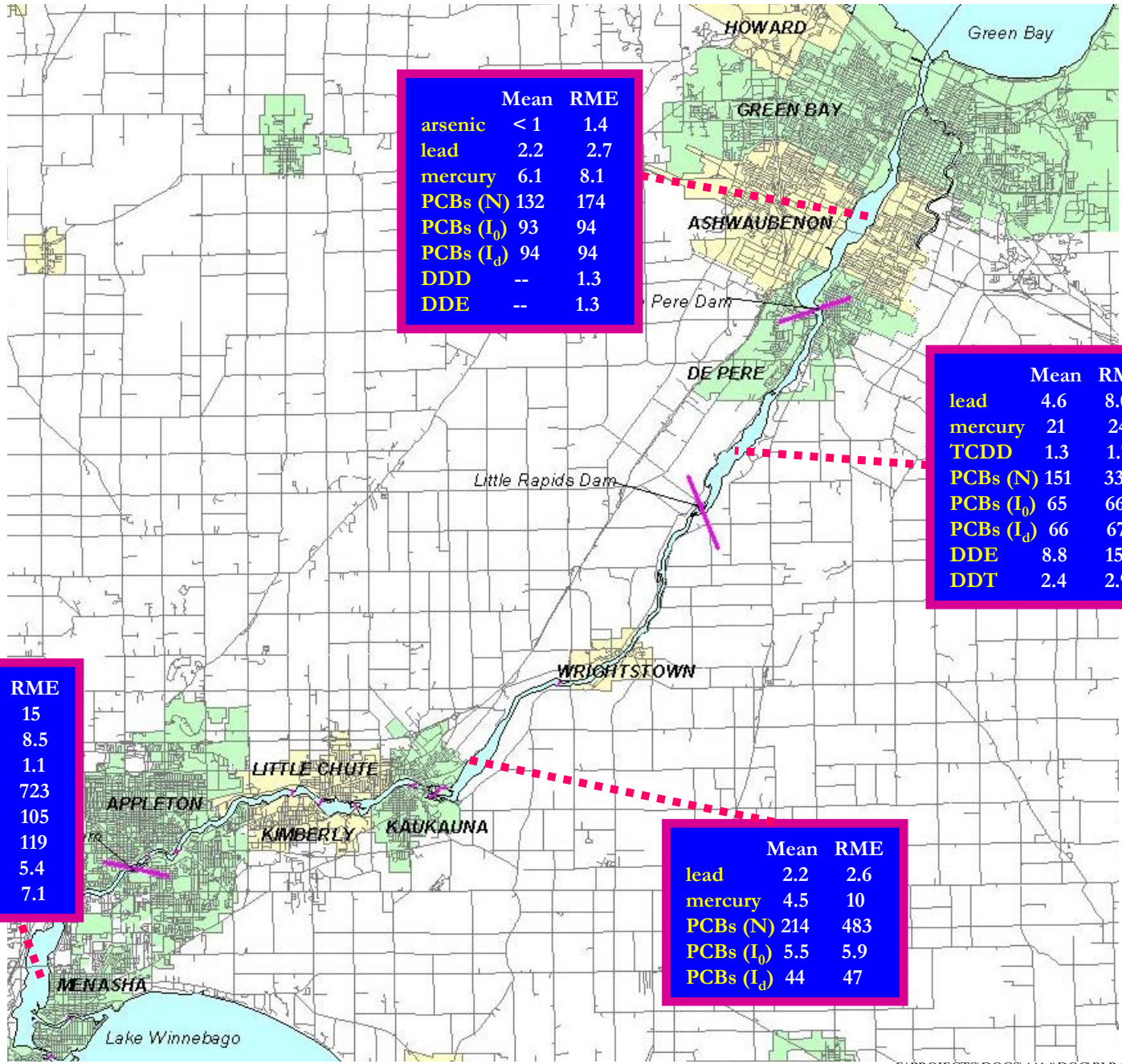
Figure 6-51 Surface Water Hazard Quotients Exceeding 1.0



Note:

* Estimated total based on the sum of filtered and particluate HQs.

Figure 6-52 Surface Sediment Hazard Quotients that Exceeded 1.0



	Mean	RME
arsenic	< 1	1.4
lead	2.2	2.7
mercury	6.1	8.1
PCBs (N)	132	174
PCBs (I_0)	93	94
PCBs (I_d)	94	94
DDD	--	1.3
DDE	--	1.3

	Mean	RME
lead	4.6	8.0
mercury	21	24
TCDD	1.3	1.7
PCBs (N)	151	334
PCBs (I_0)	65	66
PCBs (I_d)	66	67
DDE	8.8	15
DDT	2.4	2.9

	Mean	RME
lead	5.0	15
mercury	5.6	8.5
TCDD	< 1	1.1
PCBs (N)	339	723
PCBs (I_0)	104	105
PCBs (I_d)	117	119
DDD	5.0	5.4
DDT	--	7.1

	Mean	RME
lead	2.2	2.6
mercury	4.5	10
PCBs (N)	214	483
PCBs (I_0)	5.5	5.9
PCBs (I_d)	44	47

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Figure 6-53 Surface Sediment Hazard Quotients that Exceeded 1.0

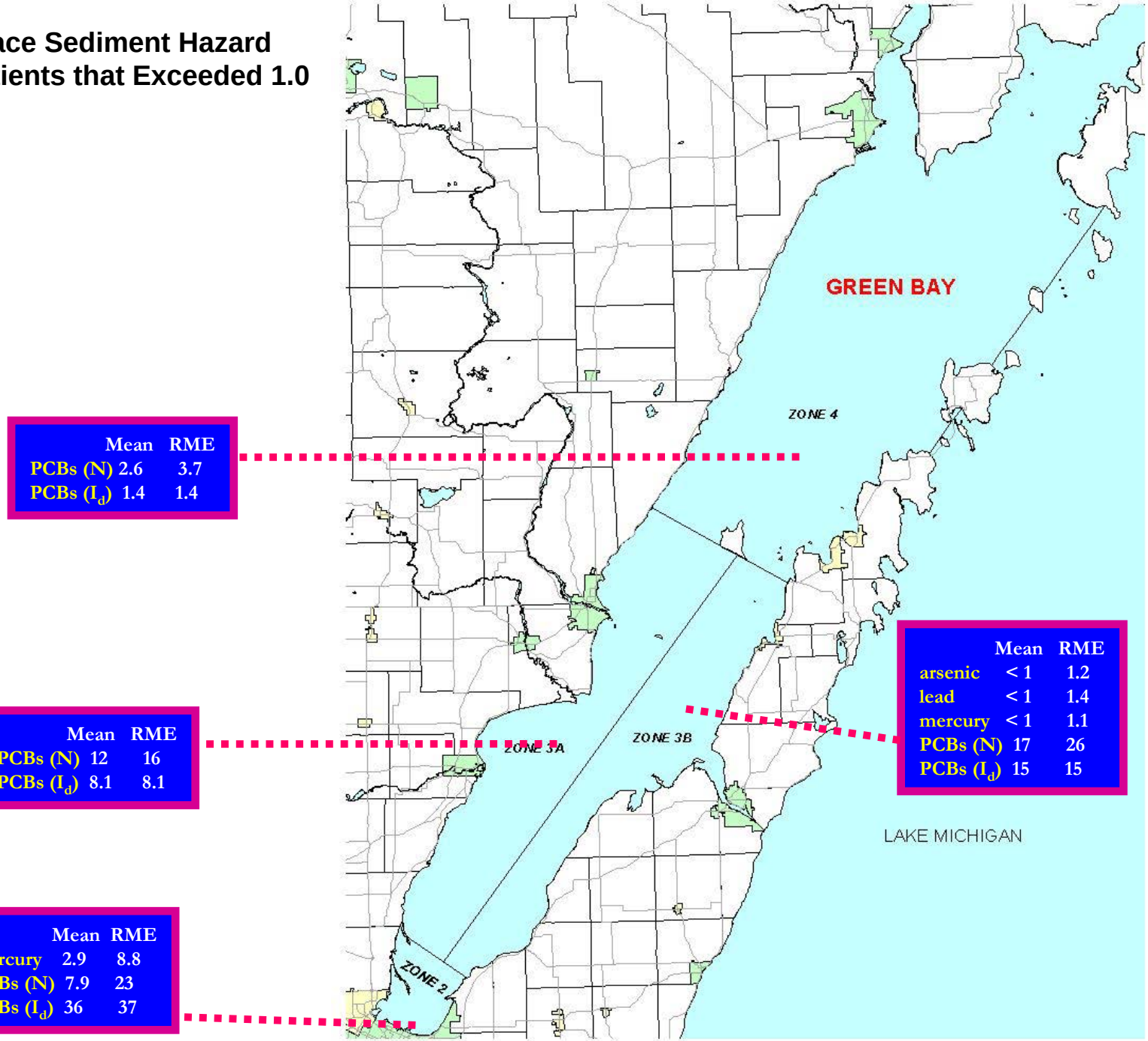

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Figure 6-54 Surface Sediment Metals Hazard Quotients that Exceeded 1.0

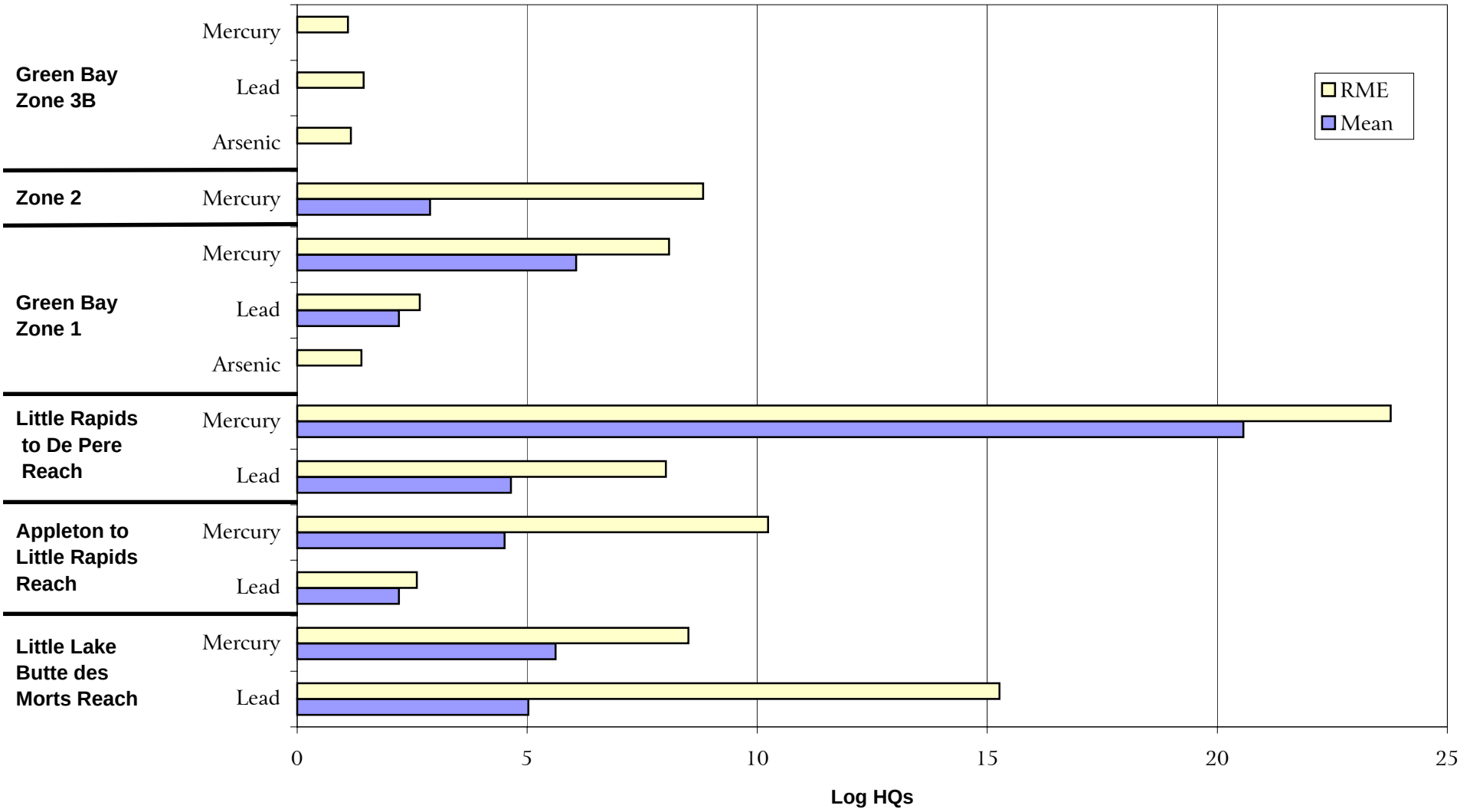

Figure 6-55 Surface Sediment Total PCB Hazard Quotients that Exceeded 1.0

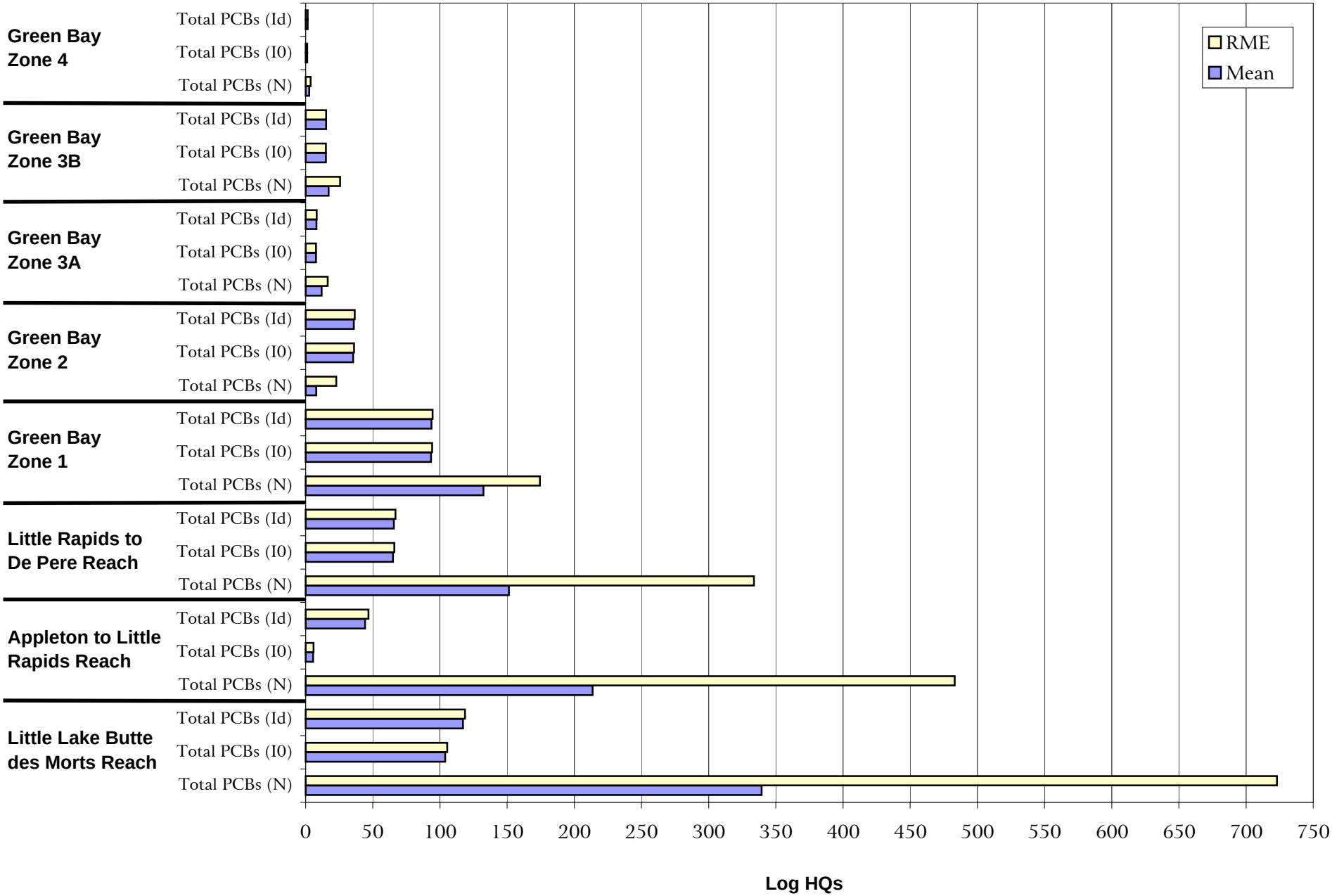

Figure 6-56 Surface Sediment DDT Hazard Quotients that Exceeded 1.0

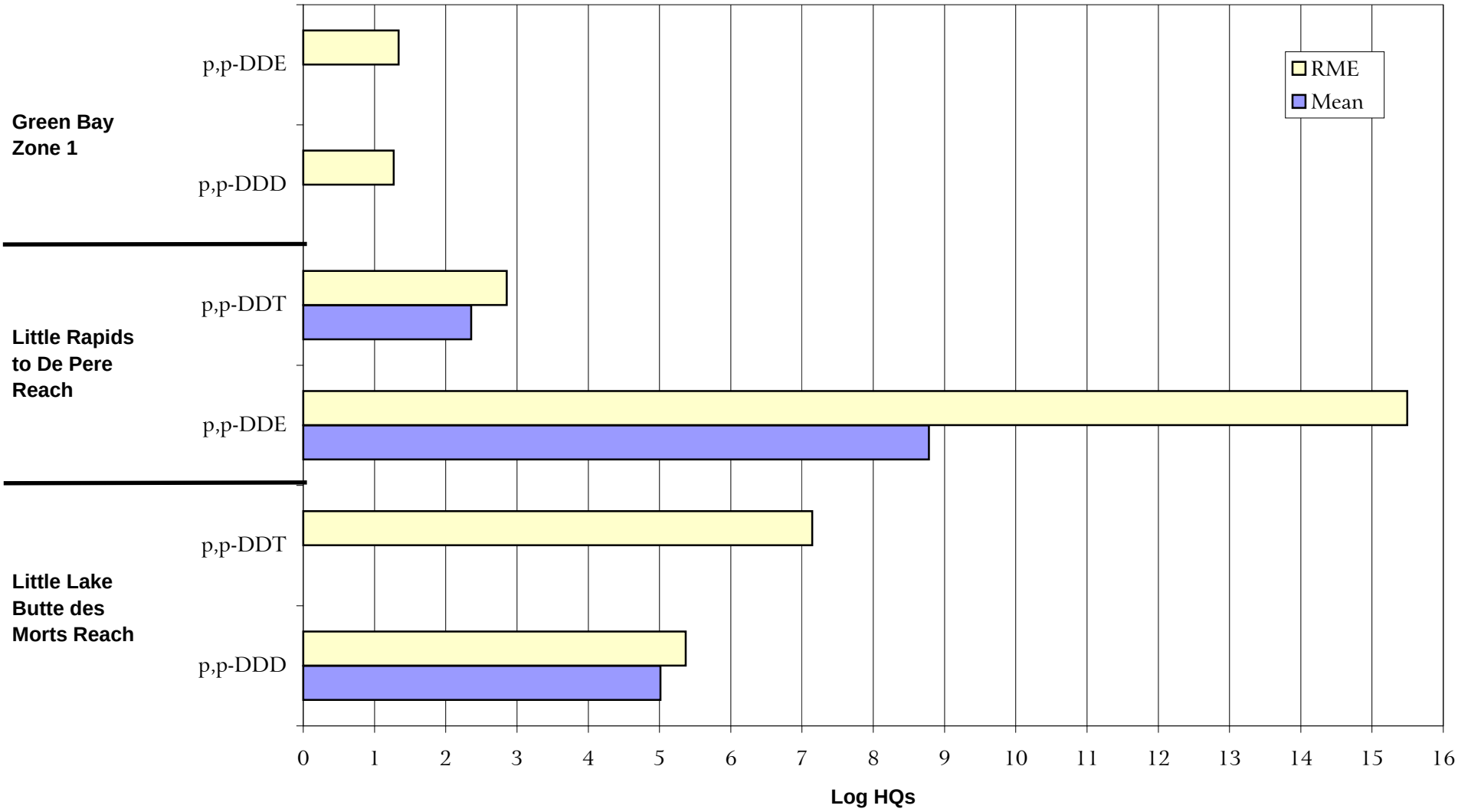

Figure 6-57 Whole Fish Hazard Quotients that Exceeded 1.0

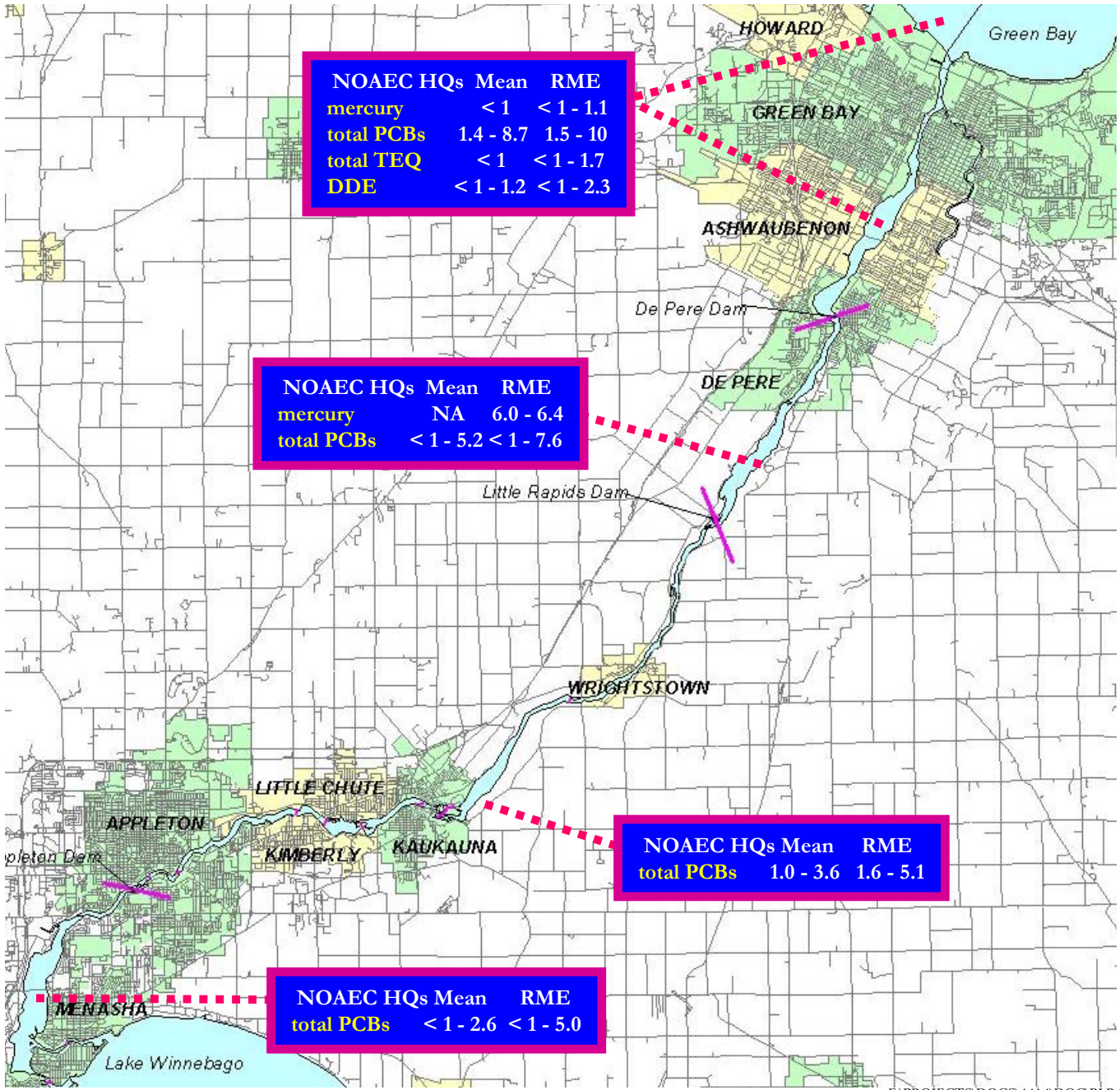

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Figure 6-58 **Whole Fish Hazard Quotients that Exceeded 1.0**

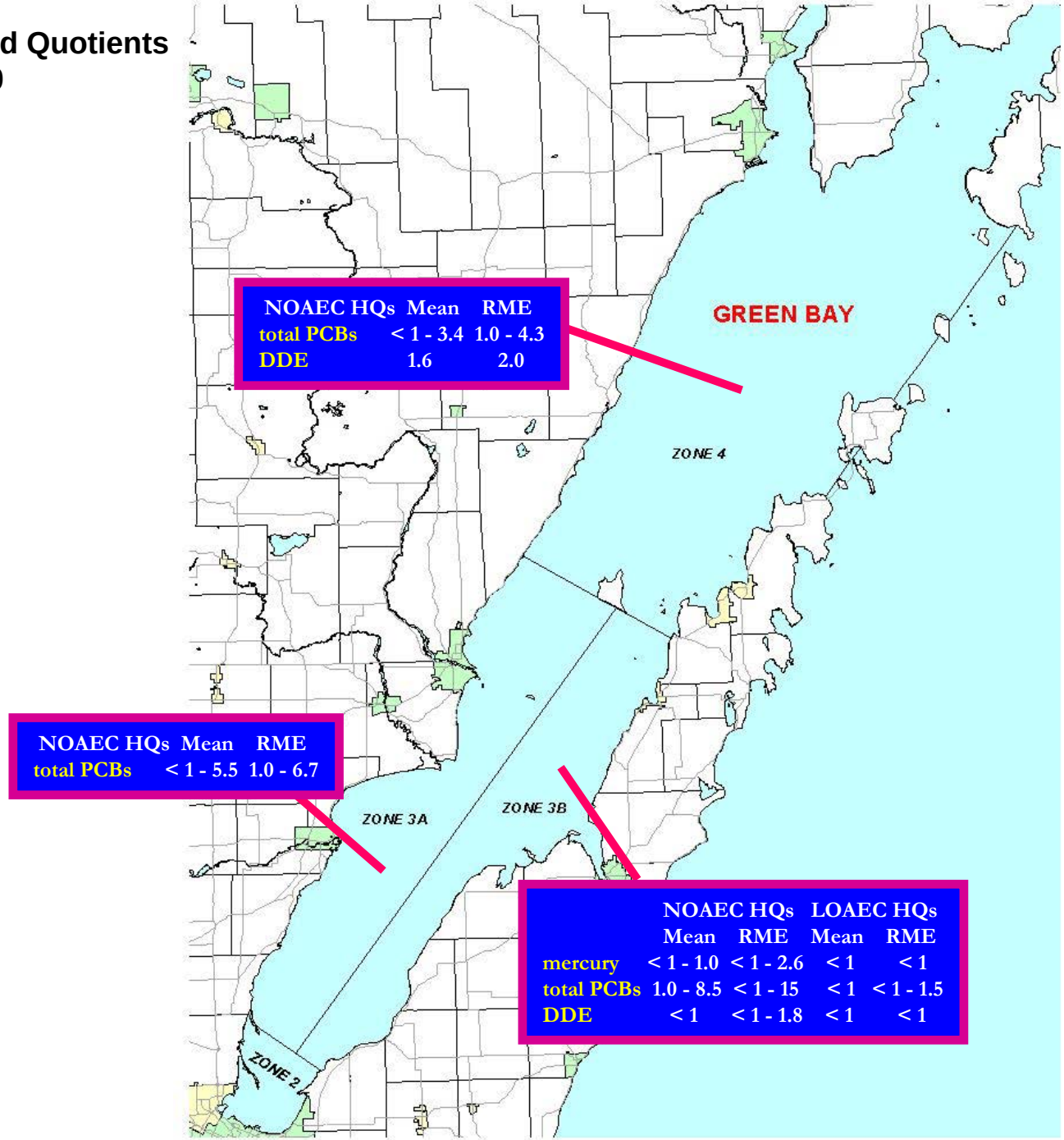

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Figure 6-59 Whole Fish Mercury Hazard Quotients that Exceeded 1.0

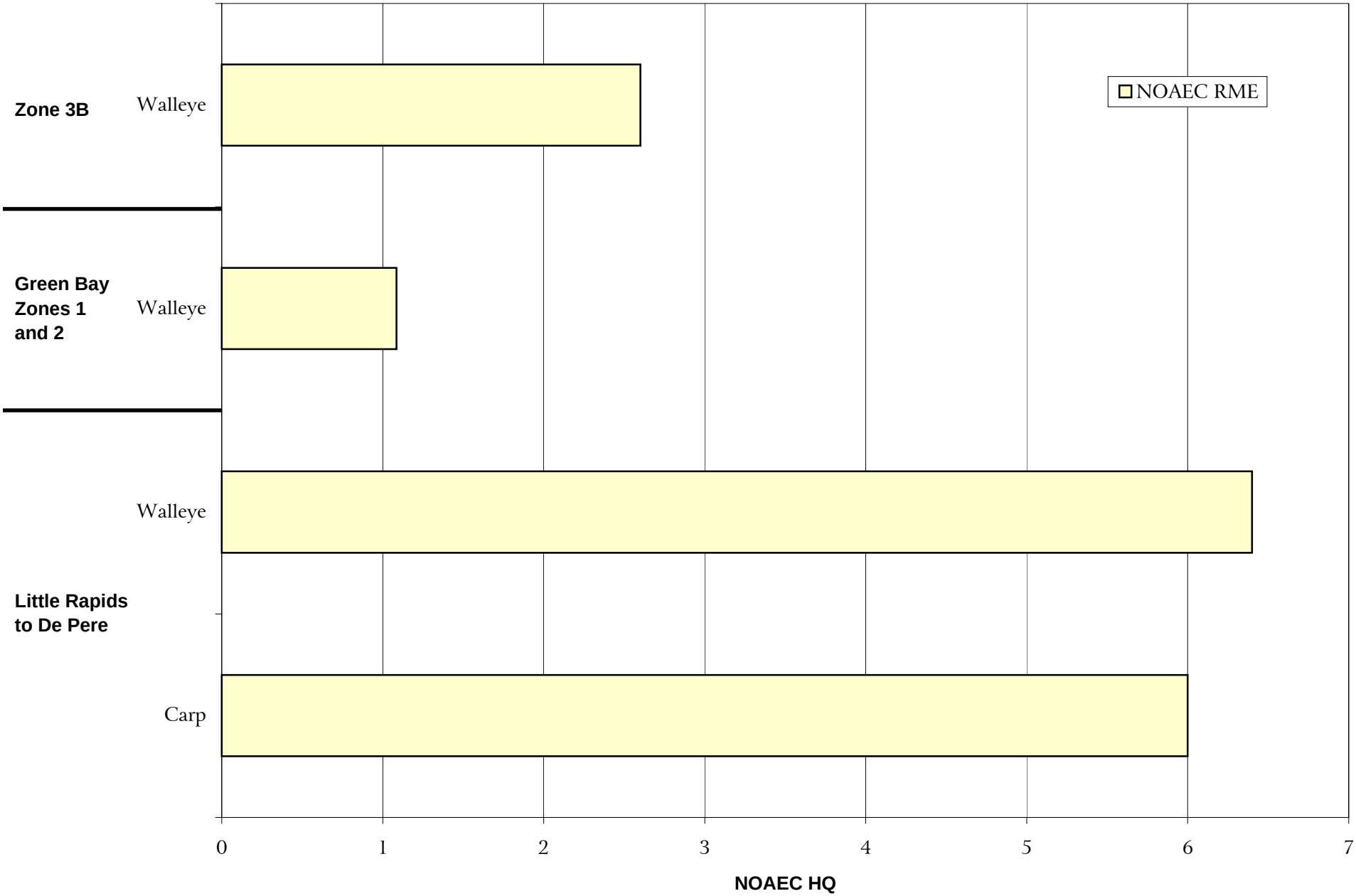


Figure 6-60 Whole Fish Total PCB Hazard Quotients that Exceeded 1.0

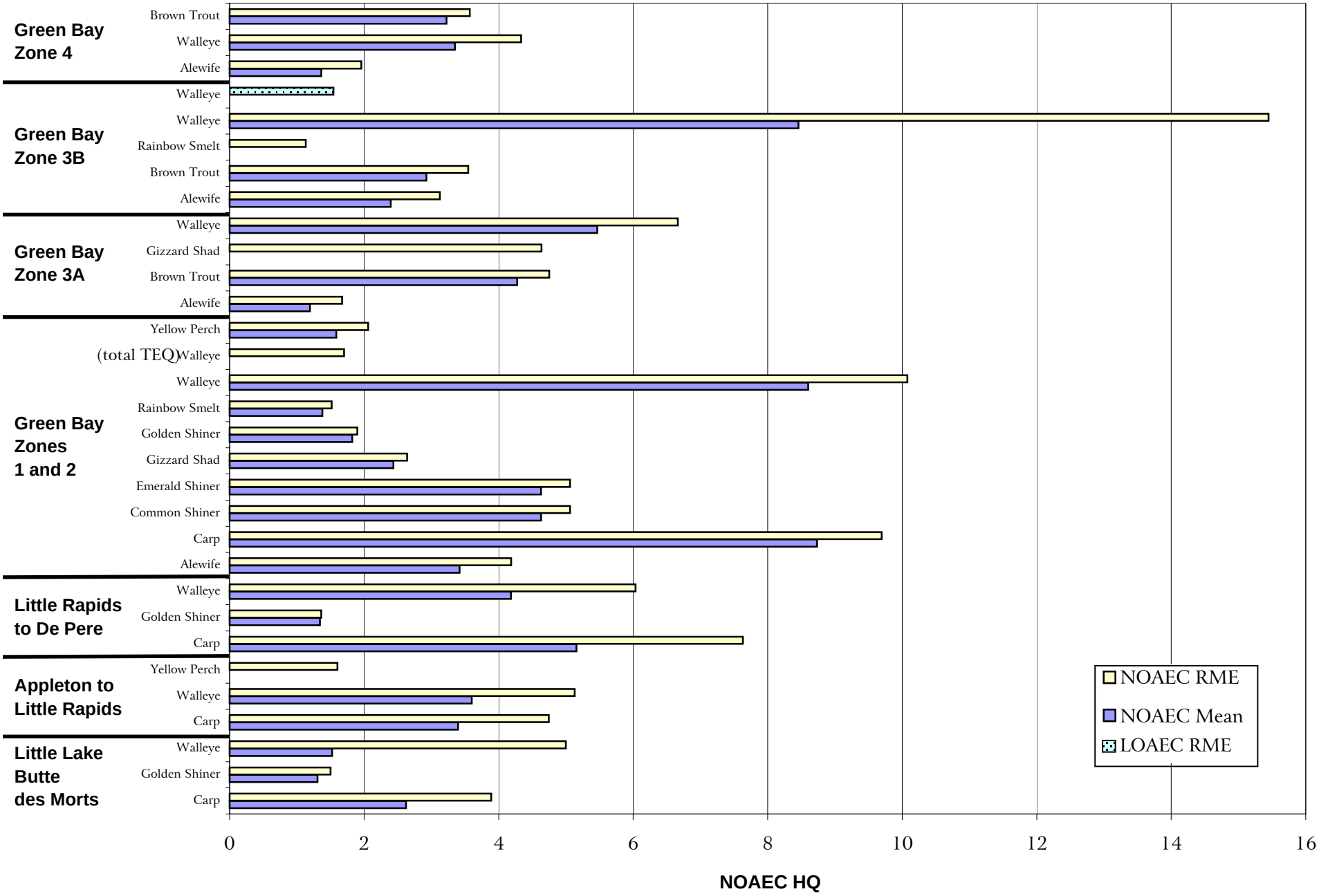


Figure 6-61 Whole Fish Total PCB Hazard Quotients that Exceeded 1.0

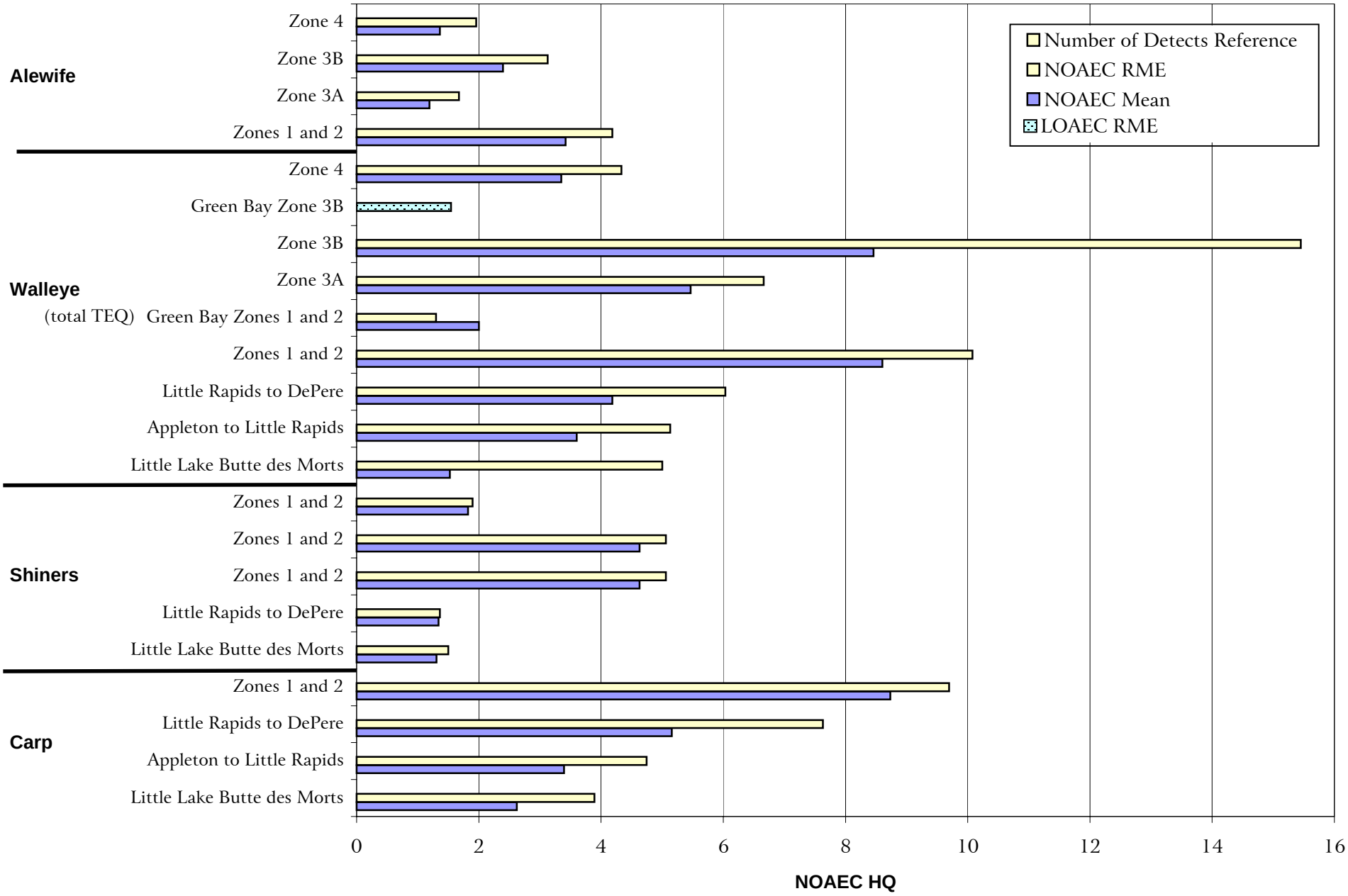


Figure 6-62 Whole Fish DDE Hazard Quotients that Exceeded 1.0

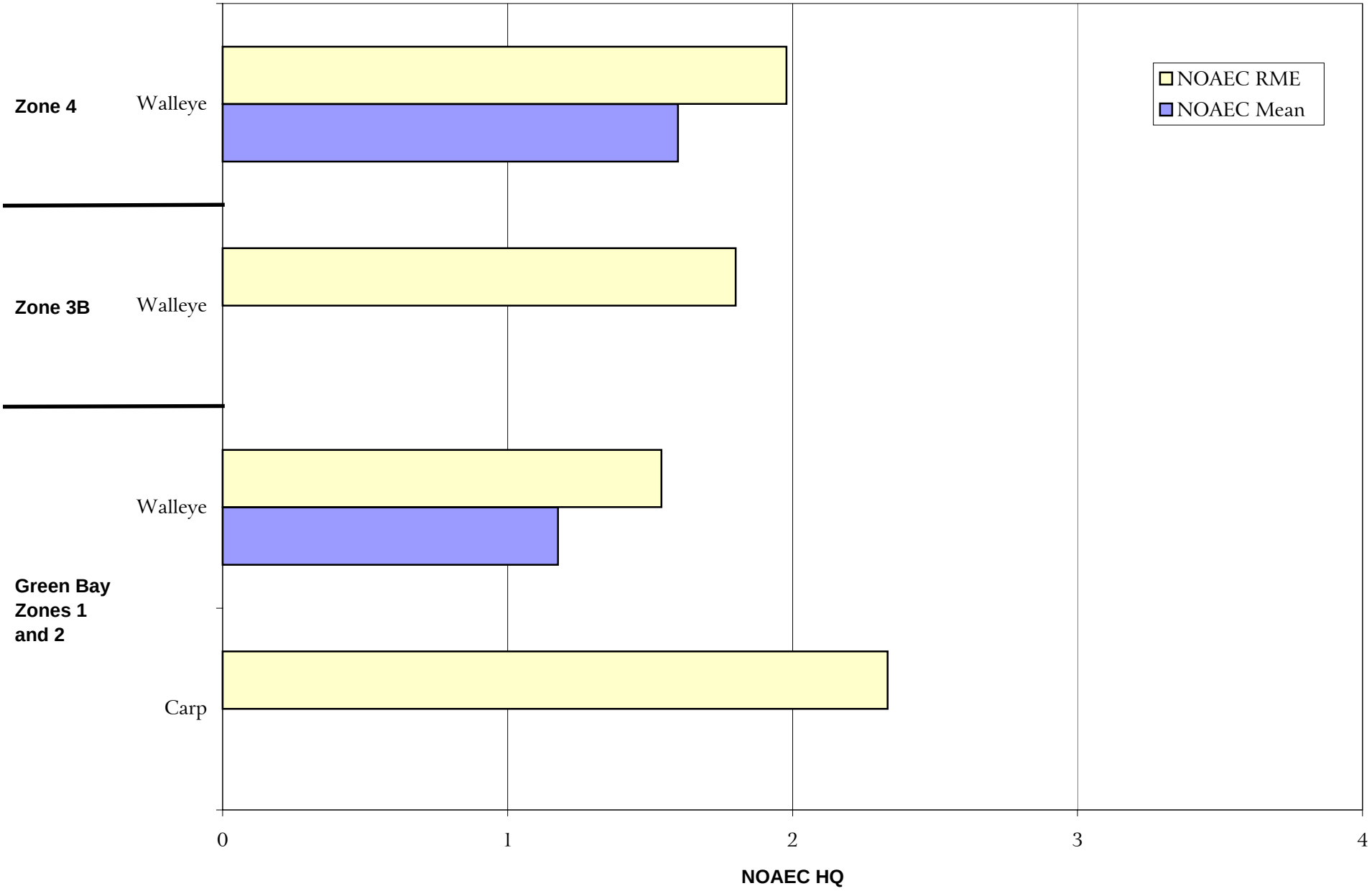
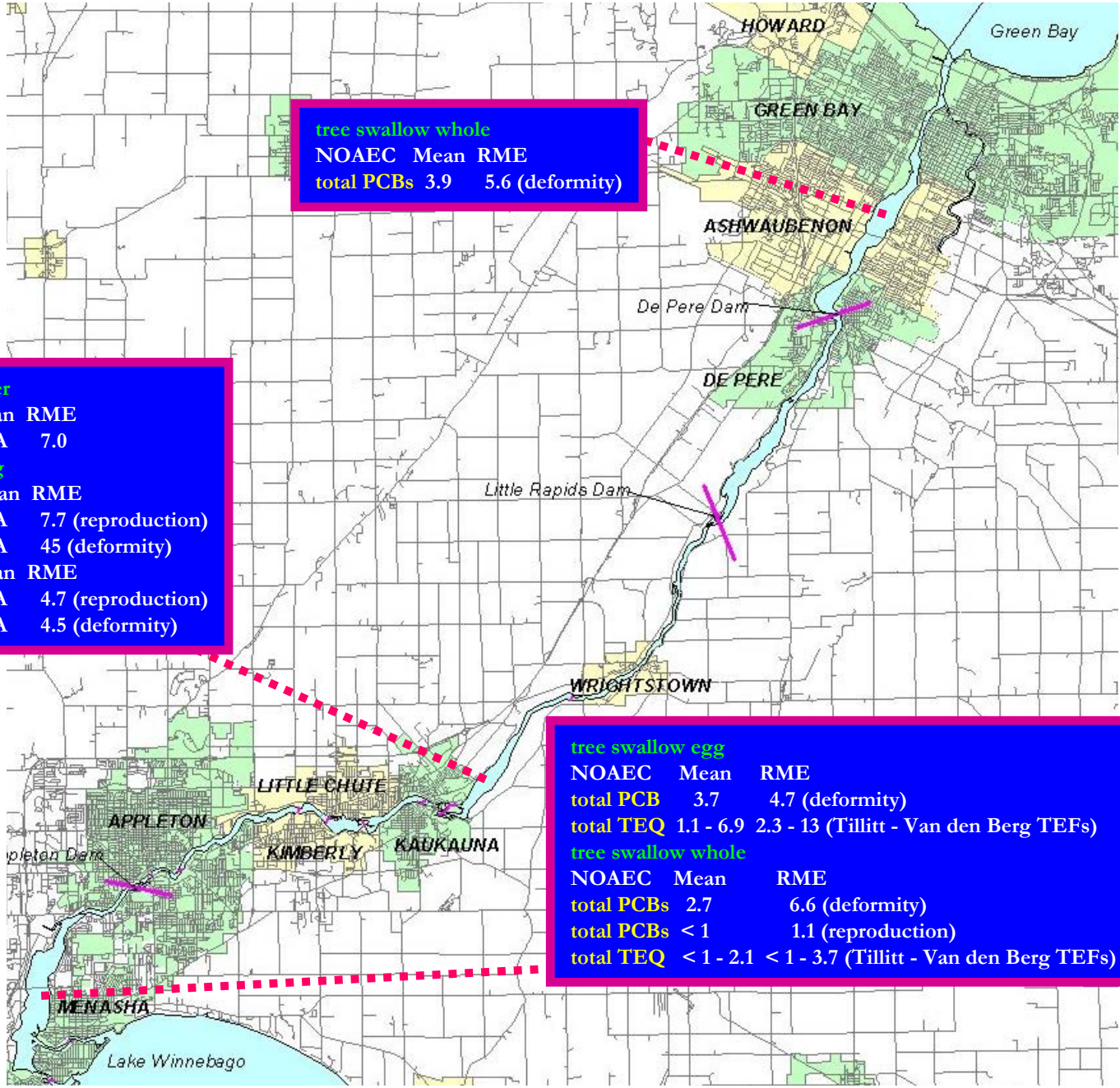


Figure 6-63 Bird Hazard Quotients that Exceeded 1.0

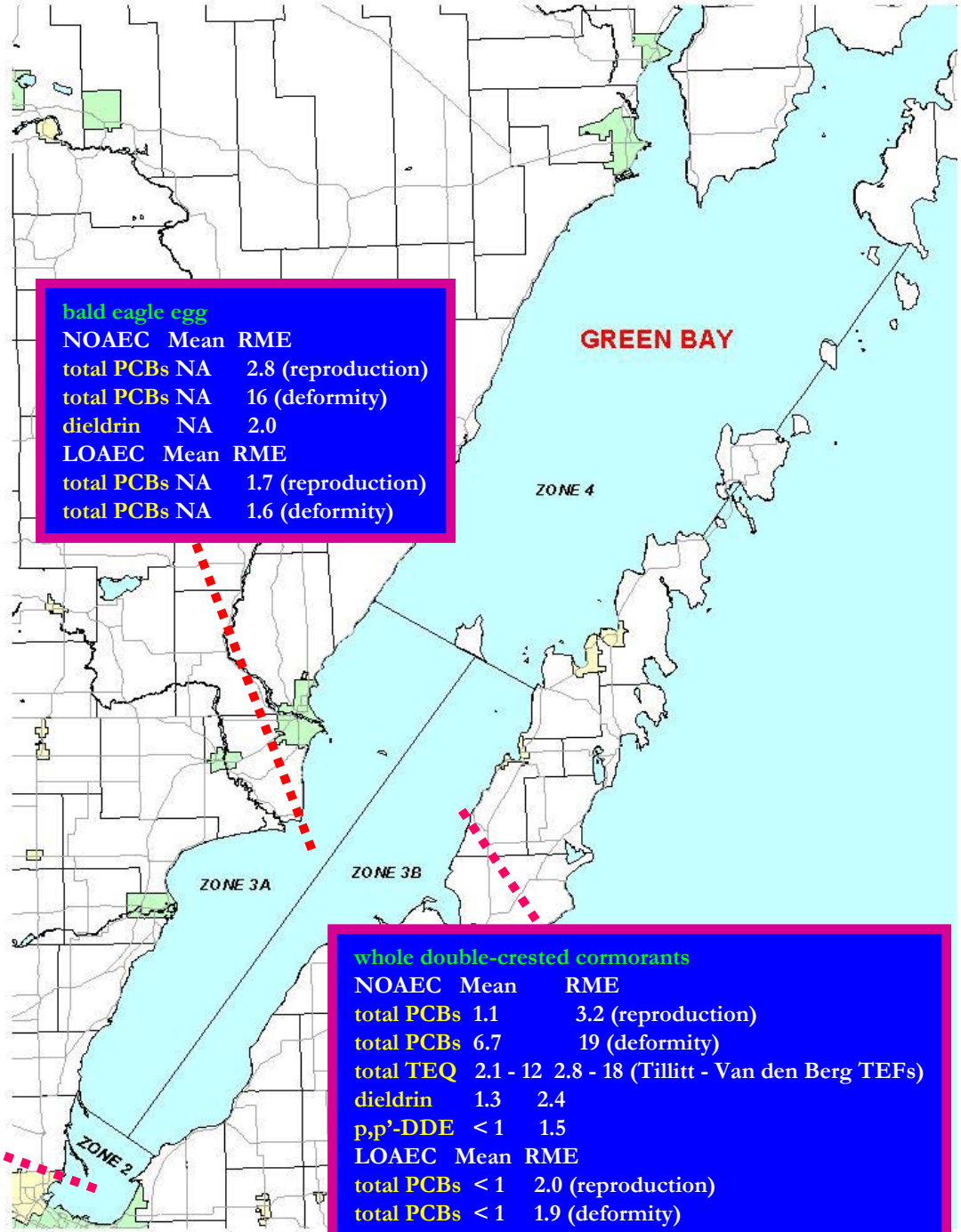


tree swallow whole
NOAEC Mean RME
total PCBs 3.9 5.6 (deformity)

bald eagle liver
NOAEC Mean RME
mercury NA 7.0
bald eagle egg
NOAEC Mean RME
total PCBs NA 7.7 (reproduction)
total PCBs NA 45 (deformity)
LOAEC Mean RME
total PCBs NA 4.7 (reproduction)
total PCBs NA 4.5 (deformity)

tree swallow egg
NOAEC Mean RME
total PCB 3.7 4.7 (deformity)
total TEQ 1.1 - 6.9 2.3 - 13 (Tillitt - Van den Berg TEFs)
tree swallow whole
NOAEC Mean RME
total PCBs 2.7 6.6 (deformity)
total PCBs < 1 1.1 (reproduction)
total TEQ < 1 - 2.1 < 1 - 3.7 (Tillitt - Van den Berg TEFs)

Figure 6-64 Bird Hazard Quotients that Exceeded 1.0



double-crested cormorant eggs

	NOAEC	Mean	RME
total PCBs		3.0	4.5 (reproduction)
total PCBs		17	26 (deformity)
total TEQ	3.8 - 31	5.1 -	46 (Tillitt - Van den Berg TEFs)
dieldrin		2.2	4.4
p,p'-DDE		1.4	2.4

	LOAEC	Mean	RME
total PCBs		1.8	2.8 (reproduction)
total PCBs		1.7	2.6 (deformity)
total TEQ	< 1 - 1.1	< 1 - 1.7	(Tillitt - Van den Berg TEFs)
p,p'-DDE		< 1	1.4

whole double-crested cormorants

	NOAEC	Mean	RME
total PCBs		2.3	3.0 (reproduction)
total PCBs		14	17 (deformity)
total TEQ	3.0 - 35	6.4 - 61	(Tillitt - Van den Berg TEFs)
dieldrin		2.0	2.4
p,p'-DDE		< 1	1.2

	LOAEC	Mean	RME
total PCBs		1.5	1.8 (reproduction)
total PCBs		1.4	1.7 (deformity)
total TEQ	< 1 - 1.3	< 1 - 2.2	(Tillitt - Van den Berg TEFs)

common tern eggs

	NOAEC	Mean	RME
total PCBs		1.0	1.3 (reproduction)
total PCBs		6.0	7.5 (deformity)
total TEQ	< 1 - 44	7.5 - 110	(Tillitt - Van den Berg TEFs)
dieldrin		< 1	1.4

	LOAEC	Mean	RME
total TEQ	< 1 - 1.6	< 1 - 4.0	

Forster's tern eggs

	NOAEC	Mean	RME
total PCBs		1.1	1.3 (reproduction)
total PCBs		6.3	7.8 (deformity)
total TEQ	< 1 - 21	3.3 - 55	(Tillitt - Van den Berg TEFs)

	LOAEC	Mean	RME
total TEQ	< 1	< 1 - 2.0	(Tillitt - Van den Berg TEFs)

bald eagle egg

	NOAEC	Mean	RME
total PCBs	NA		2.8 (reproduction)
total PCBs	NA		16 (deformity)
dieldrin	NA		2.0

	LOAEC	Mean	RME
total PCBs	NA		1.7 (reproduction)
total PCBs	NA		1.6 (deformity)

whole double-crested cormorants

	NOAEC	Mean	RME
total PCBs		1.1	3.2 (reproduction)
total PCBs		6.7	19 (deformity)
total TEQ	2.1 - 12	2.8 - 18	(Tillitt - Van den Berg TEFs)
dieldrin		1.3	2.4
p,p'-DDE		< 1	1.5

	LOAEC	Mean	RME
total PCBs		< 1	2.0 (reproduction)
total PCBs		< 1	1.9 (deformity)

Figure 6-65a Bird Total PCB* and TEQ Hazard Quotients that Exceeded 1.0 (Reproduction Endpoint)

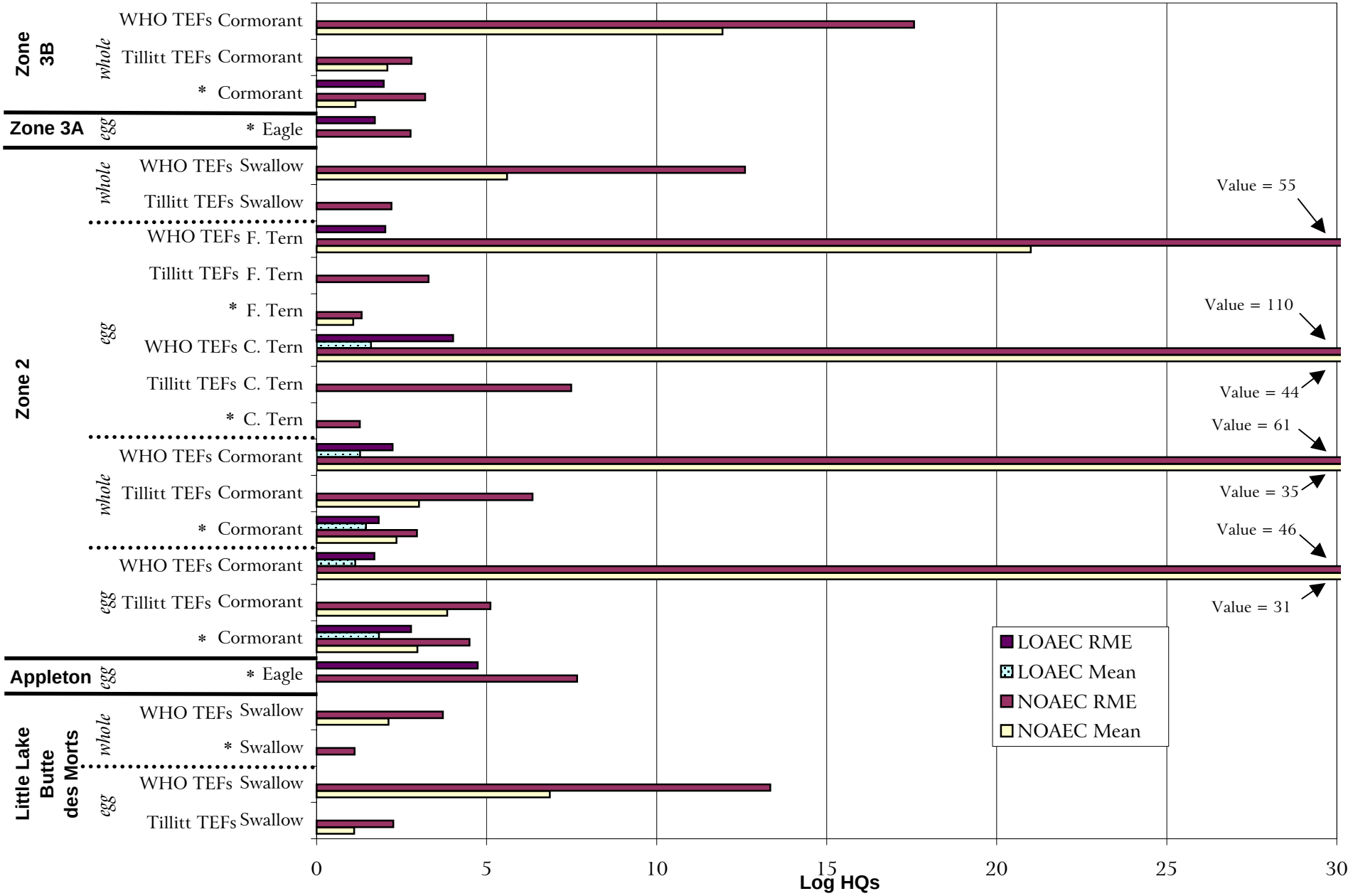


Figure 6-65b Bird Total PCB* and TEQ Hazard Quotients that Exceeded 1.0 (Deformity Endpoint)

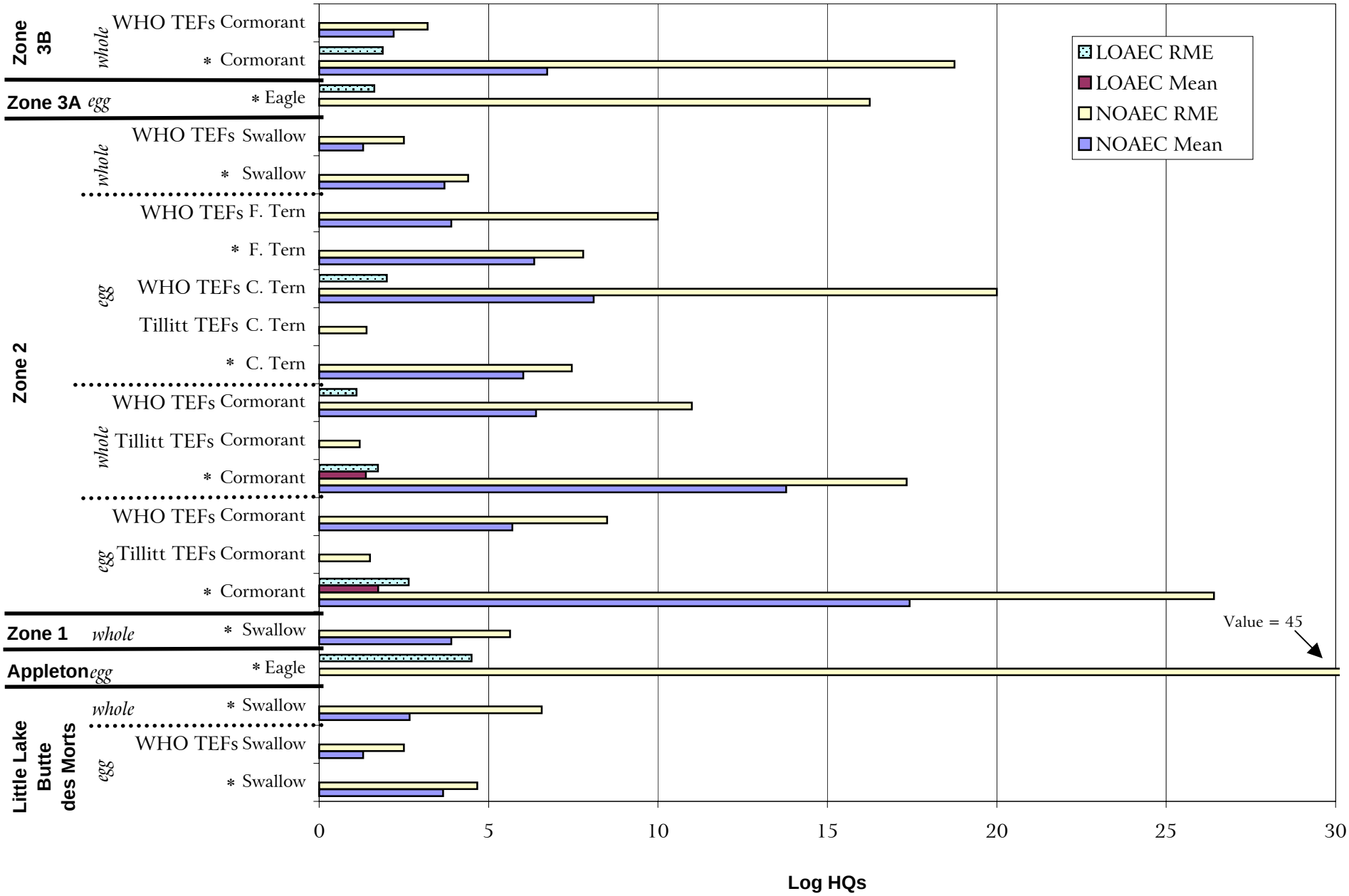


Figure 6-66 Bird Metal and Pesticide Hazard Quotients that Exceeded 1.0

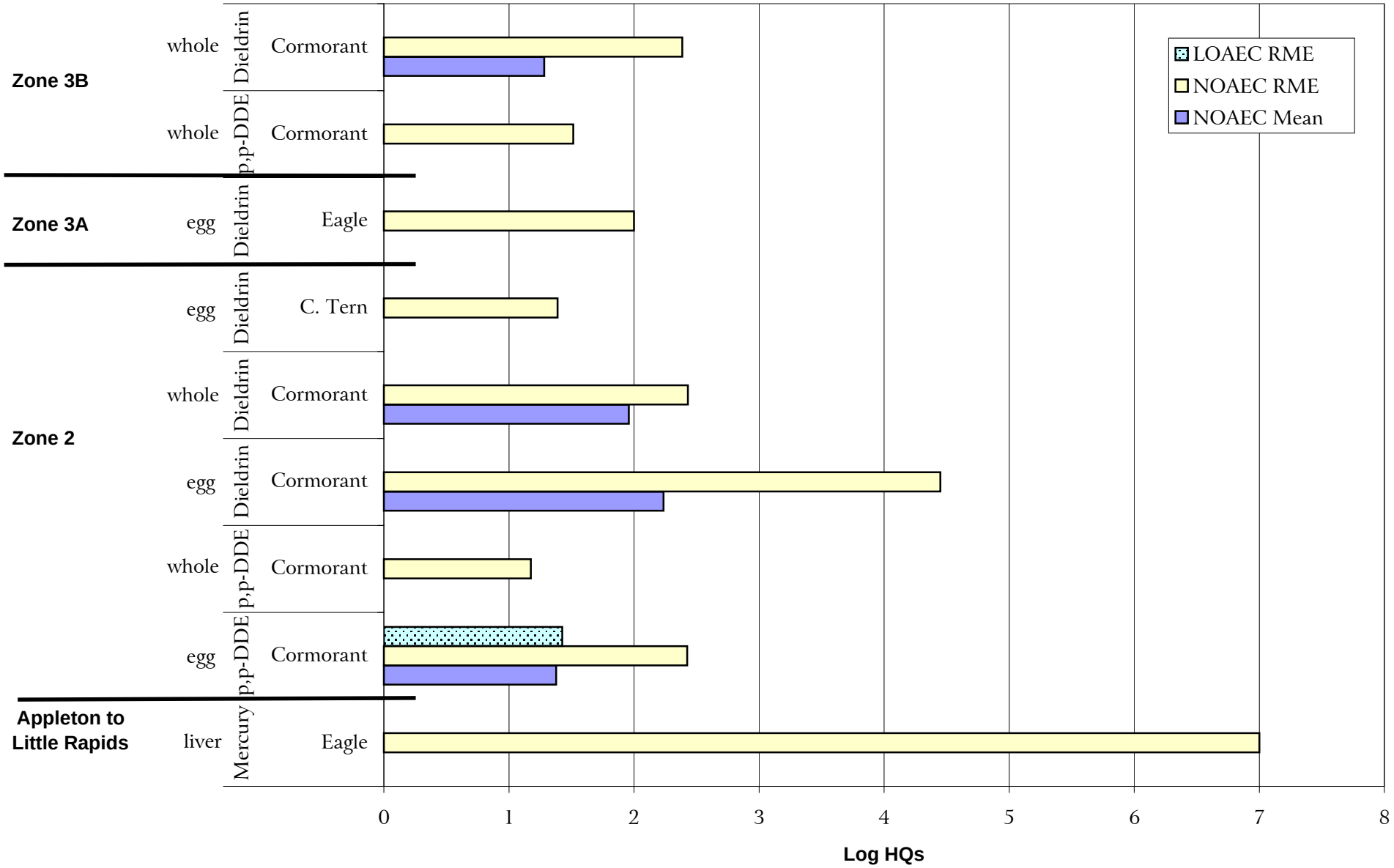


Figure 6-67 Estimated Piscivorous Bird Hazard Quotients that Exceeded 1.0

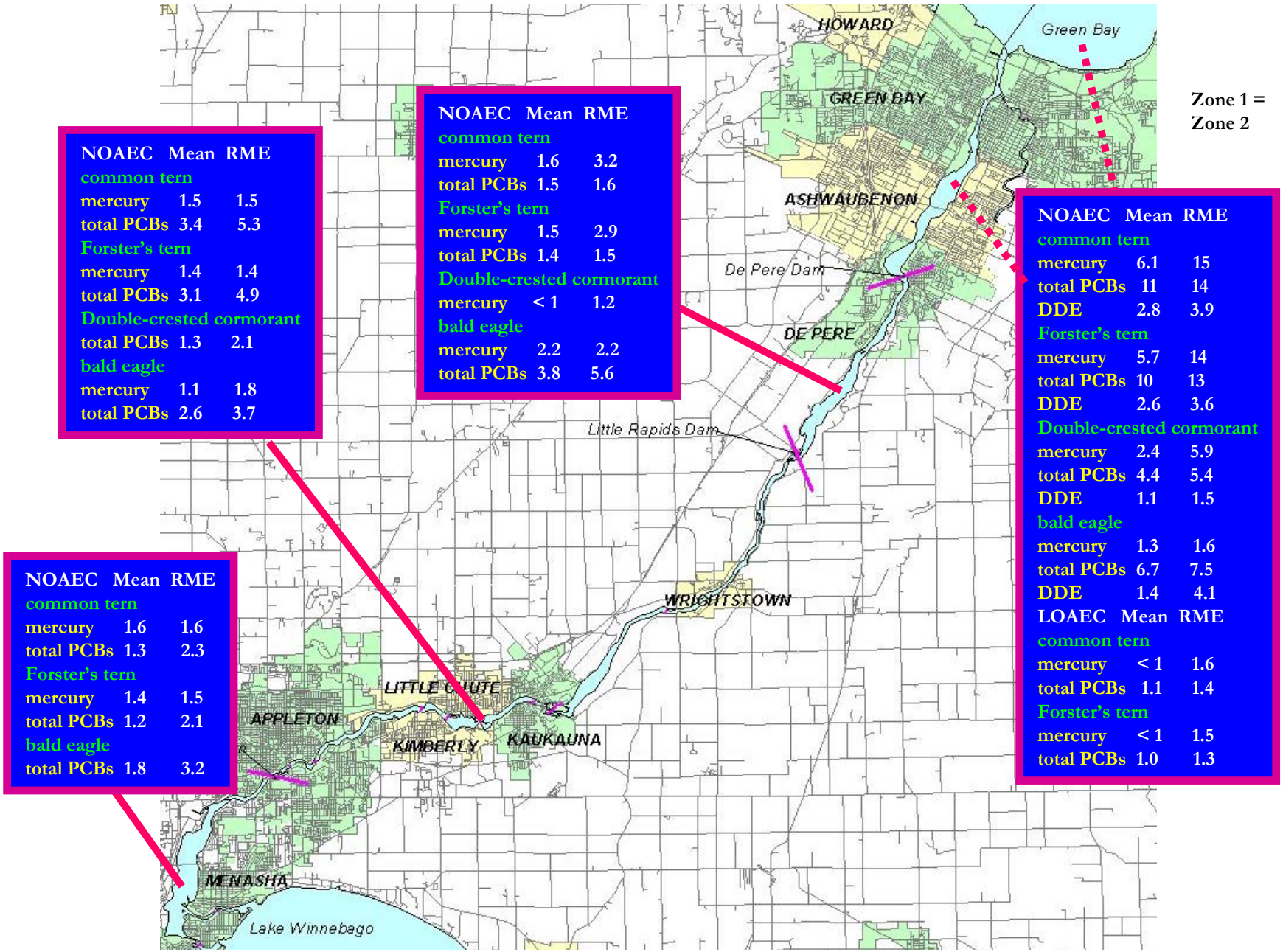


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Figure 6-68 **Estimated Piscivorous Bird Hazard Quotients that Exceeded 1.0**

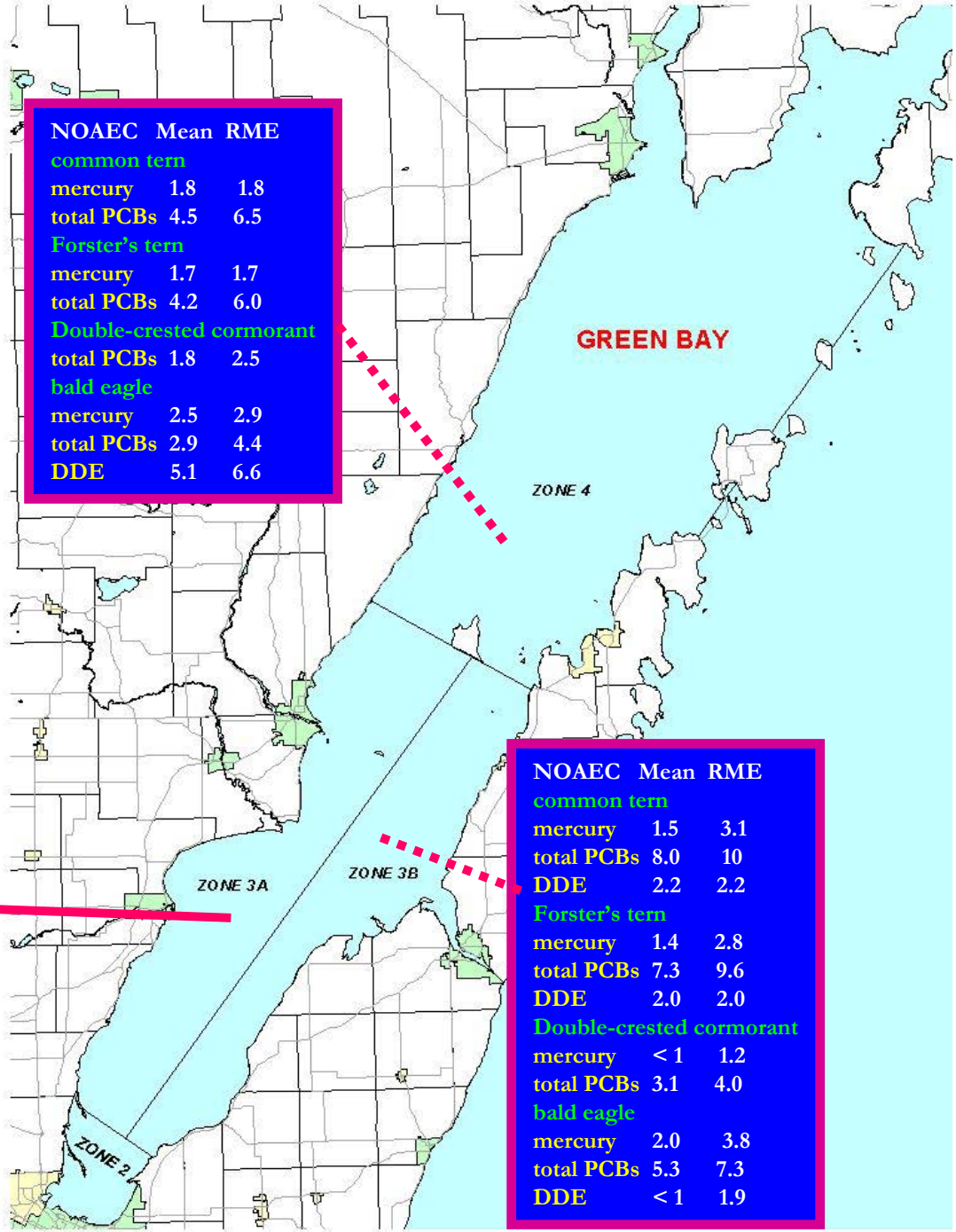

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Figure 6-69 Estimated Piscivorous Bird Mercury Hazard Quotients that Exceeded 1.0

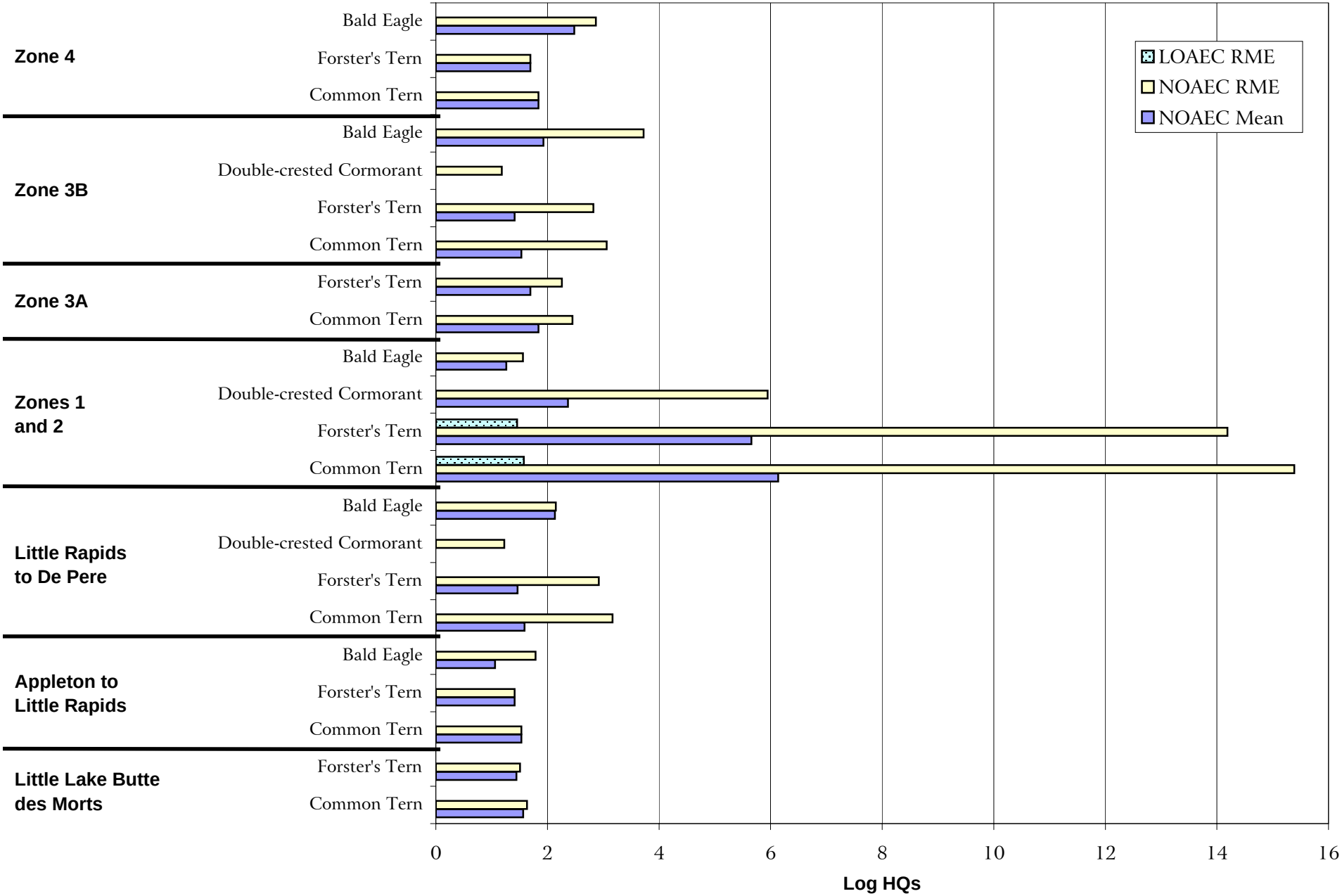


Figure 6-70 Estimated Piscivorous Bird Total PCB Hazard Quotients that Exceeded 1.0

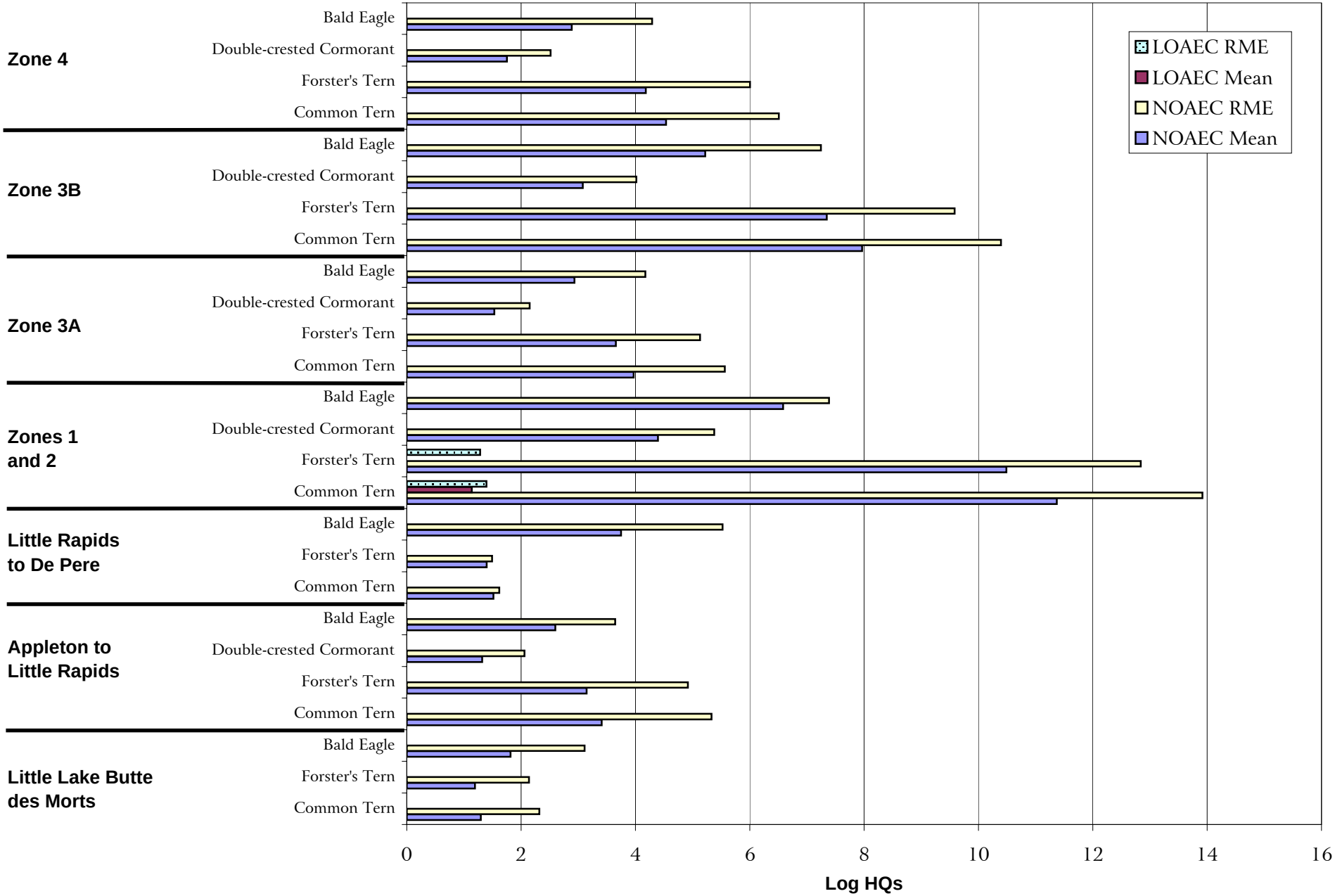
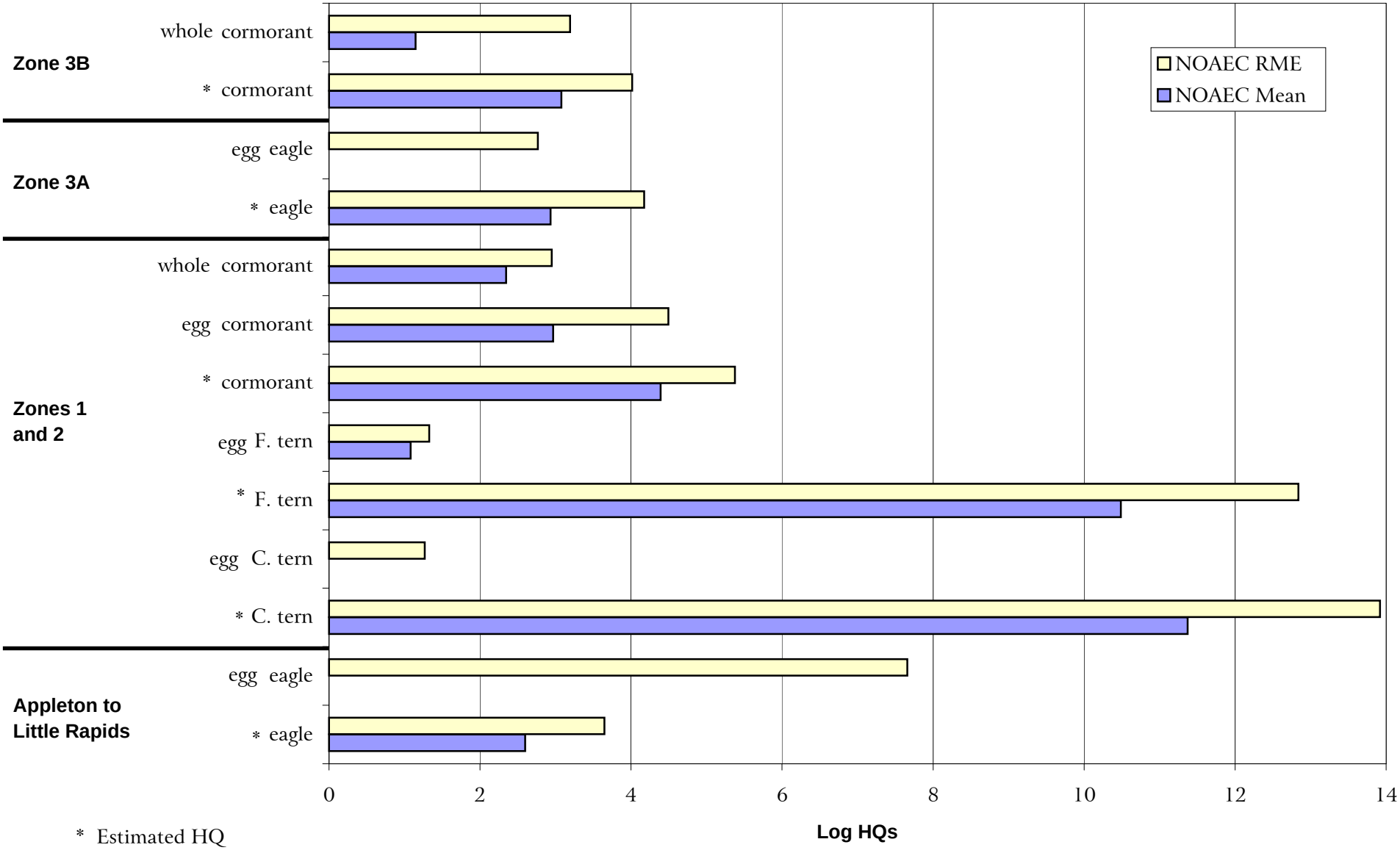


Figure 6-71 Comparison of Measured and Estimated Total PCB Hazard Quotients in Piscivorous Birds



* Estimated HQ

Figure 6-72 Estimated Piscivorous Bird DDE Hazard Quotients that Exceeded 1.0

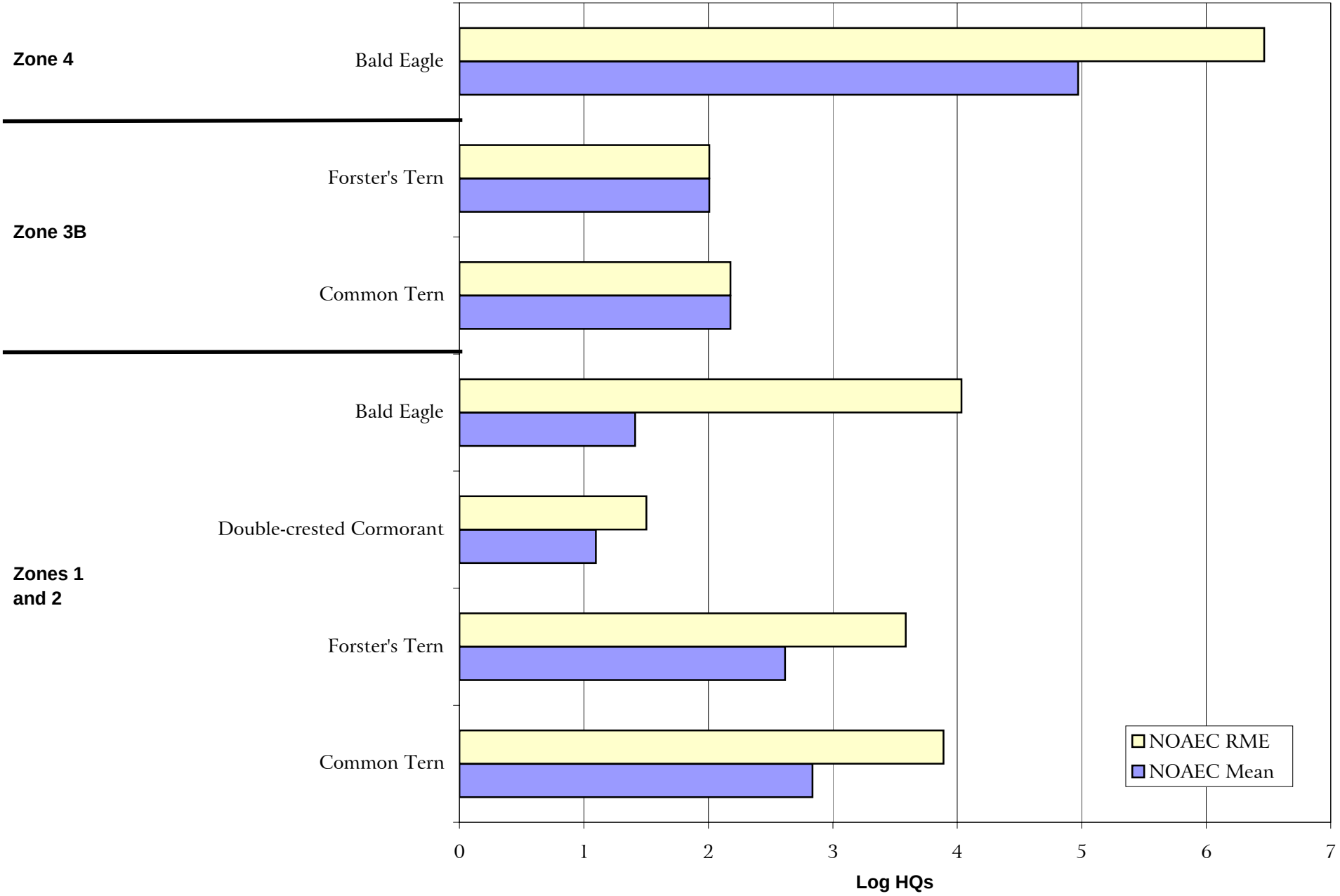


Figure 6-73 Estimated Piscivorous Mammal Hazard Quotients that Exceeded 1.0

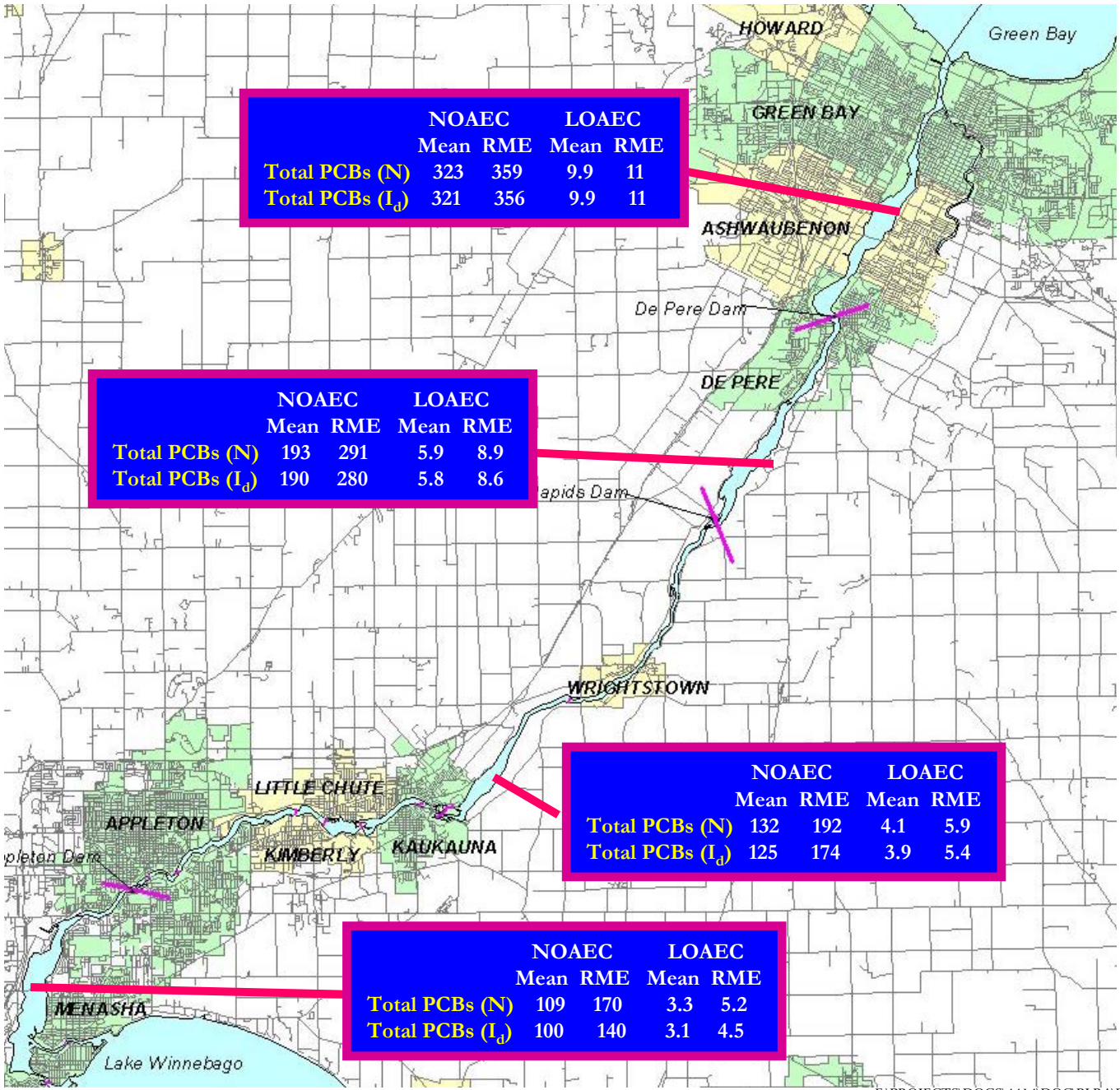


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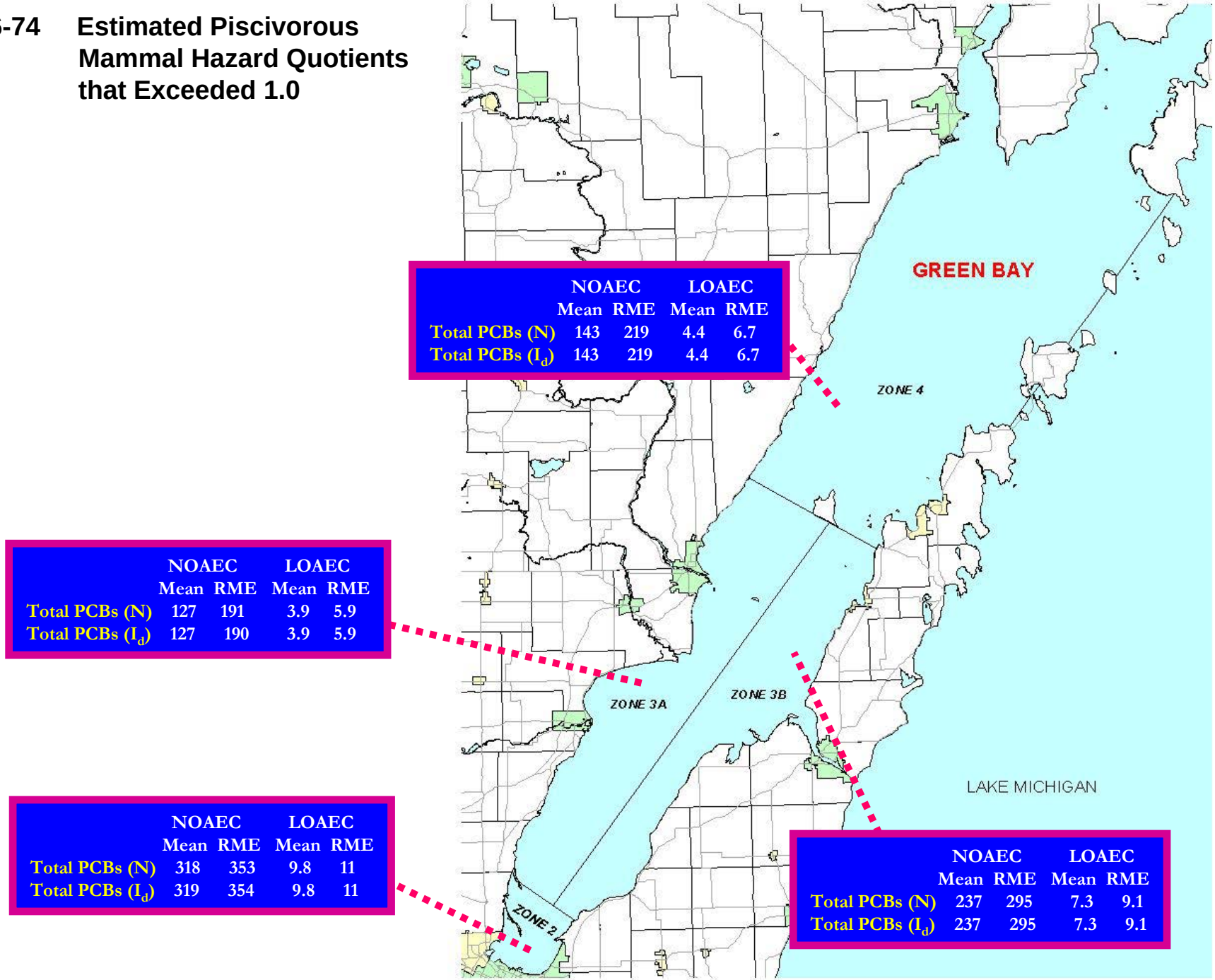
Figure 6-74 **Estimated Piscivorous Mammal Hazard Quotients that Exceeded 1.0**



	NOAEC		LOAEC	
	Mean	RME	Mean	RME
Total PCBs (N)	143	219	4.4	6.7
Total PCBs (I_d)	143	219	4.4	6.7

	NOAEC		LOAEC	
	Mean	RME	Mean	RME
Total PCBs (N)	127	191	3.9	5.9
Total PCBs (I_d)	127	190	3.9	5.9

	NOAEC		LOAEC	
	Mean	RME	Mean	RME
Total PCBs (N)	318	353	9.8	11
Total PCBs (I_d)	319	354	9.8	11

	NOAEC		LOAEC	
	Mean	RME	Mean	RME
Total PCBs (N)	237	295	7.3	9.1
Total PCBs (I_d)	237	295	7.3	9.1

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Figure 6-75 Estimated Hazard Quotients Exceeding 1.0 in Piscivorous Mammals

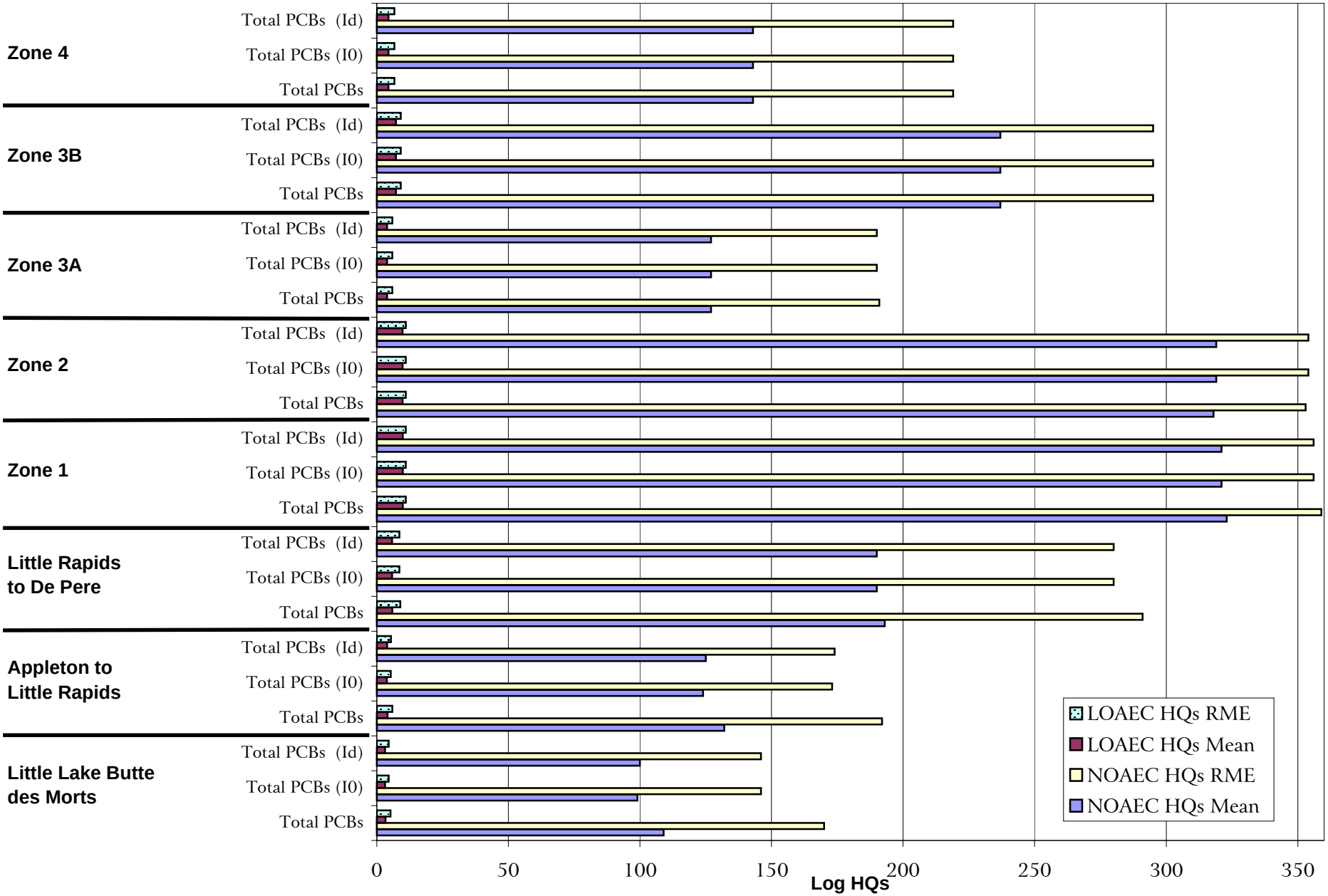


Figure 6-76 Little Lake Butte des Morts Reach Hazard Quotients that Exceed 1.0

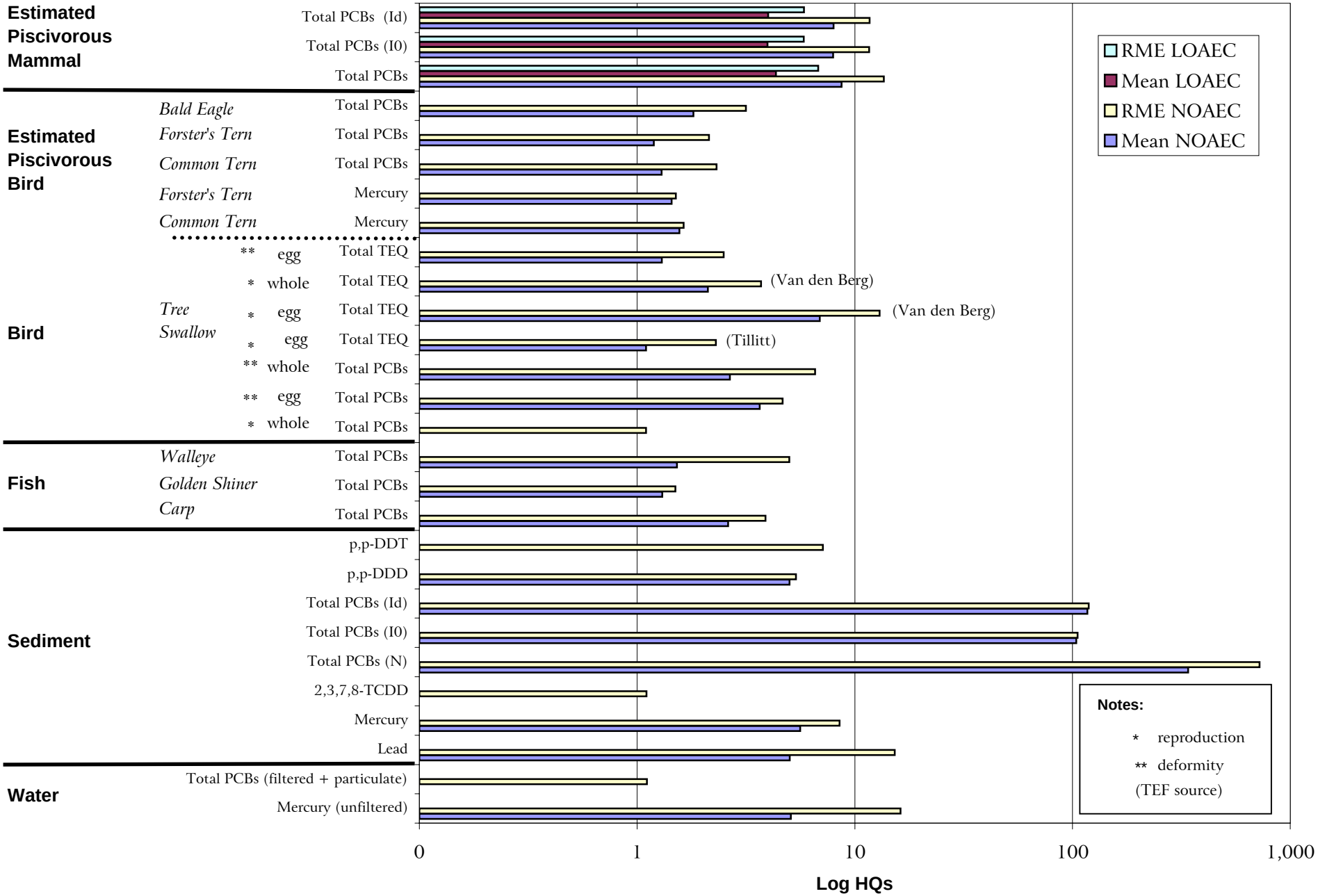


Figure 6-77 Appleton to Little Rapids Reach Hazard Quotients that Exceed 1.0

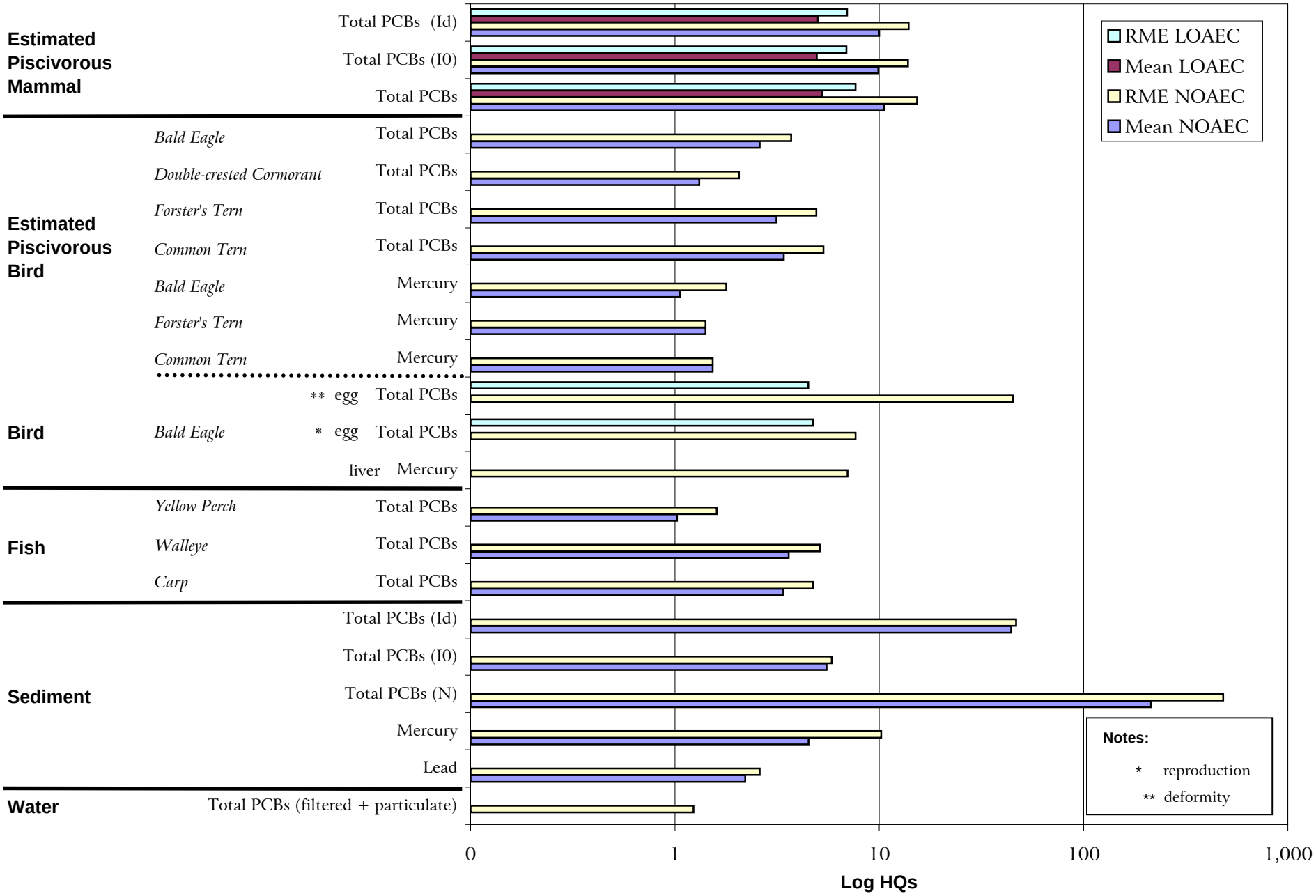


Figure 6-78 Little Rapids to De Pere Reach Hazard Quotients that Exceed 1.0

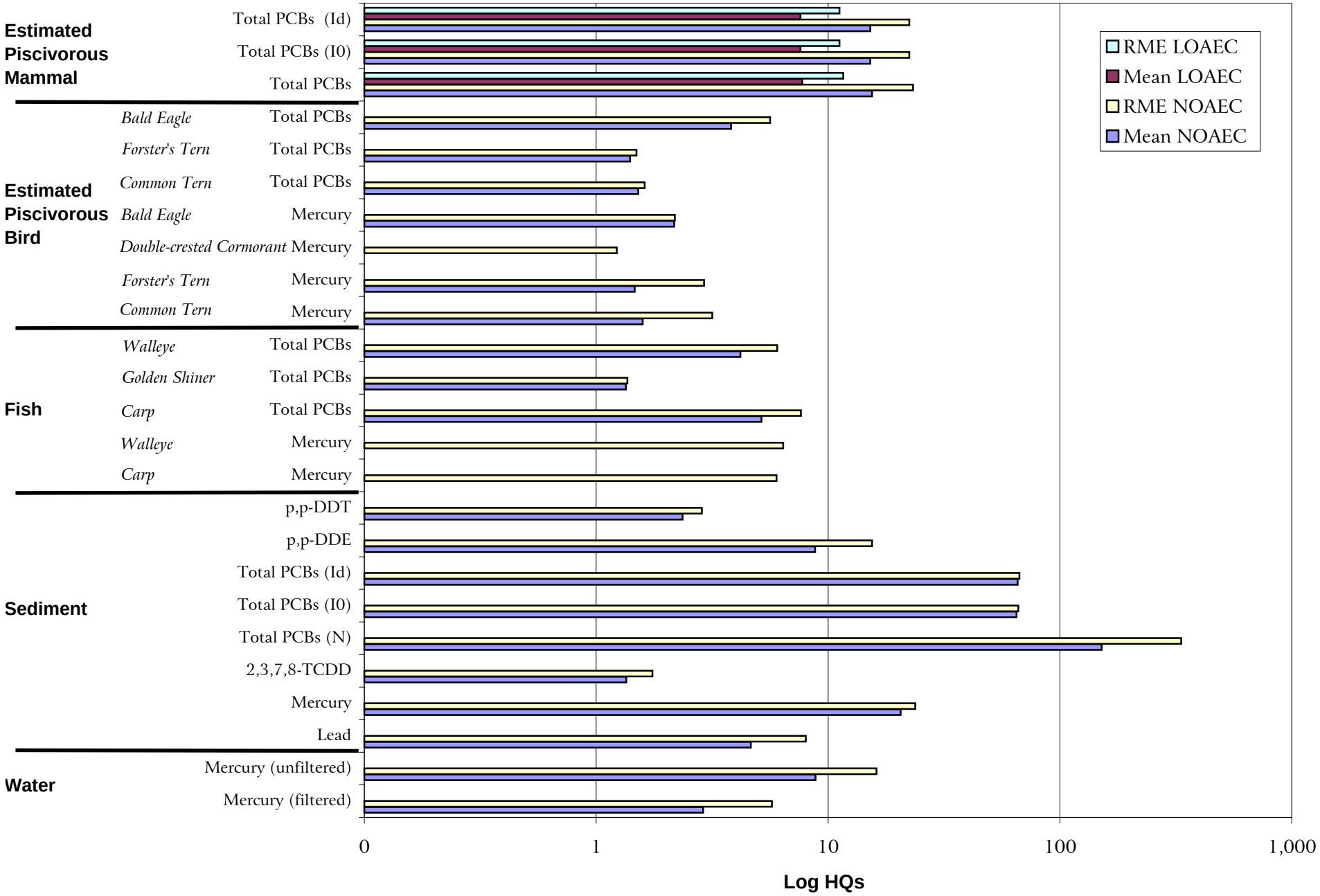


Figure 6-79 Green Bay Zone 1 Hazard Quotients that Exceed 1.0

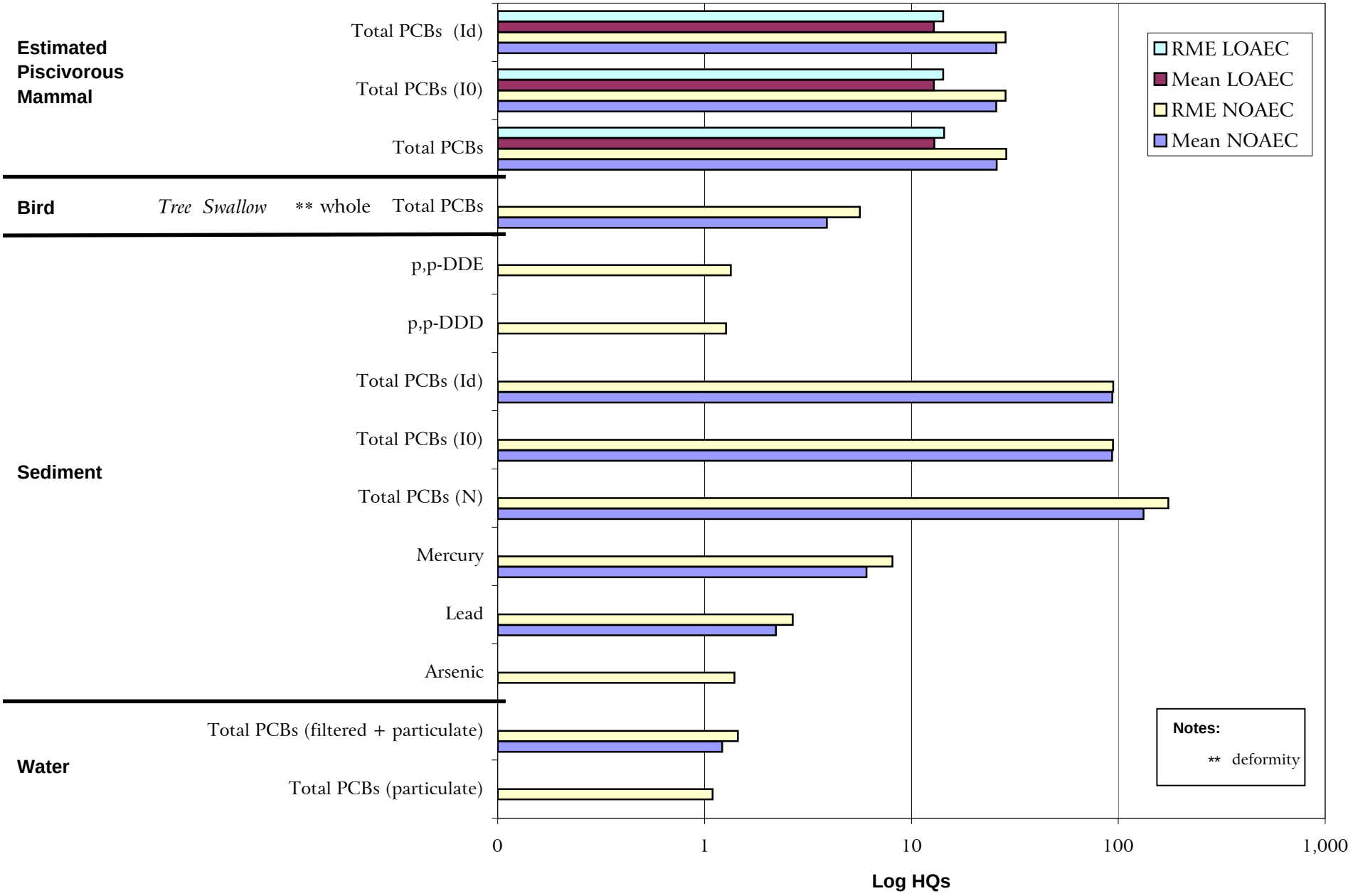


Figure 6-80a Green Bay Zone 2 Hazard Quotients that Exceed 1.0 (Part 1)

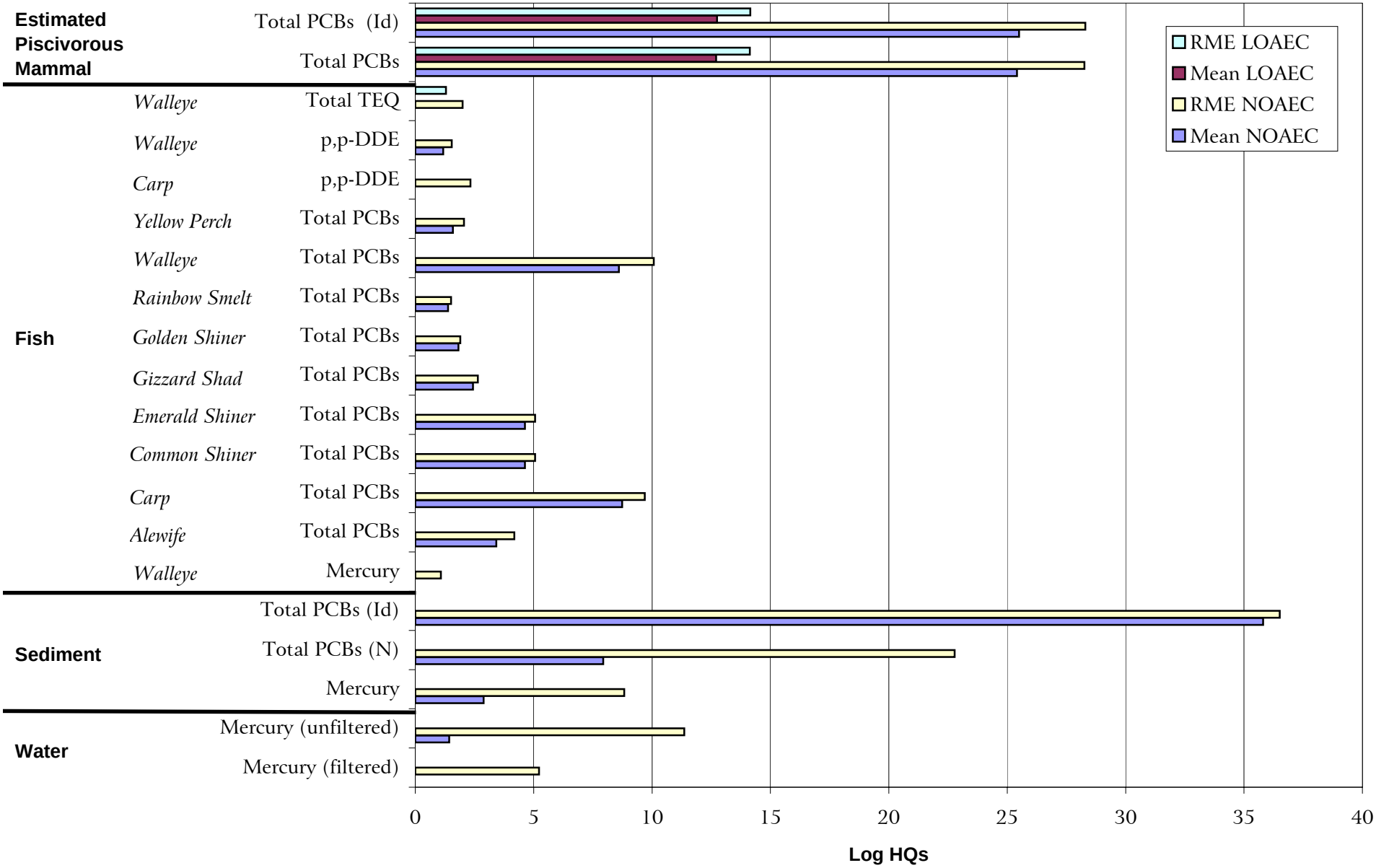


Figure 6-80b Green Bay Zone 2 Hazard Quotients that Exceed 1.0 (Part 2)

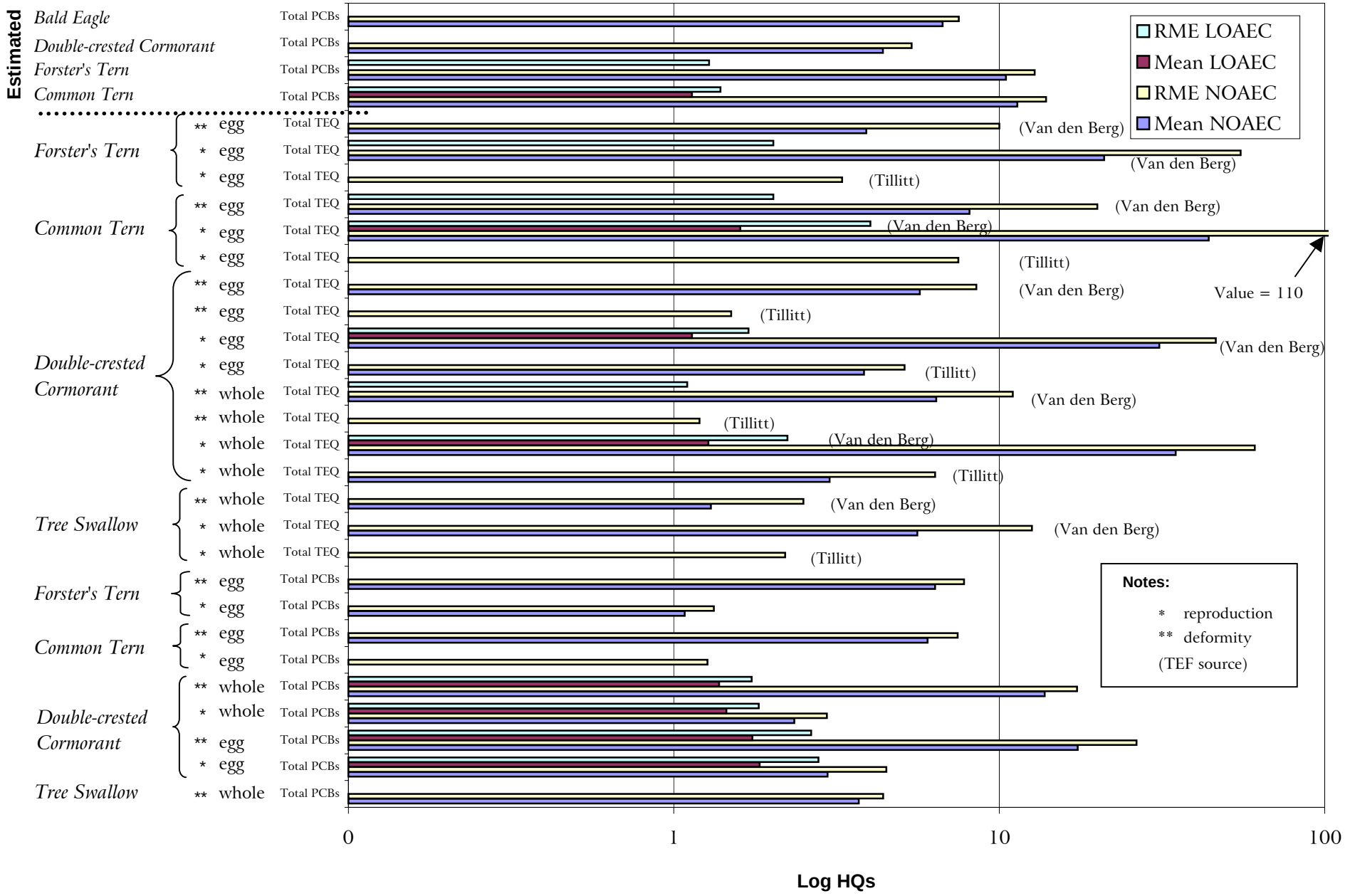


Figure 6-80c Green Bay Zone 2 Hazard Quotients that Exceed 1.0 (Part 3)

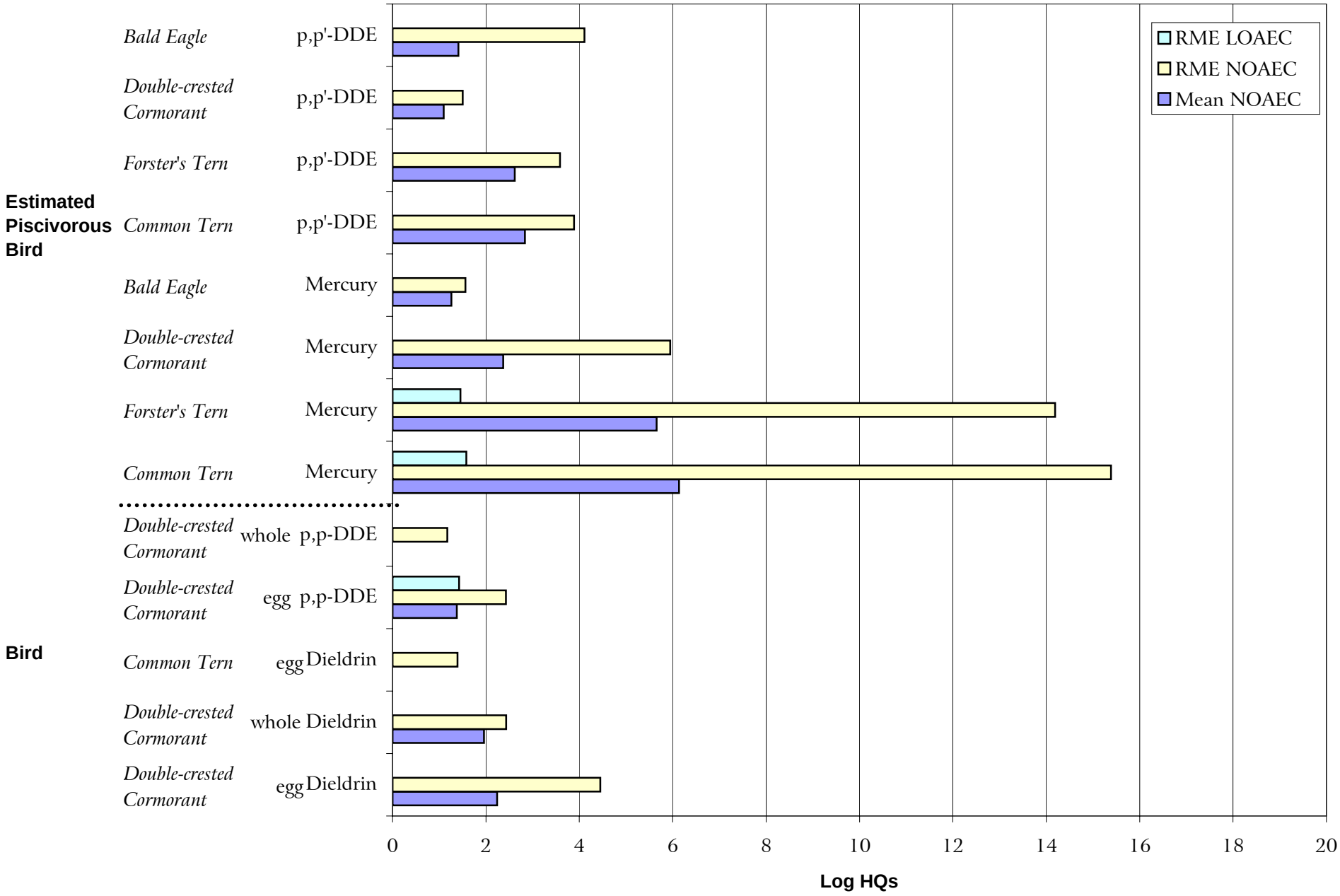


Figure 6-81 Green Bay Zone 3A Hazard Quotients that Exceed 1.0

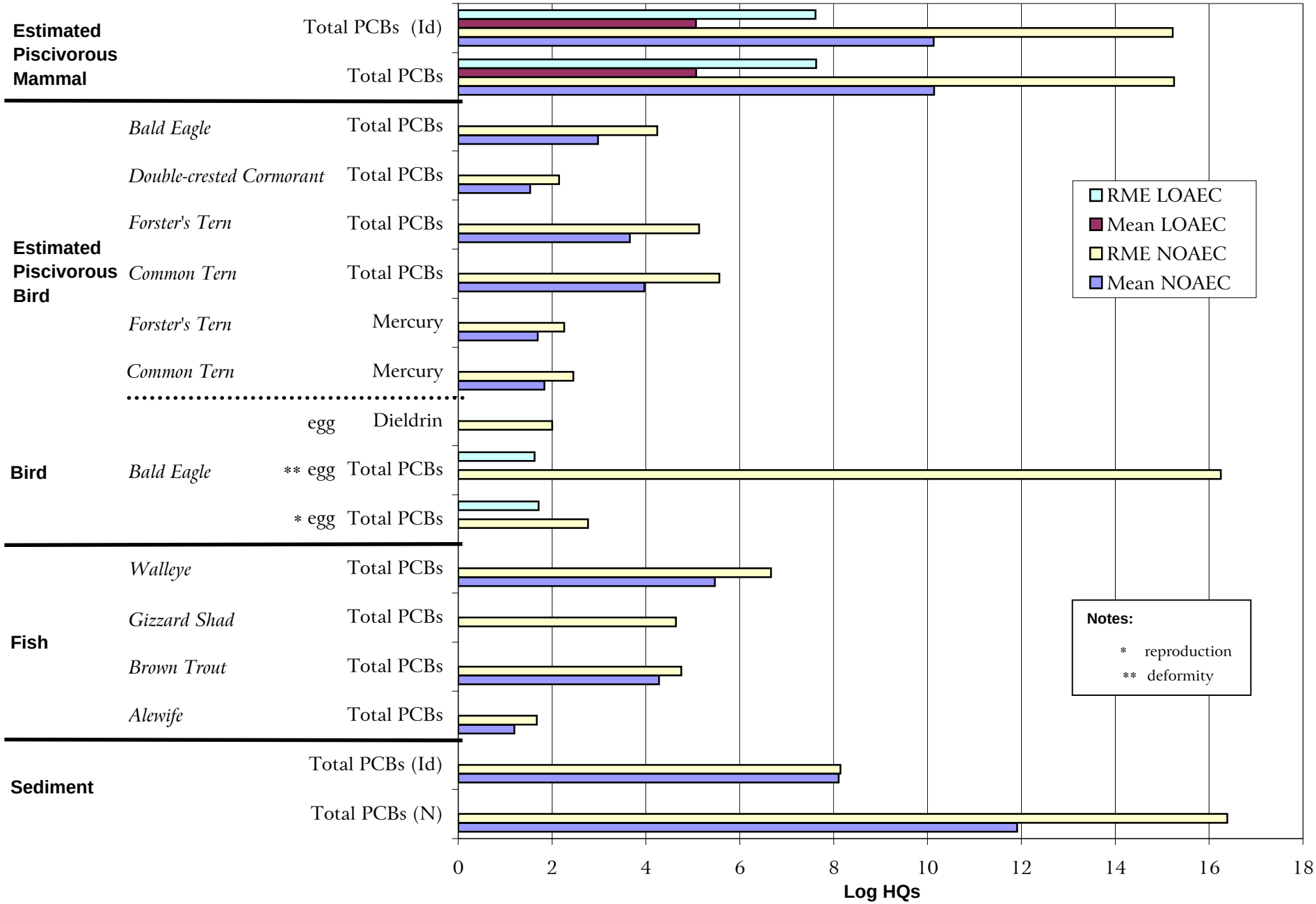


Figure 6-82 Green Bay Zone 3B Hazard Quotients that Exceed 1.0

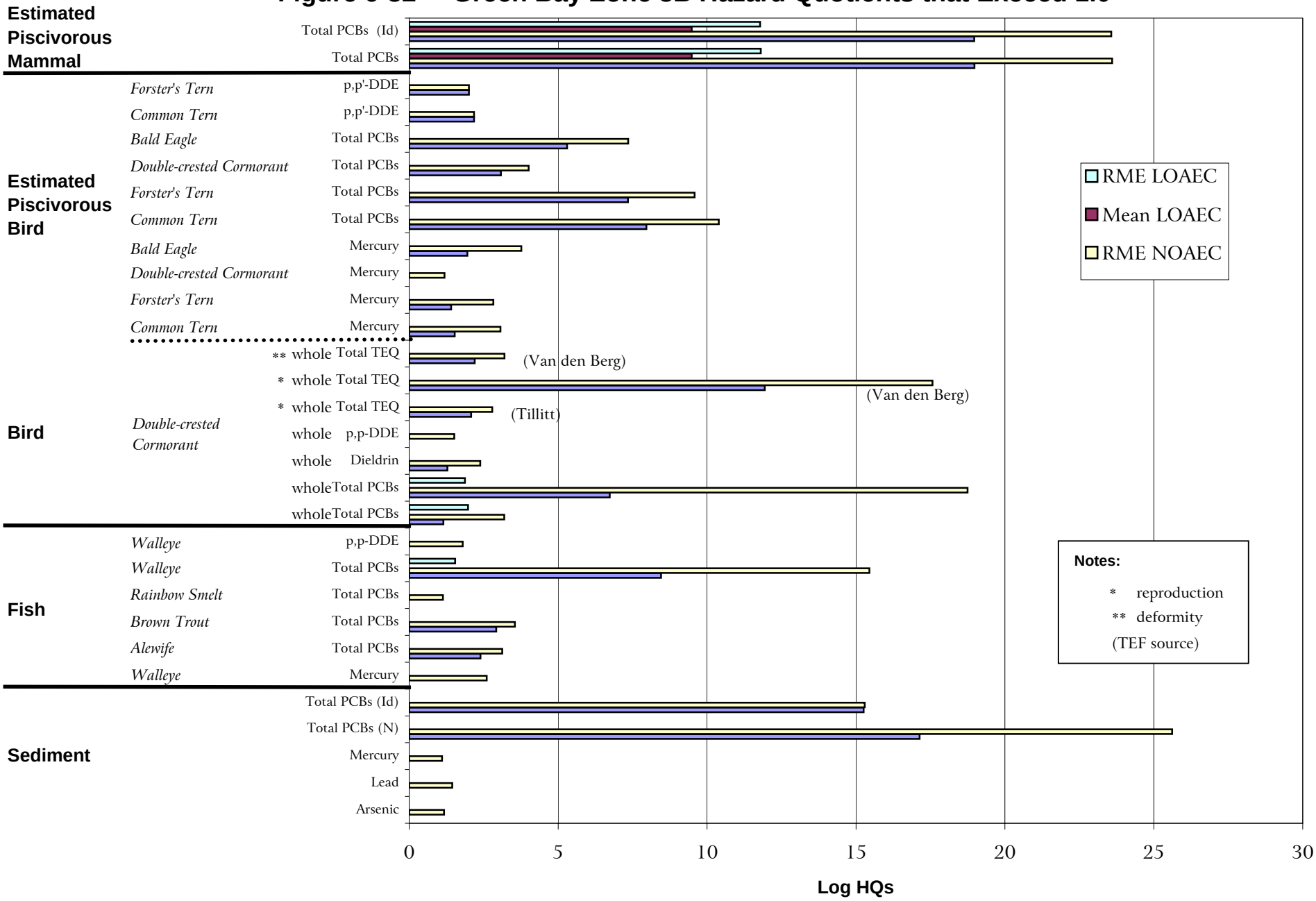


Figure 6-83 Green Bay Zone 4 Hazard Quotients that Exceed 1.0

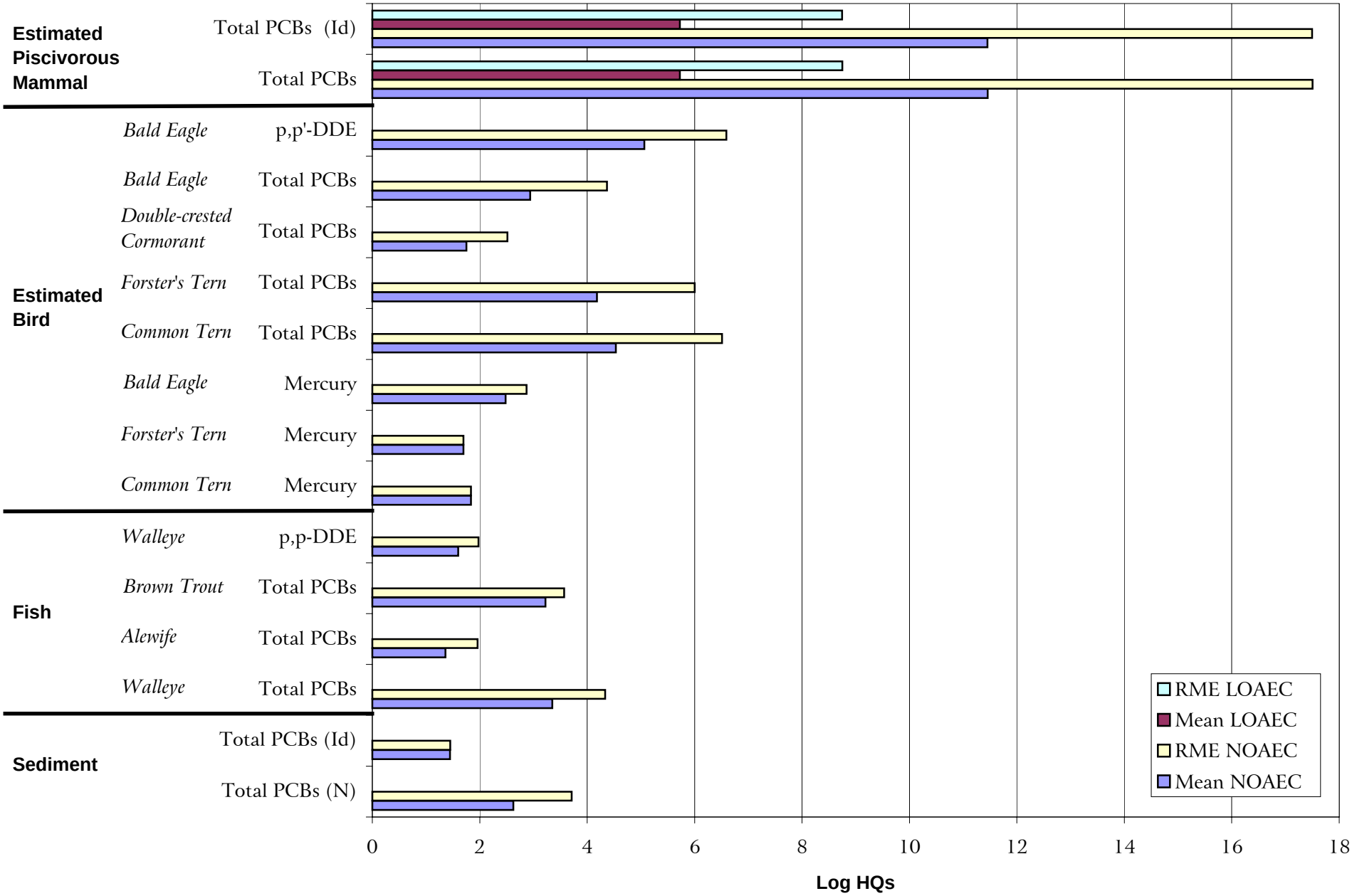
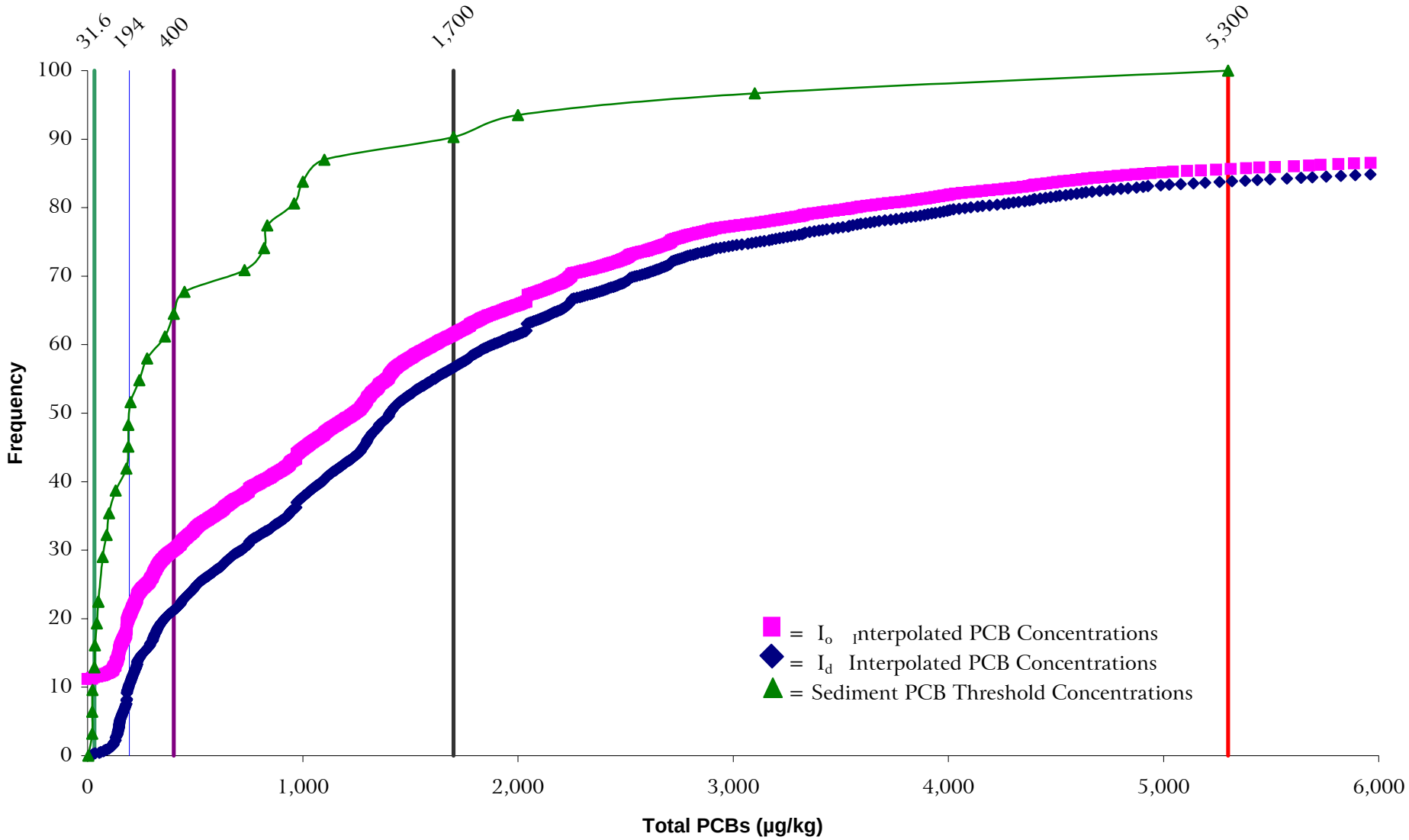
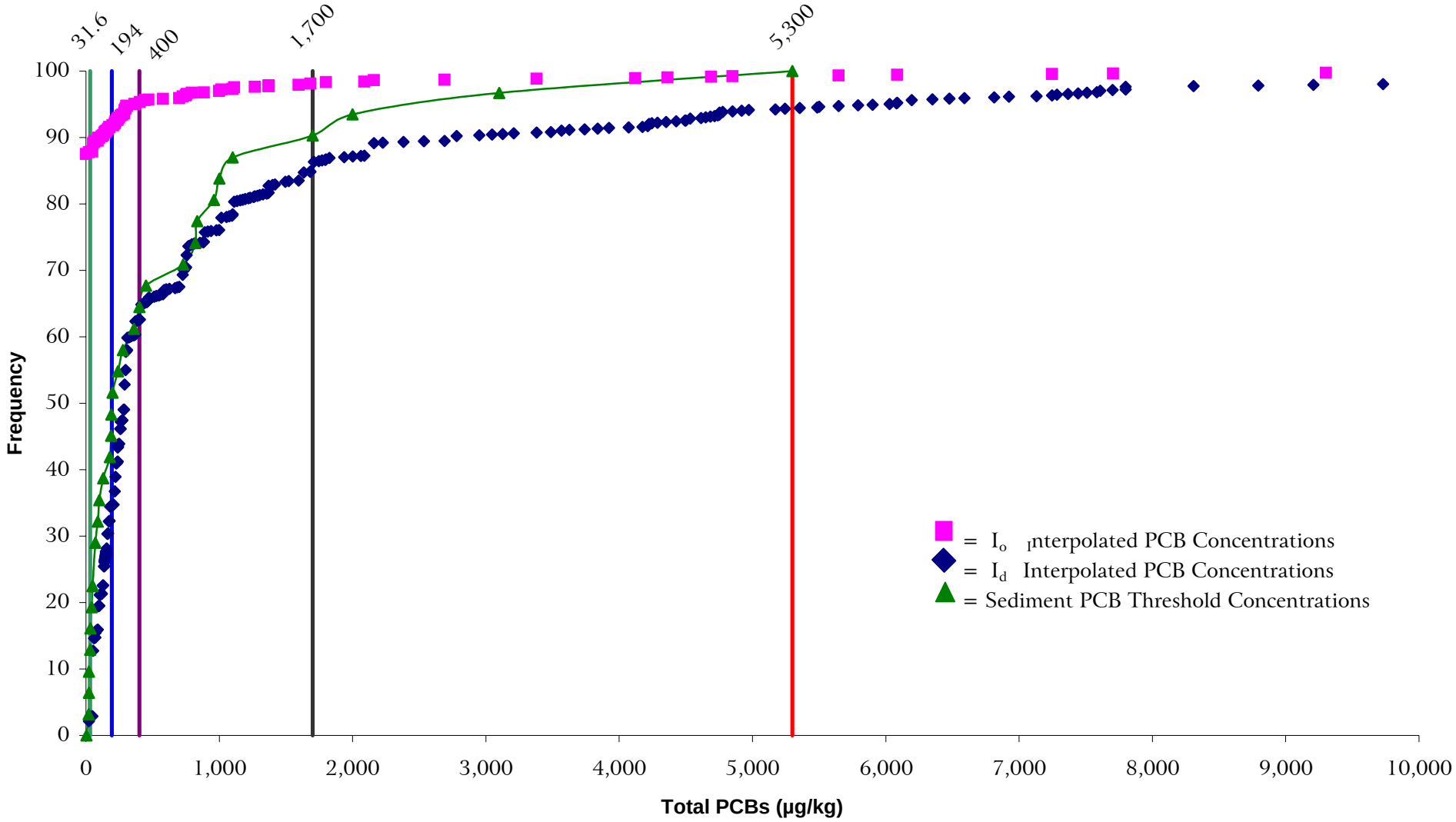


Figure 6-84 Little Lake Butte des Morts Sediment PCB Frequency Distribution



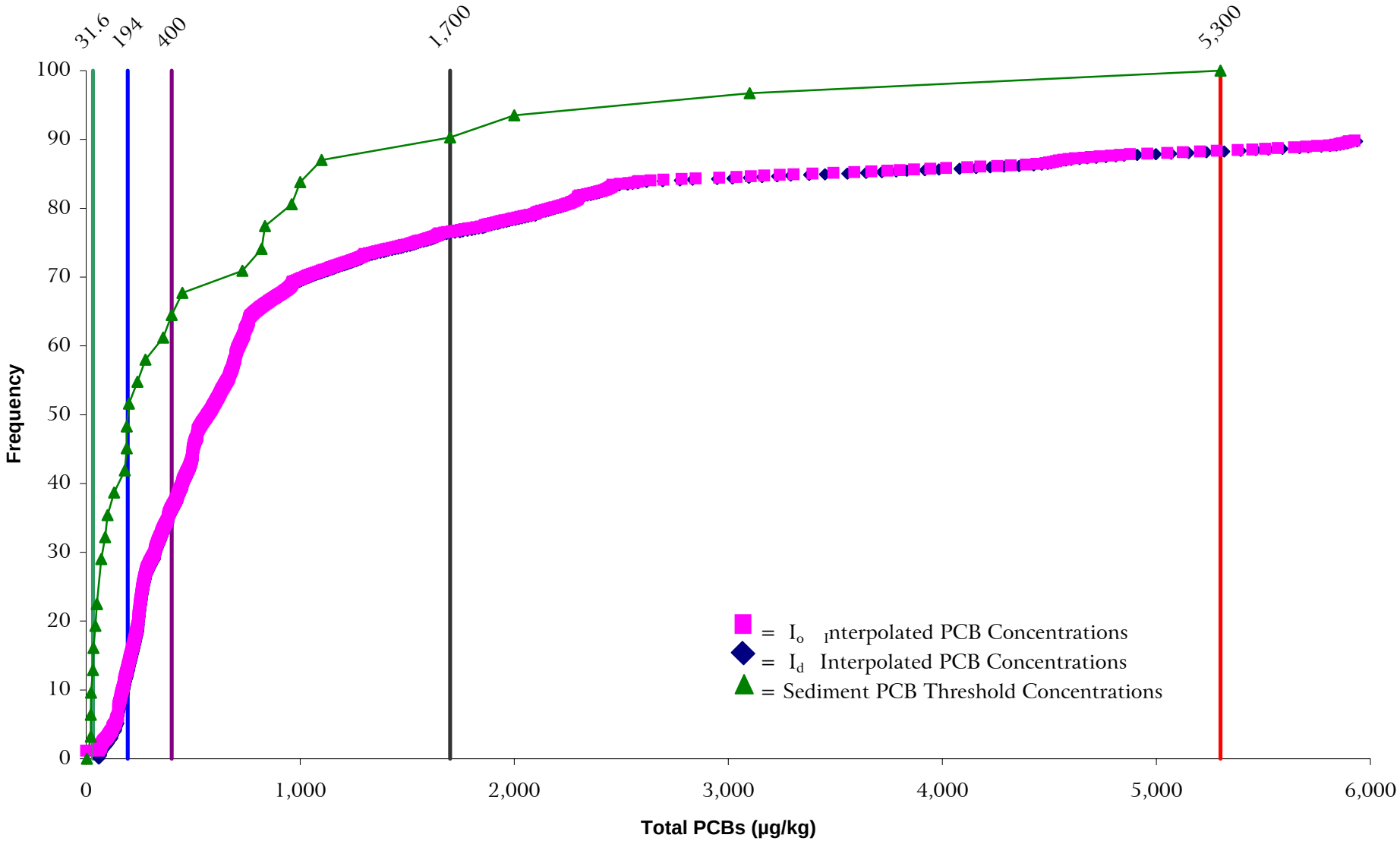
31.6 = ARCS Threshold Effects Concentration
194 = ARCS No Effects Concentration (high)
400 = Consensus-based Moderate Effects Concentration (MacDonald *et al.,* 2000)
1,700 = Consensus-based Extreme Effects Concentration (MacDonald *et al.,* 2000)
5,300 = OMOE Severe Effect Level (Persuad *et al.,* 1992)

Figure 6-85 Appleton to Little Rapids Sediment PCB Frequency Distribution



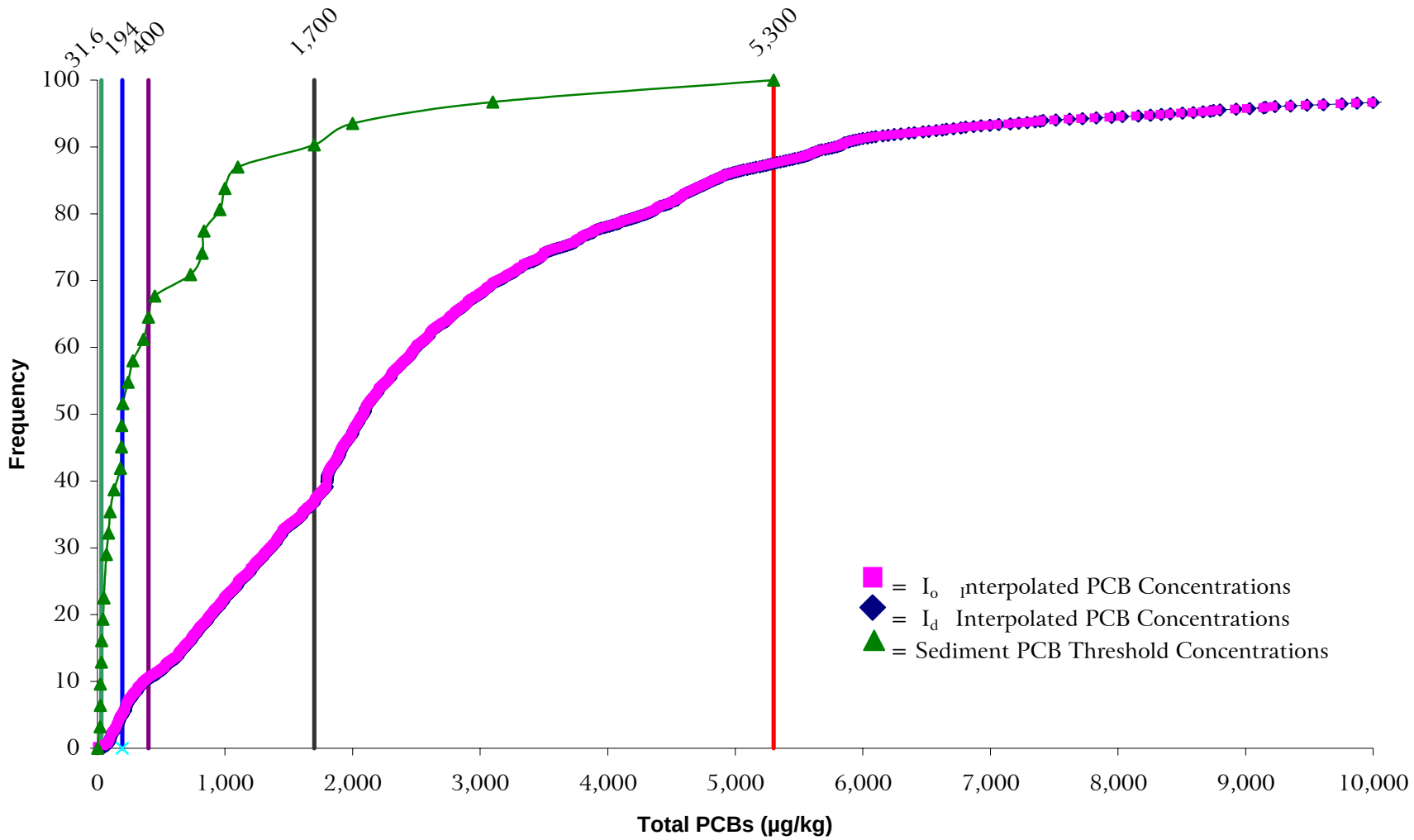
31.6 = ARCS Threshold Effects Concentration
194 = ARCS No Effects Concentration (high)
400 = Consensus-based Moderate Effects Concentration (MacDonald *et al.,* 2000)
1,700 = Consensus-based Extreme Effects Concentration (MacDonald *et al.,* 2000)
5,300 = OMOE Severe Effect Level (Persuad *et al.,* 1992)

Figure 6-86 Little Rapids to De Pere Sediment PCB Frequency Distribution



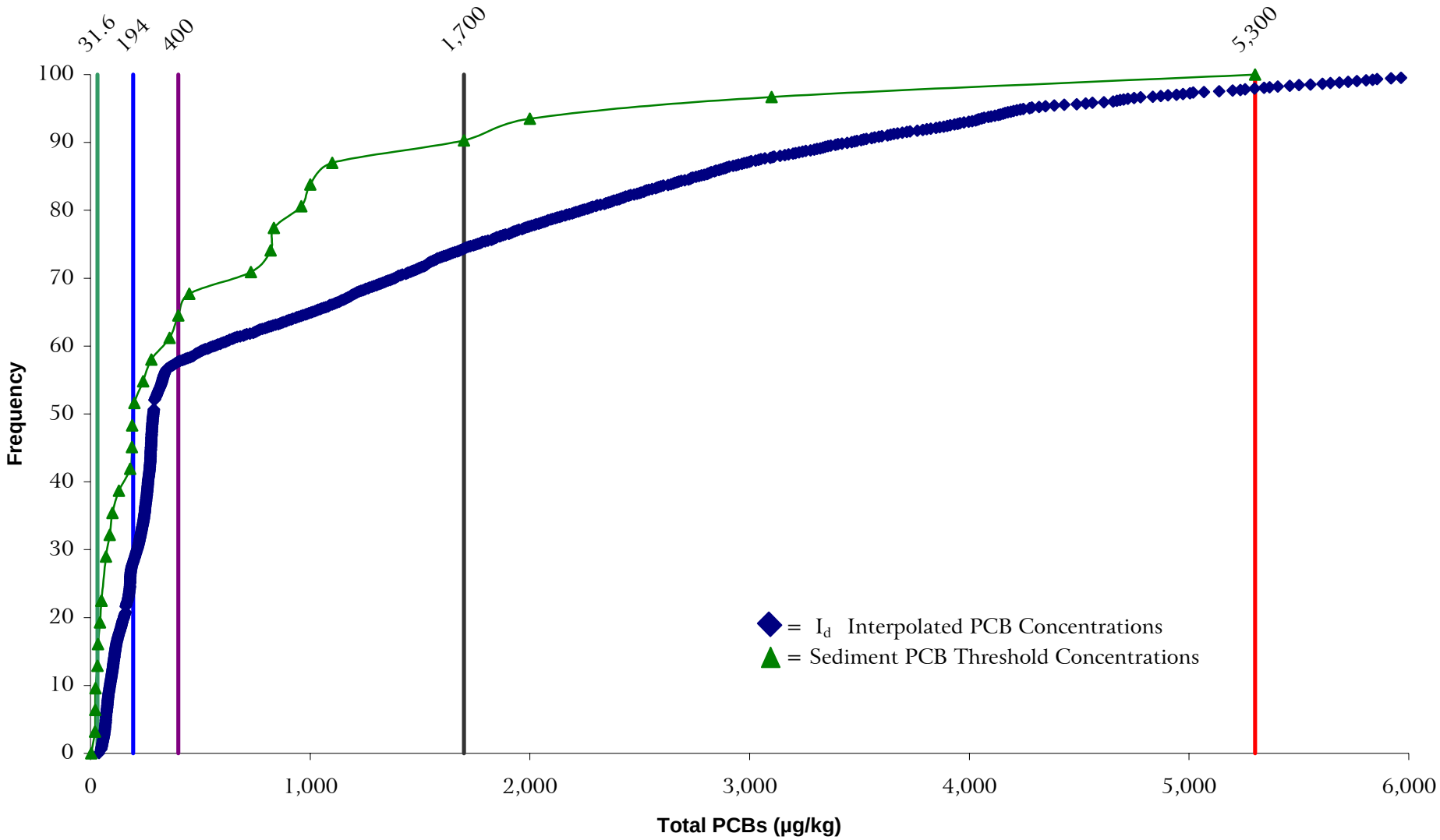
31.6 = ARCS Threshold Effects Concentration
194 = ARCS No Effects Concentration (high)
400 = Consensus-based Moderate Effects Concentration (MacDonald *et al.,* 2000)
1,700 = Consensus-based Extreme Effects Concentration (MacDonald *et al.,* 2000)
5,300 = OMOE Severe Effect Level (Persuad *et al.,* 1992)

Figure 6-87 De Pere to Green Bay (Zone 1) Sediment PCB Frequency Distribution



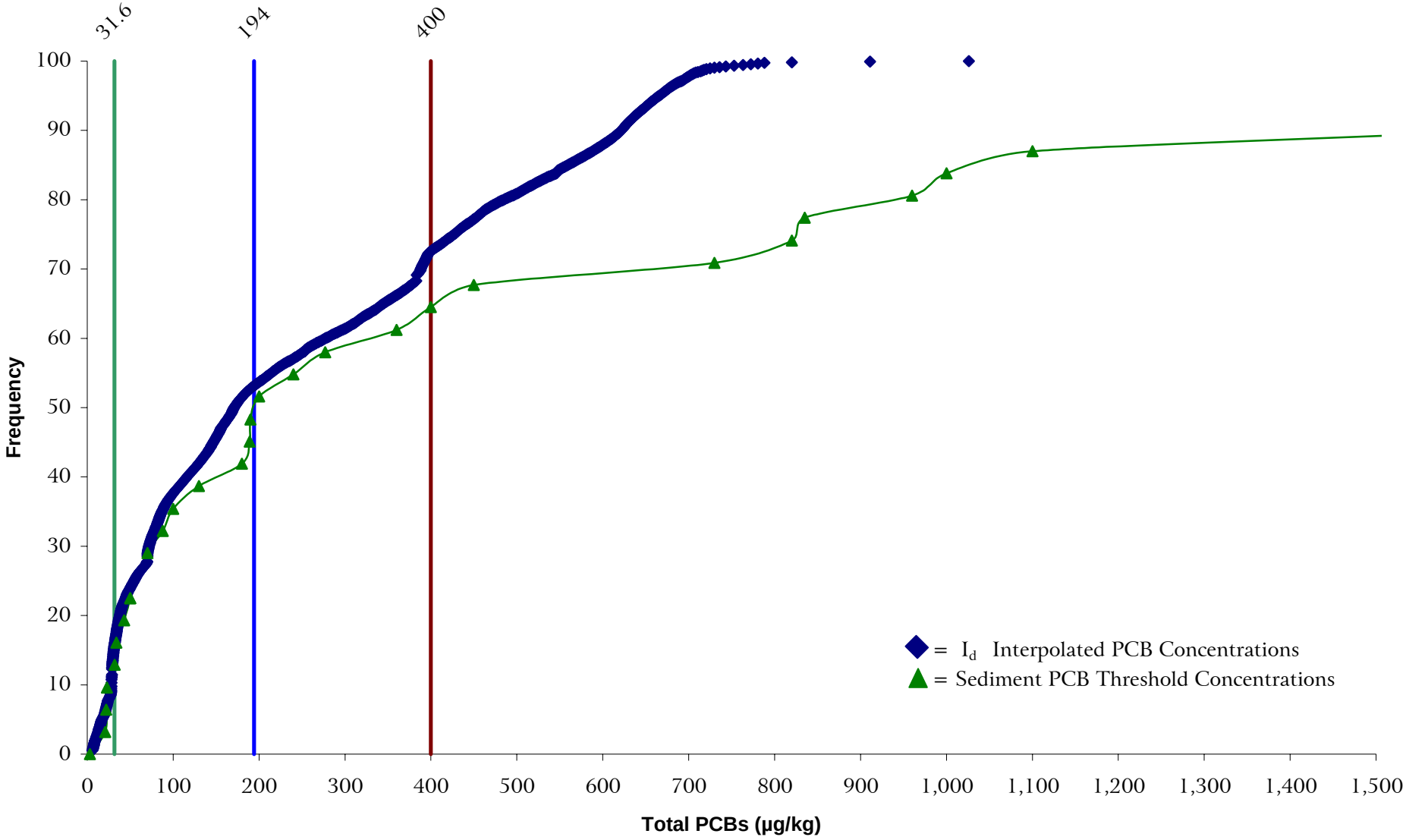
31.6 = ARCS Threshold Effects Concentration
194 = ARCS No Effects Concentration (high)
400 = Consensus-based Moderate Effects Concentration (MacDonald *et al.,* 2000)
1,700 = Consensus-based Extreme Effects Concentration (MacDonald *et al.,* 2000)
5,300 = OMOE Severe Effect Level (Persuad *et al.,* 1992)

Figure 6-88 Green Bay Zone 2 Sediment PCB Frequency Distribution



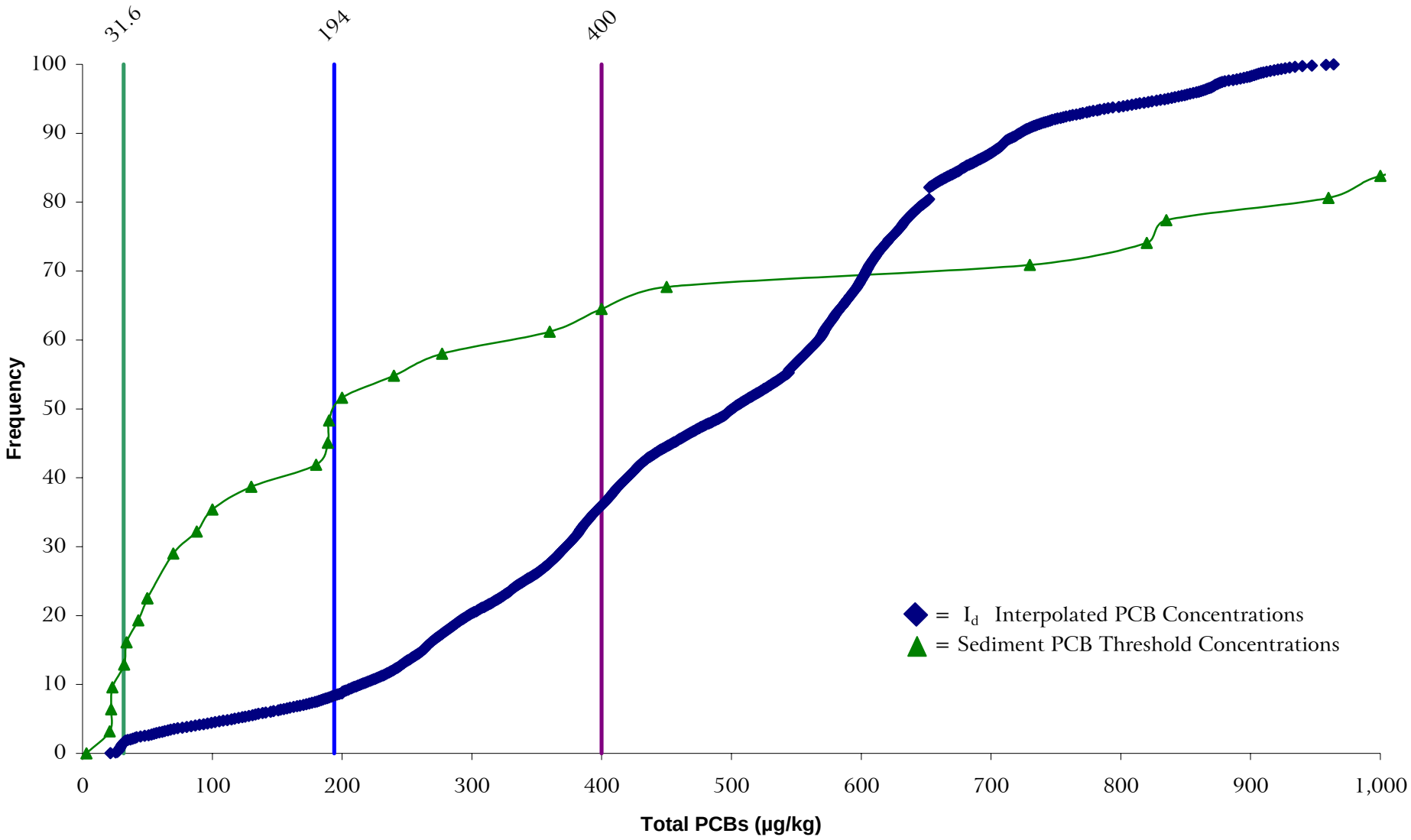
31.6 = ARCS Threshold Effects Concentration
194 = ARCS No Effects Concentration (high)
400 = Consensus-based Moderate Effects Concentration (MacDonald *et al.,* 2000)
1,700 = Consensus-based Extreme Effects Concentration (MacDonald *et al.,* 2000)
5,300 = OMOE Severe Effect Level (Persuad *et al.,* 1992)

Figure 6-89 Green Bay Zone 3A Sediment PCB Frequency Distribution



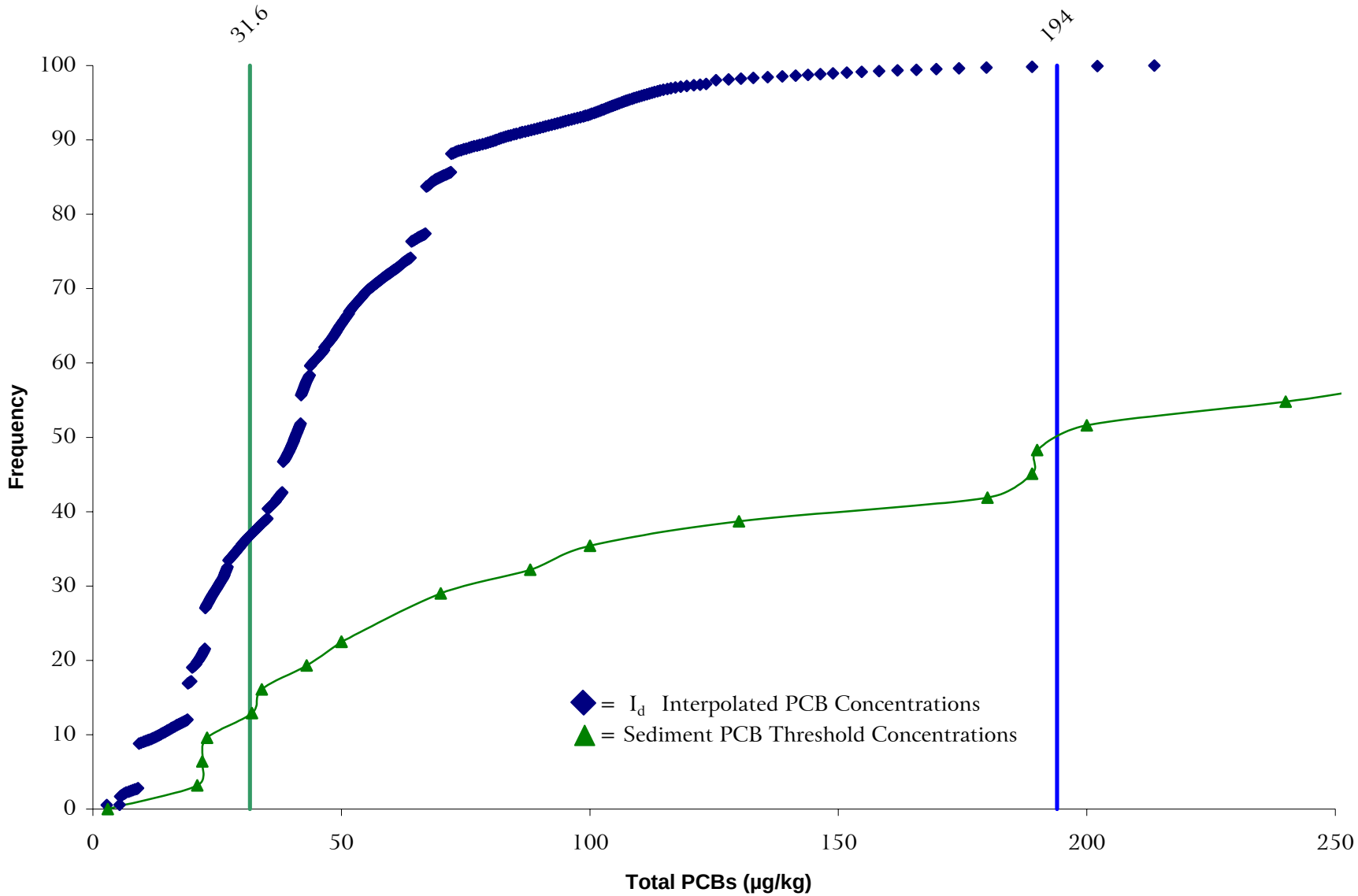
31.6 = ARCS Threshold Effects Concentration
194 = ARCS No Effects Concentration (high)
400 = Consensus-based Moderate Effects Concentration (MacDonald *et al.,* 2000)

Figure 6-90 Green Bay Zone 3B Sediment PCB Frequency Distribution



31.6 = ARCS Threshold Effects Concentration
194 = ARCS No Effects Concentration (high)
400 = Consensus-based Moderate Effects Concentration (MacDonald *et al.,* 2000)

Figure 6-91 Green Bay Zone 4 Sediment PCB Frequency Distribution



31.6 = ARCS Threshold Effects Concentration
194 = ARCS No Effects Concentration (high)

Table 6-1 Fate and Transport Properties of Potentially Bioaccumulating Chemicals of Concern

Constituent	Water Solubility (mg/L) (25 °C)	Log K_{ow}	Vapor Pressure (mm Hg) (25 °C)	Henry's Law Constant (atm-m^3/mol)
Organics				
PCB	0.24	6.3	4.06×10^{-4}	5.6×10^{-4}
TCDD	0.000317	7.0	7.2×10^{-10}	16.1×10^{-6} (25 °C)
TCDF	—	6.5	2.0×10^{-6}	—
DDT	0.00354	6.8	1×10^{-7}	1.29×10^{-5} (23 °C)
DDE	0.04 (20 °C)	6.0	6.5×10^{-6} (20 °C)	6.8×10^{-5}
DDD	20	5.9	4.68×10^{-6}	2.16×10^{-5}
Dieldrin	0.186 (20 °C)	5.5	3.1×10^{-6} (20 °C)	2×10^{-7}
Metals				
Mercury	0.056	CH_3HgCl 0.3, 0.4	2×10^{-3}	Hgo 6.97×10^{-3} $(CH_3)_2Hg$ 7.54×10^{3} $Hg(OH)_2$ 7.2×10^{-8}

References:

ATSDR., 1998c. *Toxicological Profile for Chlorinated Dibenzo-p-dioxins.* Agency for Toxic Substances and Disease Registry, U.S. Department of Health and Human Services.

EPA, 1992c. *National Study of Chemical Residues in Fish, Volume II.* EAP 823-R-92-008b. U.S. Environmental Protection Agency, Office of Science and Technology, Washington, D.C.

Montgomery, J. H., 1996. *Groundwater Chemicals Desk Reference, 2nd Edition.* Lewis Publishers, Boca Raton, Florida. 1,345 p.

Syracuse Research Corporation On-Line Log P Calculation at Website: http://esc.syrres.com/~esc1/kowint.htm.

Table 6-2 Assessment and Measurement Endpoints for the Ecological Risk Assessment

Assessment Endpoint (What is being protected?)	Risk Questions	Measurement Endpoint (What is being measured to assess environmental effects?)	Receptor Species	Risk Criteria (How are the measurements related to the assessment?)
1. Functioning water column invertebrate communities.	Are levels of site contaminants in surface water sufficient to cause adverse alterations to the functioning of water column invertebrate communities?	Surface water chemistry	Zooplankton	Water ecological benchmarks
2. Functioning benthic invertebrate communities.	Are levels of site contaminants in surface sediment sufficient to cause adverse alterations to the functioning of benthic invertebrate communities?	Sediment chemistry	Aquatic insects, molluscs, worms	Sediment ecological benchmarks
3. Benthic fish survival and reproduction.	Are levels of site contaminants sufficient to cause survival or reproductive impairment in benthic fish?	Whole fish tissue analysis	Carp	Whole body TRV
4. Pelagial fish survival and reproduction.	Are levels of site contaminants sufficient to cause survival or reproductive impairment in pelagial fish?	Whole fish tissue analysis	Shiners, rainbow smelt, gizzard shad, alewife, perch, brown trout, walleye	Whole body TRV
5. Insectivorous bird survival, physiology, and reproduction.	Are levels of site contaminants sufficient to cause survival or reproductive impairment, or deformity in insectivorous birds?	Whole body COPC levels	Tree swallow	Whole body TRV
		Egg COPC levels		Egg TRV
6. Piscivorous bird survival, physiology, and reproduction.	Are levels of site contaminants sufficient to cause survival or reproductive impairment, or deformity in piscivorous birds?	Whole body COPC levels	Double-crested cormorant, Forster's tern, common tern	Whole body TRV
		Egg COPC levels		Egg TRV
		Brain COPC levels		Brain TRV
		Food chain exposure modeling		Dietary TRV
7. Carnivorous bird survival, physiology, and reproduction.	Are levels of site contaminants sufficient to cause survival or reproductive impairment, or deformity in carnivorous birds?	Egg COPC levels	Bald eagle	Egg TRV
		Liver COPC levels		Liver TRV
		Food chain exposure modeling		Dietary TRV
8. Piscivorous mammal survival and reproduction.	Are levels of site contaminants sufficient to cause survival or reproductive impairment in piscivorous mammals?	Food chain exposure modeling	Mink	Dietary TRV

Table 6-3 Potential Ecotoxicological Effects from Chemicals Identified in the Lower Fox River/Green Bay

Chemical	Potential Ecotoxicological Effects			Exposure Medium	Exposure Routes
	Survival	Growth	Reproduction		
Chlorinated Organic Compounds					
Polychlorinated Biphenyls		✓	✓	water/sediments/food chain	ingestion, gill uptake, food chain transfer
DDT, DDD, DDE		✓	✓	sediments/food chain	ingestion, food chain transfer
Dioxin/Furans		✓	✓	sediments/food chain	ingestion, food chain transfer
Dieldrin		✓	✓	sediments/food chain	ingestion, food chain transfer
Metals					
Arsenic	✓	✓	✓	water/sediments	diffusion, ingestion
Lead	✓	✓	✓	water/sediments	diffusion, ingestion
Mercury	✓	✓	✓	water/sediments/food chain	ingestion, gill uptake, food chain transfer

Table 6-4 Exposure Modeling Input Parameters for Selected Receptor Species

Species	Body Weight (g) [1]	Food Type	Food Type as % of Diet	Food Ingestion (g/day)	Water Ingestion (L/day) [2]	Sediment as % of Diet	Sediment Ingestion (g/day) [5]
Mink	800	fish	85 [3]	153	0.081	2 [3]	4
Bald Eagle	4,650	TL3 fish TL4 fish	80 [4] 20 [4]	422 105	0.165	0	—
Common Tern	120	TL3 fish	100 [1]	58.8	0.014	0	—
Forster's Tern	158	TL3 fish	100 [1]	71.4	0.017	0	—
Double-crested Cormorant	1,680	TL3 fish	100 [1]	318	0.084	0	—

Notes:

[1] Presented in *Trophic Level and Exposure Analyses for Selected Piscivorous Birds and Mammals, Volume I: Analyses of Species in the Great Lakes Basin* (EPA, 1995e) and summarized in the *Great Lakes Water Quality Initiative (GLWQI) Technical Support Document for Wildlife Criteria* (EPA, 1995d).

[2] Calculated using reported body weight and the allometric equations presented in the EPA *Wildlife Exposure Factors Handbook* (EPA, 1993a).

[3] Based on the research by Alexander (1977) as presented in the EPA *Wildlife Exposure Factors Handbook* (EPA, 1993a) and assuming that vegetation ingestion and sediment ingestion are equivalent.

[4] While EPA sources (1995d and 1995e) indicated that bald eagles consume 92 percent fish and 8 percent birds, for the exposure modeling it was conservatively estimated that bald eagles consume 100 percent fish. The proportions of trophic level 3 and 4 fish are the same as those indicated by the EPA (1995d and 1995e).

[5] Calculated based on a total food ingestion rate of 179.9 g/day.

Table 6-5 Selected Values as Criteria or TRVs

COPC	Measurement Endpoint Receptor	Media	Concentration	Units	Type	Effect	Reference
Total PCBs	Water Column Invertebrates	water	50	ng/L	estimated NOAEC	sublethal	Niimi, 1996
			0.5	µg/L	LOAEC		
	Benthic Invertebrates	sediment	31.6	µg/kg dwt	ARCS SEC	survival, growth, sexual maturation	EPA, 1996a
	All Fish	whole body	0.76	mg/kg	NOAEC	fry growth	Mac and Seelye, 1981; Hendricks *et al.,* 1981
			7.6		estimated LOAEC		
	All Birds	egg	4.7	mg/kg	NOAEC	decreased hatching success	Hoffman *et al.,* 1993
			7.6		LOAEC		
			0.8	mg/kg ww	NOAEC	deformity	Ludwig *et al.,* 1996
			8		estimated LOAEC		
	Piscivorous and Carnivorous Birds	diet	0.112	mg/kg-BW/day	estimated NOAEC	female fertility	Tori and Peterle, 1983; Peakall and Peakall, 1973
			1.12		LOAEC		
	Mink	diet	0.05	mg/kg-BW/day	NOAEC	reproduction and kit survival	Heaton *et al.,* 1995a; Restum *et al.,* 1998
			0.1		LOAEC		
TCDD-Eq	Benthic Invertebrates	sediment	0.0039	µg/kg	sediment screening value		EPA, 1997b
	All Fish	egg	41	ng/kg	NOAEC	sac fry mortality	Johnson *et al.,* 1998b
			84		LOAEC		
	All Birds	egg	7	ng/kg	NOAEC	reproductive impairment	Giesy *et al.,* 1994b; Froese *et al.* , 1998
			191		LD_{20}	egg lethality	derived from Giesy *et al.,* 1994b and Tillitt *et al.,* 1992
			308		LD_{30}		
			38		NOAEC	deformity	Ludwig *et al.,* 1996
			380		estimated LOAEC		
4,4'-DDT	Water Column Invertebrates	water	1	ng/L	chronic NAWQC		EPA, 1998a
Total DDT	Benthic Invertebrates	sediment	7	µg/kg dwt	TEL		Smith *et al.,* 1996
4,4'-DDE			1.42				
4,4'-DDD			3.54				
DDT	All Fish	egg/embryo	0.3	mg/kg	estimated NOAEC	fry mortality	Burdick *et al.,* 1964
			2.95		LOAEC		
DDT Equivalents	All Birds	brain	1.8	mg/kg	estimated NOAEC	mortality	Blus, 1996
			18		LOAEC		

Table 6-5 Selected Values as Criteria or TRVs (Continued)

COPC	Measurement Endpoint Receptor	Media	Concentration	Units	Type	Effect	Reference
DDE	All Birds	eggs	3	mg/kg	NOAEC	mean 5-yr productivity	Wiemeyer *et al.,* 1984
			5.1		LOAEC		
	Piscivorous and Carnivorous Birds	diet	0.018	mg/kg-BW/day	estimated NOAEC	eggshell thinning and hatching success	Longcore and Samson, 1973
			0.18		LOAEC		
DDT/DDE	Mink	diet	19.1	mg/kg-BW/day	NOAEC	reproductive impairment	Aulerich and Ringer, 1970; Duby *et al.,* 1971; Giesy *et al.,* 1994d
			191		estimated LOAEC		
Dieldrin	Water Column Invertebrates	water	0.077	µg/L	Wisconsin surface water quality criteria		Wisconsin Administrative Code Chapter NR 105
	Benthic Invertebrates	sediment	11	mg/kg-organic carbon	federal sediment quality guidelines		Federal Register, Vol. 59, No. 11, January 18, 1994
	All Fish	whole body	0.37	mg/kg	estimated NOAEC	abnormal behavior/convulsions	Gakstatter and Weiss, 1967
			3.7		LOAEC		
	All Birds	brain	0.49	mg/kg	estimated NOAEC	mortality	Stickel *et al.,* 1969
			4.9		LOAEC		
		egg	0.1		NOAEC	egg mortality	Giesy *et al.,* 1995; Wiemeyer *et al.,* 1984
			1		LOAEC		
	Piscivorous and Carnivorous Birds	diet	0.11	mg/kg-BW/day	estimated NOAEC	chick survival	Dahlgren and Linder, 1974
			1.06		LOAEC		
	Mink	diet	0.009	mg/kg-BW/day	NOAEC	reproductive impairment	Harr *et al.,* 1970b
			0.018		LOAEC		
Arsenic	Water Column Invertebrates	water	152.2	µg/L	Wisconsin surface water quality criteria		Wisconsin Administrative Code Chapter NR 105
	Benthic Invertebrates	sediment	12.1	mg/kg dwt	ARCS SEC		EPA, 1996a
	All Fish	whole body	0.5	mg/kg	NOAEC	mortality	Barrows *et al.,* 1980
			5		estimated LOAEC		
Lead	Water Column Invertebrates	water	49.42	µg/L	Wisconsin surface water quality criteria		Wisconsin Administrative Code Chapter NR 105
	Benthic Invertebrates	sediment	34.2	mg/kg dwt	ARCS SEC		EPA, 1996a

Table 6-5 Selected Values as Criteria or TRVs (Continued)

COPC	Measurement Endpoint Receptor	Media	Concentration	Units	Type	Effect	Reference
Mercury	Water Column Invertebrates	water	0.44	µg/L	Wisconsin surface water quality criteria		Wisconsin Administrative Code Chapter NR 105
	Benthic Invertebrates	sediment	0.17	mg/kg dwt	TEL		Smith *et al.,* 1996
	All Fish	whole	0.25	mg/kg ww	NOAEC	juvenile growth and gonad development	Friedmann *et al.,* 1996
			2.37		LOAEC		
	All Birds	egg	0.08	mg/kg	estimated NOAEC	egg mortality	Heinz, 1979
			0.8		LOAEC		
		liver	0.2		estimated NOAEC	reproductive impairment	EPA, 1997e; Fimreite, 1971; Scheuhammer, 1987
			2		LOAEC		
	Piscivorous and Carnivorous Birds	diet	0.008	mg/kg-BW/day	estimated NOAEC	mortality	Heinz, 1979
			0.078		LOAEC		
	Mink	diet	0.084	mg/kg-BW/day	NOAEC	lesions the nervous system	Wobeser *et al.,* 1976a
			0.21		LOAEC		

Table 6-6 Toxic Equivalent Factors (TEFs) Used for TCDD-Eq Calculations

Congener No.	Structure	Avian			Fish		
		Van den Berg *et al.*, 1998 *	Tillitt *et al.*, 1991b*	Kennedy *et al.*, 1996	Newsted *et al.*, 1995	Van den Berg *et al.*, 1998 *	Zabel *et al.*, 1995
	2,3,7,8-TCDD	1	1	1		1	1
	1,2,3,7,8,-PeCDD	1		1.1		1	0.73
	1,2,3,4,7,8-HxCDD	0.05				0.5	0.319
	1,2,3,6,7,8-HxCDD	0.01				0.01	0.024
	1,2,3,7,8,9-HxCDD	0.1				0.01	
	1,2,3,4,6,7,8-HpCDD	<0.001				0.001	0.002
	OCDD	0.0001				<0.0001	
	2,3,7,8-TCDF	1	0.0064	1.1		0.05	0.028
	1,2,3,7,8-PeCDF	0.1				0.05	0.034
	2,3,4,7,8-PCDF	1				0.5	0.359
	1,2,3,4,7,8-HxCDF	0.1				0.1	0.28
	1,2,3,6,7,8-HxCDF	0.1				0.1	
	1,2,3,7,8,9-HxCDF	0.1				0.1	
	2,3,4,6,7,8-HxCDF	0.1				0.1	
	1,2,3,4,6,7,8-HpCDF	0.01				0.01	
	1,2,3,4,7,8,9-HpCDF	0.01				0.01	
	OCDF	0.0001				<0.0001	
77	3,3', 4,4'-tetrachlorobiphenyl	0.05	0.000018	0.03	0.0054	0.0001	0.00016
81	3,4,4',5-tetrachlorobiphenyl	0.1	0.0019	0.2	—	0.0005	0.00056
105	2,3,3',4,4'-pentachlorobiphenyl	0.0001	0.0000076	0.005	<0.00005	<0.000005	0.00000172
118	2,3',4,4',5-pentachlorobiphenyl	0.00001	0.00000037	0.001	<0.00006	<0.000005	0.00000302
126	3,3',4,4',5-pentachlorobiphenyl	0.1	0.022	0.3	0.13	0.005	0.005
169	3,3'4,4',5,5'-hexachlorobiphenyl	0.001	0.00047	0.02	0.0088	0.00005	0.000041

Notes:

* TEFs selected for use in this assessment.

Van den Berg *et al.,* 1998 TEFs are the WHO TEFs.

Table 6-7 Determination of Effects-based TRV for Piscivorous Bird Eggs of the Lower Fox River and Green Bay

Bird Species	Observed Effects vs. Reported Concentrations in Eggs			Predicted Effect Level Based on the Giesy Regression Formula
	TCDD-Eq (pg/g) [1]	TCDD-Eq (pg/g) [2]	LD_n - egg	TCDD-Eq (pg/g)
Double-crested Cormorant	1,029		100	
Caspian Tern	750		50	
Double-crested Cormorant	460		50	
Double-crested Cormorant		299	39	
Double-crested Cormorant	344		37	
Double-crested Cormorant		344	37	
Caspian Tern	416		35	
Double-crested Cormorant		344	32	
Double-crested Cormorant		248	30	308
Double-crested Cormorant		217	27	
Double-crested Cormorant	217		27	
Double-crested Cormorant		103	26	
Double-crested Cormorant		95	25	
Double-crested Cormorant		206	24	
Double-crested Cormorant		192	24	
Double-crested Cormorant		157	23	
Double-crested Cormorant		201	22	
Double-crested Cormorant		85	21	
Double-crested Cormorant			20	191
Double-crested Cormorant	35		8	
Double-crested Cormorant		35	8	
No Observed Effects Level[3]	7		0	
Line Regression	y = 0.085x + 3.806	y = 0.067x +13.1		
R^2 Value	0.923	0.703		
p value	0.00015	0.0003		

Notes:

[1] LD_n data from Giesy *et al.,* 1994a.

[2] Values generated from Tillitt *et al.*'s (1992) regression equation for observed effects on double-crested cormorants (DCC).

[3] NOEL value for avifauna from Froese *et al.,* 1998.

Table 6-8 Surface Water Concentrations in Little Lake Butte des Morts Reach

Analyte	Number of Samples	Number of Detects	Detection Frequency (%)	Detected Minimum	Detected Maximum	Mean	Data Distribution	95% UCL	90th Percentile	RME	Criteria NOAEC	Criteria LOAEC	Reference	Hazard Quotients Mean NOAEC	Hazard Quotients Mean LOAEC	Hazard Quotients RME NOAEC	Hazard Quotients RME LOAEC
Metals (ng/L)																	
Lead (filtered)	1	1	100	117	117	117	—	—	—	117	49,420		[1]	< 0.1		< 0.1	
Lead (unfiltered)	1	1	100	1,450	1,450	1,450	—	—	—	1,450	49,420		[1]	< 0.1		< 0.1	
Mercury (filtered)	2	0															
Mercury (unfiltered)	6	5	83	0.2	7,140	2,237	Lognormal	4.85E+20	0.0	7,140	440		[1]	5.1		16	
PCBs (ng/L)																	
Total PCBs (filtered)	46	40	87	1.4	19.0	11.1	Other	15.3	25.0	15.3	50	500	[2]	0.2	< 0.1	0.3	< 0.1
Total PCBs (unfiltered)	6	0															
Total PCBs (particulate)	41	34	83	0.1	40.2	16.6	Other	53.8	36.1	40.2	50	500	[2]	0.3	< 0.1	0.8	0.1
Total PCBs (filtered + particulate)				1.5	59.2	27.6		69.1	61.1	55.5	50	500	[2]	0.6	0.1	1.1	0.1

Notes:

[1] Wisconsin Administrative Code Chapter NR 105.

[2] Niimi, 1996.

Table 6-9 Surface Water Concentrations in Lake Winnebago

Analyte	Number of Samples	Number of Detects	Detection Frequency (%)	Detected Minimum	Detected Maximum	Mean	Data Distribution	95% UCL	90th Percentile
Metals (ng/L)									
Mercury (filtered)	1	0							
Mercury (unfiltered)	1	0							
PCBs (ng/L)									
Total PCBs (filtered)	10	2	20	5	7	—	Other	14.3	13.5
Total PCBs (particulate)	10	3	30	3.2	6	—	Other	15.6	13.5
Total PCBs (filtered + particulate)				8	13	—		29.9	26.9

Note:

"—" indicates that the calculated mean exceeded the detected maximum and, therefore, the mean was not evaluated.

Table 6-10 Surface Sediment Concentrations in Little Lake Butte des Morts Reach

Analyte	Number of Samples	Number of Detects	Detection Frequency (%)	Detected Minimum	Detected Maximum	Mean	Data Distribution	95% UCL	90th Percentile	RME	Criteria	Reference	Hazard Quotients	
													Mean	RME
Metals (mg/kg)														
Arsenic	27	24	89	2.6	6.8	4.6	Normal	5.1	6.2	5.1	12.1	1	0.4	0.4
Lead	27	27	100	3.8	522	172	Other	723	457	522	34.2	1	5.0	15
Mercury	86	71	83	0.2	3.3	1.0	Other	1.4	2.2	1.4	0.17	3	5.6	8.5
Dioxins/Furans (µg/kg)														
2,3,7,8-TCDD	5	4	80	1.8E-03	5.4E-03	2.5E-03	Normal	4.3E-03	0.0	4.3E-03	3.90E-03	4	0.6	1.1
2,3,7,8-TCDF	5	5	100	5.0E-02	7.1E-02	6.4E-02	Normal	7.2E-02	0.0	7.1E-02				
PCBs (µg/kg)														
Total PCBs (N)	302	294	97	25.0	130,000	10,724	Lognormal	22,848	33,400	22,848	31.6	1	339	723
Total PCBs (I_0)	57,724	57,724	100	0.0	60,000	3,284	Other	3,330	8,733	3,330	31.6	1	104	105
Total PCBs (I_d)	51,261	51,261	100	20.5	60,000	3,699	Other	3,749	9,951	3,749	31.6	1	117	119
Pesticides (µg/kg)														
Dieldrin	15	1	7	5.9	5.9	NE	Other	68.9	148	5.9	11,000	2	NE	< 0.1
p,p'-DDD	23	4	17	4.7	19.0	17.8	Lognormal	41.8	37.6	19.0	3.54	3	5.0	5.4
p,p'-DDE	20	0												
p,p'-DDT	20	2	10	13.0	50.0	NE	Other	114	50.0	50.0	7.00	3	NE	7.1
Total Organic Carbon (mg/kg)														
Total Organic Carbon	275	255	93	4,960	778,000	142,037	Other	160,586	484,400	160,586				

Notes:

N indicates that the data was not interpolated based on depth.

I_0 indicates that interpolated grid areas for which "no values" existed were assumed to equal zero.

I_d indicates that interpolated grid areas for which "no values" existed were deleted from the database.

[1] ARCS SEC (EPA, 1996a).

[2] Federal Sediment Quality Guidelines (µg/kg OC) (EPA, 1997d).

[3] Environment Canada TEL (Smith *et al.,* 1996); p,p'-DDT TEL based on total DDT TEL.

[4] EPA Sediment Screening Values (EPA, 1997b).

NE indicates that the calculated mean exceeded the detected maximum and, therefore, the mean was not evaluated.

Table 6-11 Surface Sediment Concentrations in Lake Winnebago

Analyte	Number of Samples	Number of Detects	Detection Frequency (%)	Detected Minimum	Detected Maximum	Mean	Data Distribution	95% UCL	90th Percentile
Metals (mg/kg)									
Arsenic	3	3	100	4.0	6.0	5.3	Other	10.6	0.0
Lead	3	3	100	30.0	39.0	35.0	Normal	42.7	0.0
Mercury	3	3	100	0.1	0.2	0.1	Normal	0.2	0.0
PCBs (µg/kg)									
Total PCBs	5	5	100	6.0	36.0	22.0	Normal	35.1	0.0
Pesticides (µg/kg)									
Dieldrin	3	0							
p,p'-DDD	3	0							
p,p'-DDE	3	2	67	2.4	3.5	2.7	Normal	3.9	0.0
p,p'-DDT	3	0							

Table 6-12 PCB Congener Concentrations in Surface Sediment in Little Lake Butte des Morts Reach

Analyte	Number of Samples	Number of Detects	Detection Frequency (%)	Detected Minimum	Detected Maximum	Mean	Data Distribution	95% UCL	90th Percentile	RME
PCBs (µg/kg)										
PCB Congener 77	18	14	78	1.5	52.0	11.3	Lognormal	26.4	36.8	26.4
PCB Congener 81	16	10	63	0.05	0.6	NE	Other	3.6	2.3	0.6
PCB Congener 105	18	16	89	1.2	48.0	6.6	Lognormal	10.6	16.1	10.6
PCB Congener 118	46	46	100	1.3	3700	257.1	Lognormal	596	443	596
PCB Congener 126	18	8	44	0.02	0.3	NE	Other	4.3	2.3	0.3
PCB Congener 169	20	0								

Note:

NE indicates that the calculated mean exceeded the detected maximum and, therefore, the mean was not evaluated.

Table 6-13 PCB Congener Concentrations in Surface Sediment in Lake Winnebago

Analyte	Number of Samples	Number of Detects	Detection Frequency (%)	Detected Minimum	Detected Maximum	Mean	Data Distribution	95% UCL	90th Percentile
PCBs (µg/kg)									
PCB Congener 77	3	0							
PCB Congener 81	3	0							
PCB Congener 105	3	0							
PCB Congener 118	3	0							
PCB Congener 126	3	0							
PCB Congener 169	3	0							

Table 6-14 Little Lake Butte des Morts Reach Whole Fish Concentrations

Analyte	Species	Number of Samples	Number of Detects	Detection Frequency (%)	Detected Minimum	Detected Maximum	Mean	Data Distribution	95% UCL	90th Percentile	RME
Metals (mg/kg)											
Arsenic	Carp	2	2	100	0.13	0.15	0.14	Normal	0.20	0.0	0.15
Mercury	Yellow Perch	2	0								
	Carp	5	3	60	0.04	0.06	0.05	Normal	0.06	0.0	0.06
	Walleye	4	1	25	0.03	0.03	NE	Other	0.07	0.0	0.03
PCBs (µg/kg)											
Total PCBs	Gizzard Shad	4	4	100	54.0	530	296	Normal	544	0.0	530
	Golden Shiner	2	2	100	845	1,140	993	Normal	1,924	0.0	1,140
	Yellow Perch	1	1	100	363	NA	—				363
	Carp	30	30	100	245	11,400	1,992	Lognormal	2,957	4,060	2,957
	Walleye	13	11	85	98.9	3,800	1,159	Lognormal	4,892	3,800	3,800
Pesticides (µg/kg)											
Dieldrin	Yellow Perch	2	0								
	Carp	6	2	33	0.7	1.0	NE	Lognormal	356	0.0	1.0
	Walleye	7	0								
o,p'-DDD	Carp	5	0								
	Walleye	4	0								
o,p'-DDE	Carp	4	1	25	5.8	5.8	NE	Normal	21.6	0.0	5.8
	Walleye	4	1	25	16.0	16.0	12.5	Normal	24.3	0.0	16.0
o,p'-DDT	Carp	5	0								
	Walleye	4	0								
p,p'-DDD	Yellow Perch	2	0								
	Carp	7	3	43	2.4	5.2	NE	Lognormal	52.6	0.0	5.2
	Walleye	7	1	14	78.0	78.0	23.5	Normal	44.9	0.0	44.9
p,p'-DDE	Yellow Perch	2	2	100	8.0	11.0	9.5	Normal	19.0	0.0	11.0
	Carp	7	5	71	8.0	30.0	16.9	Normal	23.8	0.0	23.8
	Walleye	7	5	71	26.0	77.0	47.6	Normal	61.7	0.0	61.7
p,p'-DDT	Yellow Perch	2	0								
	Carp	7	0								
	Walleye	7	0								

Note:

NA - Not applicable.

NE indicates that the calculated mean exceeded the detected maximum and, therefore, the mean was not evaluated.

Table 6-15 Dioxin and PCB Congener Concentrations in Little Lake Butte des Morts Reach Whole Fish

Analyte	Species	Number of Samples	Number of Detects	Detection Frequency (%)	Detected Minimum	Detected Maximum	Mean	Data Distribution	95% UCL	90th Percentile	RME
Dioxins/Furans (µg/kg)											
1,2,3,4,6,7,8,9-OCDD	Carp	3	3	100	7.00E-03	1.48E-02	1.04E-02	Normal	1.71E-02	0.0	1.48E-02
1,2,3,4,6,7,8,9-OCDF	Carp	3	0								
1,2,3,4,6,7,8-HPCDD	Carp	3	3	100	5.20E-03	7.30E-03	6.30E-03	Normal	8.08E-03	0.0	7.30E-03
1,2,3,4,6,7,8-HPCDF	Carp	3	3	100	8.90E-04	1.10E-03	9.93E-04	Normal	1.17E-03	0.0	1.10E-03
1,2,3,4,7,8,9-HPCDF	Carp	3	0								
1,2,3,4,7,8-HXCDD	Carp	3	3	100	5.40E-04	1.00E-03	7.80E-04	Normal	1.17E-03	0.0	1.00E-03
1,2,3,4,7,8-HXCDF	Carp	3	2	67	3.20E-04	3.40E-04	2.58E-04	Normal	4.68E-04	0.0	3.40E-04
1,2,3,6,7,8-HXCDD	Carp	3	3	100	1.80E-03	2.60E-03	2.17E-03	Normal	2.85E-03	0.0	2.60E-03
1,2,3,6,7,8-HXCDF	Carp	3	3	100	3.10E-04	4.00E-04	3.63E-04	Normal	4.99E-04	0.0	4.00E-04
1,2,3,7,8,9-HXCDD	Carp	3	2	67	3.80E-04	3.90E-04	3.48E-04	Normal	4.56E-04	0.0	3.90E-04
1,2,3,7,8,9-HXCDF	Carp	3	0								
1,2,3,7,8-PECDD	Carp	3	3	100	4.20E-04	9.70E-04	6.40E-04	Normal	1.13E-03	0.0	9.70E-04
1,2,3,7,8-PECDF	Carp	3	3	100	5.30E-04	5.50E-04	5.37E-04	Other	5.59E-04	0.0	5.50E-04
2,3,4,6,7,8-HXCDF	Carp	3	3	100	1.80E-04	2.40E-04	2.13E-04	Normal	2.65E-04	0.0	2.40E-04
2,3,4,7,8-PECDF	Carp	3	2	67	5.70E-04	6.60E-04	5.12E-04	Normal	8.23E-04	0.0	6.60E-04
2,3,7,8-TCDD	Carp	3	3	100	2.10E-04	2.90E-04	2.53E-04	Normal	3.21E-04	0.0	2.90E-04
2,3,7,8-TCDF	Carp	3	3	100	1.90E-03	2.50E-03	2.20E-03	Normal	2.71E-03	0.0	2.50E-03
1,2,3,4,6,7,8,9-OCDD	Walleye	3	2	67	9.90E-04	1.20E-03	9.30E-04	Normal	1.44E-03	0.0	1.20E-03
1,2,3,4,6,7,8,9-OCDF	Walleye	3	0								
1,2,3,4,6,7,8-HPCDD	Walleye	3	2	67	1.10E-03	1.20E-03	9.25E-04	Normal	1.59E-03	0.0	1.20E-03
1,2,3,4,6,7,8-HPCDF	Walleye	3	2	67	4.50E-04	4.50E-04	3.53E-04	Other	1.03E-02	0.0	4.50E-04
1,2,3,4,7,8,9-HPCDF	Walleye	3	0								
1,2,3,4,7,8-HXCDD	Walleye	3	2	67	2.50E-04	3.10E-04	2.18E-04	Normal	4.05E-04	0.0	3.10E-04
1,2,3,4,7,8-HXCDF	Walleye	3	0								
1,2,3,6,7,8-HXCDD	Walleye	3	3	100	1.20E-03	1.30E-03	1.27E-03	Other	1.36E-03	0.0	1.30E-03
1,2,3,6,7,8-HXCDF	Walleye	3	3	100	2.30E-04	2.70E-04	2.43E-04	Other	2.82E-04	0.0	2.70E-04
1,2,3,7,8,9-HXCDD	Walleye	3	3	100	2.20E-04	2.90E-04	2.53E-04	Normal	3.13E-04	0.0	2.90E-04
1,2,3,7,8,9-HXCDF	Walleye	3	0								
1,2,3,7,8-PECDD	Walleye	3	3	100	6.30E-04	8.60E-04	7.30E-04	Normal	9.29E-04	0.0	8.60E-04
1,2,3,7,8-PECDF	Walleye	3	3	100	7.00E-04	1.00E-03	8.37E-04	Normal	1.09E-03	0.0	1.00E-03
2,3,4,6,7,8-HXCDF	Walleye	3	3	100	2.40E-04	4.60E-04	3.83E-04	Normal	5.93E-04	0.0	4.60E-04
2,3,4,7,8-PECDF	Walleye	3	3	100	5.20E-04	7.40E-04	6.40E-04	Normal	8.28E-04	0.0	7.40E-04
2,3,7,8-TCDD	Walleye	2	2	100	3.50E-04	5.40E-04	4.45E-04	Normal	1.04E-03	0.0	5.40E-04
2,3,7,8-TCDF	Walleye	3	3	100	4.20E-03	6.20E-03	5.43E-03	Normal	7.25E-03	0.0	6.20E-03

Table 6-15 Dioxin and PCB Congener Concentrations in Little Lake Butte des Morts Reach Whole Fish (Continued)

Analyte	Species	Number of Samples	Number of Detects	Detection Frequency (%)	Detected Minimum	Detected Maximum	Mean	Data Distribution	95% UCL	90th Percentile	RME
PCBs (µg/kg)											
PCB Congener 77	Golden Shiner	2	2	100	2.2	2.2	2.2	Normal	2.4	0.0	2.2
PCB Congener 81/87/115	Golden Shiner	2	2	100	8.8	14.0	11.4	Normal	27.8	0.0	14.0
PCB Congener 105	Golden Shiner	2	2	100	4.6	4.7	4.7	Normal	4.9	0.0	4.7
PCB Congener 118	Golden Shiner	2	2	100	17.4	25.0	21.2	Normal	45.2	0.0	25.0
PCB Congener 126	Golden Shiner	2	1	50	0.04	0.04	0.03	Normal	0.1	0.0	0.04
PCB Congener 169	Golden Shiner	2	0								
PCB Congener 77	Yellow Perch	1	1	100	0.1	NA	—			0.0	0.1
PCB Congener 81	Yellow Perch	1	0								
PCB Congener 105	Yellow Perch	1	1	100	2.0	NA	—			0.0	2.0
PCB Congener 118	Yellow Perch	1	1	100	8.6	NA	—			0.0	8.6
PCB Congener 126	Yellow Perch	1	0								
PCB Congener 169	Yellow Perch	1	0								
PCB Congener 77	Carp	7	7	100	0.1	8.5	1.9	Lognormal	50.7	0.0	8.5
PCB Congener 81	Carp	2	0								
PCB Congener 105	Carp	7	6	86	2.9	35.0	8.6	Lognormal	8,700	0.0	35.0
PCB Congener 118	Carp	7	7	100	6.6	150	35.4	Lognormal	183	0.0	150
PCB Congener 126	Carp	7	1	14	0.03	0.03	NE	Lognormal	5.2	0.0	0.03
PCB Congener 169	Carp	6	0								
PCB Congener 77	Walleye	7	6	86	0.1	6.4	2.1	Normal	3.7	0.0	3.7
PCB Congener 81	Walleye	3	0								
PCB Congener 105	Walleye	7	7	100	0.6	20.0	9.3	Normal	14.0	0.0	14.0
PCB Congener 118	Walleye	7	7	100	3.9	77.0	36.8	Normal	55.1	0.0	55.1
PCB Congener 126	Walleye	7	4	57	0.1	1.1	0.2	Lognormal	79.0	0.0	1.1
PCB Congener 169	Walleye	7	1	14	0.1	0.1	0.1	Lognormal	15.7	0.0	0.1

Note:

NA - Not applicable.

NE indicates that the calculated mean exceeded the detected maximum and, therefore, the mean was not evaluated.

Table 6-16 Little Lake Butte des Morts Reach Bird Tissue Concentrations

Analyte	Species	Tissue	Number of Samples	Number of Detects	Detection Frequency (%)	Detected Minimum	Detected Maximum	Mean	Data Distribution	95% UCL	90th Percentile	RME
PCBs (µg/kg)												
Total PCBs	Tree Swallow	egg	5	5	100	1,790	4,030	2,924	Normal	3,732	0.0	3,732
Total PCBs	Tree Swallow	whole body	24	24	100	79.0	7,400	2,135	Lognormal	5,254	5,300	5,254
Pesticides (µg/kg)												
Dieldrin	Tree Swallow	whole body	18	0								
o,p'-DDD	Tree Swallow	whole body	18	0								
o,p'-DDE	Tree Swallow	whole body	18	0								
o,p'-DDT	Tree Swallow	whole body	18	0								
p,p'-DDD	Tree Swallow	whole body	18	0								
p,p'-DDE	Tree Swallow	whole body	18	18	100	38.0	530	155	Lognormal	239	359	239
p,p'-DDT	Tree Swallow	whole body	18	0								

Table 6-17 Estimated Exposure Concentrations for Piscivorous Birds in Little Lake Butte des Morts Reach

Analyte	Surface Water (µg/L) [1]		TL3 Fish: Yellow Perch and Gizzard Shad (µg/kg)		Common Tern Ingestion				Total Common Tern Ingestion (µg/day)		Total Common Tern Ingestion (µg/kg-BW/day)	
					Surface Water (µg/day) [1]		TL3 Fish (µg/day)					
	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME
Mercury	2.2	7.1	25.0	25.0	0.03	0.10	1.5	1.5	1.5	1.6	12.5	13.1
Total PCBs	0.03	0.06	296	530	0.0004	0.001	17.4	31.2	17.4	31.2	145	260
Dieldrin	NA	NA	1.3	1.3	—	—	0.1	0.1	0.1	0.1	0.6	0.6
p,p'-DDE	NA	NA	9.5	11.0	—	—	0.6	0.6	0.6	0.6	4.7	5.4

Analyte	Surface Water (µg/L) [1]		TL3 Fish: Yellow Perch and Gizzard Shad (µg/kg)		Forster's Tern Ingestion				Total Forster's Tern Ingestion (µg/day)		Total Forster's Tern Ingestion (µg/kg-BW/day)	
					Surface Water (µg/day) [1]		TL3 Fish (µg/day)					
	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME
Mercury	2.2	7.1	25.0	25.0	0.04	0.12	1.8	1.8	1.8	1.9	11.5	12.1
Total PCBs	0.03	0.06	296	530	0.0005	0.001	21.1	37.8	21.1	37.8	134	240
Dieldrin	NA	NA	1.3	1.3	—	—	0.1	0.1	0.1	0.1	0.6	0.6
p,p'-DDE	NA	NA	9.5	11.0	—	—	0.7	0.8	0.7	0.8	4.3	5.0

Analyte	Surface Water (µg/L) [1]		TL3 Fish: Yellow Perch and Gizzard Shad (µg/kg)		Double-crested Cormorant Ingestion				Double-crested Cormorant Ingestion (µg/day)		Total Double-crested Cormorant Ingestion (µg/kg-BW/day)	
					Surface Water (µg/day) [1]		TL3 Fish (µg/day)					
	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME
Mercury	2.2	7.1	25.0	25.0	0.2	0.6	8.0	8.0	8.1	8.5	4.8	5.1
Total PCBs	0.03	0.06	296	530	0.002	0.005	94.1	169	94.1	169	56.0	100
Dieldrin	NA	NA	1.3	1.3	—	—	0.4	0.4	0.4	0.4	0.2	0.2
p,p'-DDE	NA	NA	9.5	11.0	—	—	3.0	3.5	3.0	3.5	1.8	2.1

Notes:

[1] Mercury in unfiltered water, total PCBs on a filtered + particulate basis.

NA indicates that data was not collected for this reach and this contaminant.

Shaded areas indicate that the concentration is estimated as half of the detection limit.

"—" indicates that the concentration could not be calculated.

Table 6-17 Estimated Exposure Concentrations for Piscivorous Birds in Little Lake Butte des Morts Reach (Continued)

Analyte	Surface Water (µg/L) [1]		TL3 Fish: Carp (µg/kg)		TL4 Fish: Walleye (µg/kg)		Bald Eagle Ingestion						Total Bald Eagle Ingestion (µg/day)		Total Bald Eagle Ingestion (µg/kg-BW/day)	
							Surface Water (µg/day) [1]		TL3 Fish (µg/day)		TL4 Fish (µg/day)					
	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME
Mercury	2.2	7.1	50.0	60.0	50.0	30.0	0.4	1.2	21.1	25.3	5.3	3.2	26.7	29.6	5.7	6.4
Total PCBs	0.03	0.06	1,992	2,957	1,159	3,800	0.005	0.01	841	1,248	122	399	962	1,647	207	354
Dieldrin	NA	NA	3.0	1.0	5.2	46.5	—	—	1.3	0.4	0.5	4.9	1.8	5.3	0.4	1.1
p,p'-DDE	NA	NA	16.9	23.8	47.6	61.7	—	—	7.1	10.1	5.0	6.5	12.1	16.5	2.6	3.6

Notes:

[1] Mercury in unfiltered water, total PCBs on a filtered + particulate basis.

NA indicates that data was not collected for this reach and this contaminant.

Shaded areas indicate that the concentration is estimated as half of the detection limit.

"—" indicates that the concentration could not be calculated.

Model Assumptions:

Common Tern

Food Ingestion =	0.0588	kg/day
Water Ingestion =	0.014	L/day
Body Weight =	0.12	kg

Forster's Tern

Food Ingestion =	0.0714	kg/day
Water Ingestion =	0.017	L/day
Body Weight =	0.158	kg

Double-crested Cormorant

Food Ingestion =	0.318	kg/day
Water Ingestion =	0.084	L/day
Body Weight =	1.68	kg

Bald Eagle

Food Ing. (TL3 Fish) =	0.422	kg/day
Food Ing. (TL4 Fish) =	0.105	kg/day
Water Ingestion =	0.165	L/day
Body Weight =	4.65	kg

Table 6-18 PCB Congeners in Tree Swallows from Little Lake Butte des Morts Reach

Analyte	Tissue	Number of Samples	Number of Detects	Detection Frequency (%)	Detected Minimum	Detected Maximum	Mean	Data Distribution	95% UCL	90th Percentile	RME
PCBs (µg/kg)											
PCB Congener 77	egg	5	5	100	0.1	0.5	0.2	Normal	0.4	0.0	0.4
PCB Congener 105	egg	5	5	100	1.4	36.0	20.7	Normal	32.8	0.0	32.8
PCB Congener 118/106	egg	5	5	100	56.0	120	85.2	Normal	108	0.0	108
PCB Congener 126	egg	5	5	100	0.2	0.7	0.3	Lognormal	0.8	0.0	0.7
PCB Congener 169	egg	5	1	20	0.2	0.2	0.1	Other	0.2	0.0	0.2
PCBs (µg/kg)											
PCB Congener 77	whole	15	0								
PCB Congener 105	whole	15	15	100	1.7	50.0	16.7	Lognormal	37.1	44.6	37.1
PCB Congener 118/106	whole	15	15	100	6.5	150	58.4	Lognormal	129	144	129
PCB Congener 126	whole	15	6	40	0.1	0.4	0.1	Other	0.2	0.3	0.2
PCB Congener 169	whole	15	0								

Table 6-19 Estimated Exposure Concentrations for Mink in Little Lake Butte des Morts Reach

Analyte	Surface Water (µg/L) [1]		Surface Sediment (µg/kg) [2]		Whole Carp (µg/kg)		Mink Ingestion						Total Mink Ingestion (µg/day)		Total Mink Ingestion (µg/kg-BW/day)	
							Surface Water (µg/day) [1]		Surface Sediment (µg/day)		Whole Carp (µg/day)					
	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME
Mercury	2.2	7.1	1,000	1,400	50.0	60.0	0.2	0.6	4.0	5.6	7.7	9.2	11.8	15.4	14.8	19.2
Total PCBs (N)	0.03	0.06	10,724	22,848	1,992	2,957	0.002	0.004	42.9	91.4	305	452	348	544	435	680
Total PCBs (I_0)	0.03	0.06	3,284	3,330	1,992	2,957	0.002	0.004	13.1	13.3	305	452	318	466	397	582
Total PCBs (I_d)	0.03	0.06	3,699	3,749	1,992	2,957	0.002	0.004	14.8	15.0	305	452	320	467	400	584
Dieldrin	NA	NA	NE	5.9	3.0	1.0	—	—	—	0.02	0.5	0.2	0.5	0.2	0.6	0.2
p,p'-DDE	NA	NA	NE	50.0	16.9	23.8	—	—	—	0.2	2.6	3.6	2.6	3.8	3.2	4.8

Notes:

[1] Mercury in unfiltered water, total PCBs on a filtered + particulate basis.

[2] p,p'-DDT rather than p,p'-DDE was used because this was the predominant form in the sediment.

NE indicates that the calculated mean exceeded the detected maximum and, therefore, the mean was not evaluated.

NA indicates that data was not collected for this reach and this contaminant.

"—" indicates that the concentration could not be calculated.

Model Assumptions:

Food Ingestion =	0.153	kg/day
Water Ingestion =	0.081	L/day
Sediment Ingestion =	0.004	kg/day
Body Weight =	0.8	kg

Table 6-20 Surface Water Concentrations in Appleton to Little Rapids Reach

Analyte	Number of Samples	Number of Detects	Detection Frequency (%)	Detected Minimum	Detected Maximum	Mean	Data Distribution	95% UCL	90th Percentile	RME	Criteria NOAEC	Criteria LOAEC	Reference	Hazard Quotients Mean NOEL	Hazard Quotients Mean LOEL	Hazard Quotients RME NOEL	Hazard Quotients RME LOEL
Metals (ng/L)																	
Arsenic (unfiltered)	3	0															
Lead (unfiltered)	3	3	100	900	1,800	1,397	Normal	2,167	0.0	1,800	49,420		[1]	< 0.1		< 0.1	
Mercury (filtered)	2	1	50	90.0	90.0	65.0	Normal	223	0.0	90.0	440		[1]	0.1		0.2	
Mercury (unfiltered)	5	2	40	47.0	90.0	66.4	Normal	119	0.0	90.0	440		[1]	0.2		0.2	
Dioxins/Furans (ng/L)																	
2,3,7,8-TCDF (unfiltered)	1	0															
PCBs (ng/L)																	
Total PCBs (filtered)	85	84	99	0.03	18.9	4.8	Lognormal	9.45	13.5	9.4	50	500	[2]	0.1	0.1	0.2	< 0.1
Total PCBs (unfiltered)	1	0															
Total PCBs (particulate)	86	82	95	0.01	52.2	11.9	Other	60.15	33.8	52.2	50	500	[2]	0.2	0.2	1.0	0.1
Total PCBs (filtered + particulate)				0.04	71.0	16.8		69.60	47.3	61.6	50	500	[2]	0.3	<0.1	1.2	0.1
Pesticides (ng/L)																	
Dieldrin (unfiltered)	3	0															
p,p'-DDD (unfiltered)	3	0															
p,p'-DDE (unfiltered)	3	0															
p,p'-DDT (unfiltered)	3	0															

Notes:

[1] Wisconsin Administrative Code Chapter NR 105.

[2] Niimi, 1996.

Table 6-21 Surface Sediment Concentrations in Appleton to Little Rapids Reach

Analyte	Number of Samples	Number of Detects	Detection Frequency (%)	Detected Minimum	Detected Maximum	Mean	Data Distribution	95% UCL	90th Percentile	RME	Criteria	Reference	Hazard Quotients Mean	Hazard Quotients RME
Metals (mg/kg)														
Arsenic	10	6	60	2.8	9.7	4.4	Lognormal	6.4	9.5	6.4	12.1	[1]	0.4	0.5
Lead	10	10	100	44.0	130	75.6	Normal	88.9	126	88.9	34.2	[1]	2.2	2.6
Mercury	10	10	100	0.2	2.1	0.8	Lognormal	1.7	2.0	1.7	0.17	[2]	4.5	10
PCBs (µg/kg)														
Total PCBs (N)	131	122	93	35.0	74,200	6,751	Lognormal	15,267	25,360	15,267	31.6	[1]	214	483
Total PCBs (I_0)	72,865	72,865	100	0.0	63,377	175	Other	185	100	185	31.6	[1]	5.5	5.9
Total PCBs (I_d)	9,096	9,096	100	21.4	63,377	1,398	Other	1,479	2,700	1,479	31.6	[1]	44	47
Pesticides (µg/kg)														
Dieldrin	10	0												
p,p'-DDD	10	2	20	1.0	1.7	NE	Lognormal	46.6	44.1	1.7	3.54	[2]	NE	0.5
p,p'-DDE	10	0												
p,p'-DDT	10	1	10	3.4	3.4	NE	Lognormal	37.5	44.1	3.4	7.00	[2]	NE	0.5

Notes:

N indicates that the data was not interpolated based on depth.

I_0 indicates that interpolated grid areas for which "no values" existed were assumed to equal zero.

I_d indicates that interpolated grid areas for which "no values" existed were deleted from the database.

[1] ARCS SEC (EPA, 1996a).

[2] Environment Canada TEL (Smith *et al.,* 1996); p,p'-DDT TEL based on total DDT TEL.

NE indicates that the calculated mean exceeded the detected maximum and, therefore, the mean was not evaluated.

Table 6-22 PCB Congener Concentrations in Surface Sediment in Appleton to Little Rapids Reach

Analyte	Number of Samples	Number of Detects	Detection Frequency (%)	Detected Minimum	Detected Maximum	Mean	Data Distribution	95% UCL	90th Percentile	RME
PCBs (µg/kg)										
PCB Congener 77	9	5	56	0.8	35.0	6.5	Lognormal	104	0.0	35.0
PCB Congener 81	9	5	56	0.02	0.4	0.3	Normal	0.5	0.0	0.4
PCB Congener 105	13	9	69	0.4	140	15.2	Lognormal	138	94.6	138
PCB Congener 118	21	21	100	0.9	590	54.2	Lognormal	181	221	181
PCB Congener 126	9	2	22	0.1	0.1	NE	Other	3.6	0.0	0.1
PCB Congener 169	13	0								

Note:

NE indicates that the calculated mean exceeded the detected maximum and, therefore, the mean was not evaluated.

Table 6-23 Appleton to Little Rapids Reach Whole Fish Concentrations

Analyte	Species	Number of Samples	Number of Detects	Detection Frequency (%)	Detected Minimum	Detected Maximum	Mean	Data Distribution	95% UCL	90th Percentile	RME
Metals (mg/kg)											
Mercury	Yellow Perch	4	0								
	Carp	5	1	20	0.12	0.12	0.06	Other	0.11	0.0	0.11
	Walleye	3	2	67	0.18	0.20	0.14	Normal	0.28	0.0	0.20
PCBs (µg/kg)											
Total PCBs	Yellow Perch	4	4	100	425	1,298	779	Normal	1,219	0.0	1,219
	Carp	12	12	100	160	6,600	2,581	Normal	3,606	6,270	3,606
	Walleye	4	4	100	1,431	3,900	2,737	Normal	4,061	0.0	3,900
Pesticides (µg/kg)											
Dieldrin	Yellow Perch	1	0								
	Carp	4	0								
	Walleye	3	0								
o,p'-DDD	Carp	2	0								
o,p'-DDT	Carp	2	0								
p,p'-DDD	Yellow Perch	1	0								
	Carp	6	0								
	Walleye	3	1	33	8.0	8.0	7.5	Normal	9.7	0.0	8.0
p,p'-DDE	Yellow Perch	1	1	100	10.0	10.0	NA			0.0	10.0
	Carp	6	4	67	9.0	89.0	47.8	Normal	75.2	0.0	75.2
	Walleye	3	3	100	53.0	65.0	57.0	Other	75.2	0.0	65.0
p,p'-DDT	Yellow Perch	1	0								
	Carp	6	0								
	Walleye	3	0								

Note:
NA - Not applicable.

Table 6-24 PCB Congener Concentrations in Appleton to Little Rapids Reach Whole Fish

Analyte	Species	Number of Samples	Number of Detects	Detection Frequency (%)	Detected Minimum	Detected Maximum	Mean	Data Distribution	95% UCL	95th Percentile	RME
PCBs (µg/kg)											
PCB Congener 77	Yellow Perch	4	4	100	0.1	1.8	0.6	Lognormal	850	0.0	1.8
PCB Congener 81	Yellow Perch	4	1	25	0.4	0.4	0.1	Other	6.2E+09	0.0	0.4
PCB Congener 105	Yellow Perch	4	4	100	1.8	16.0	5.9	Lognormal	932	0.0	16.0
PCB Congener 118	Yellow Perch	4	4	100	11.0	48.0	23.3	Normal	42.9	0.0	42.9
PCB Congener 126	Yellow Perch	4	2	50	0.02	0.04	0.02	Normal	0.0	0.0	0.0
PCB Congener 169	Yellow Perch	4	0								
PCB Congener 77	Carp	5	4	80	0.2	1.7	0.7	Normal	1.5	0.0	1.5
PCB Congener 81	Carp	5	0								
PCB Congener 105	Carp	5	5	100	0.7	36.0	18.6	Normal	34.4	0.0	34.4
PCB Congener 118	Carp	5	5	100	4.4	98.0	56.3	Normal	100.5	0.0	98.0
PCB Congener 126	Carp	5	3	60	0.04	0.8	0.2	Lognormal	121,161	0.0	0.8
PCB Congener 169	Carp	5	2	40	0.1	0.1	0.04	Other	15.6	0.0	0.1
PCB Congener 77	Walleye	3	3	100	0.4	4.5	3.0	Normal	6.9	0.0	4.5
PCB Congener 81	Walleye	3	0								
PCB Congener 105	Walleye	3	3	100	13.0	20.0	16.3	Normal	22.3	0.0	20.0
PCB Congener 118	Walleye	3	3	100	59.0	110	80.3	Normal	125.0	0.0	110
PCB Congener 126	Walleye	3	2	67	0.3	0.3	0.2	Normal	0.5	0.0	0.3
PCB Congener 169	Walleye	3	0								

Table 6-25 Appleton to Little Rapids Reach Bird Tissue Concentrations

Analyte	Species	Tissue	Number of Samples	Number of Detects	Detection Frequency (%)	Detected Minimum	Detected Maximum	Mean	Data Distribution	95% UCL	90th Percentile	RME
Metals (mg/kg)												
Mercury	Bald Eagle	liver	1	1	100	1.4	1.4	—	—	—	0.0	1.4
PCBs (µg/kg)												
Total PCBs	Bald Eagle	egg	1	1	100	36,000	36,000	—	—	—	0.0	36,000
Pesticides (µg/kg)												
Dieldrin	Bald Eagle	egg	1	1	100	70.0	70.0	—	—	—	0.0	70.0
p,p'-DDD	Bald Eagle	egg	1	1	100	160	160	—	—	—	0.0	160
p,p'-DDE	Bald Eagle	egg	1	1	100	1,100	1,100	—	—	—	0.0	1,100
p,p'-DDT	Bald Eagle	egg	1	0								

Table 6-26 Estimated Exposure Concentrations for Piscivorous Birds in Appleton to Little Rapids Reach

Analyte	Surface Water (µg/L) [1]		TL3 Fish: Yellow Perch (µg/kg)		Common Tern Ingestion				Total Common Tern Ingestion (µg/day)		Total Common Tern Ingestion (µg/kg-BW/day)	
					Surface Water (µg/day) [1]		TL3 Fish (µg/day)					
	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME
Mercury	0.07	0.09	25.0	25.0	0.001	0.001	1.5	1.5	1.5	1.5	12.3	12.3
Total PCBs	0.02	0.06	779	1,219	0.0002	0.001	45.8	71.7	45.8	71.7	382	597
Dieldrin	0.001	0.002	1.3	2.5	0.00001	0.00002	0.1	0.1	0.1	0.1	0.6	1.2
p,p'-DDE	0.001	0.002	10.0	10.0	0.00002	0.00002	0.6	0.6	0.6	0.6	4.9	4.9

Analyte	Surface Water (µg/L) [1]		TL3 Fish: Yellow Perch (µg/kg)		Forster's Tern Ingestion				Total Forster's Tern Ingestion (µg/day)		Total Forster's Tern Ingestion (µg/kg-BW/day)	
					Surface Water (µg/day) [1]		TL3 Fish (µg/day)					
	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME
Mercury	0.07	0.09	25.0	25.0	0.001	0.002	1.8	1.8	1.8	1.8	11.3	11.3
Total PCBs	0.02	0.06	779	1,219	0.0003	0.001	55.6	87.0	55.6	87.0	352	551
Dieldrin	0.001	0.002	1.3	2.5	0.00002	0.00003	0.1	0.2	0.1	0.2	0.6	1.1
p,p'-DDE	0.001	0.002	10.0	10.0	0.00002	0.00003	0.7	0.7	0.7	0.7	4.5	4.5

Analyte	Surface Water (µg/L) [1]		TL3 Fish: Yellow Perch (µg/kg)		Double-crested Cormorant Ingestion				Double-crested Cormorant Ingestion (µg/day)		Total Double-crested Cormorant Ingestion (µg/kg-BW/day)	
					Surface Water (µg/day) [1]		TL3 Fish (µg/day)					
	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME
Mercury	0.07	0.09	25.0	25.0	0.01	0.01	8.0	8.0	8.0	8.0	4.7	4.7
Total PCBs	0.02	0.06	779	1,219	0.001	0.01	248	388	248	388	148	231
Dieldrin	0.001	0.002	1.3	2.5	0.0001	0.0001	0.4	0.8	0.4	0.8	0.2	0.5
p,p'-DDE	0.001	0.002	10.0	10.0	0.0001	0.0001	3.2	3.2	3.2	3.2	1.9	1.9

Notes:

[1] Mercury in unfiltered water, total PCBs on a filtered + particulate basis.

Shaded areas indicate that the concentration is estimated as half of the detection limit.

Table 6-26 Estimated Exposure Concentrations for Piscivorous Birds in Appleton to Little Rapids Reach (Continued)

Analyte	Surface Water (µg/L) [1]		TL3 Fish: Carp (µg/kg)		TL4 Fish: Walleye (µg/kg)		Bald Eagle Ingestion						Total Bald Eagle Ingestion (µg/day)		Total Bald Eagle Ingestion (µg/kg-BW/day)	
							Surface Water (µg/day) [1]		TL3 Fish (µg/day)		TL4 Fish (µg/day)					
	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME
Mercury	0.07	0.09	60.0	110	140	200	0.01	0.01	25.3	46.4	14.7	21.0	40.0	67.4	8.6	14.5
Total PCBs	0.02	0.06	2,581	3,606	2,737	3,900	0.003	0.01	1,089	1,522	287	410	1,376	1,931	296	415
Dieldrin	**0.001**	**0.002**	**2.0**	**2.7**	**2.3**	**4.5**	0.0001	0.0002	0.8	1.1	0.2	0.5	1.1	1.6	0.2	0.3
p,p'-DDE	**0.001**	**0.002**	47.8	75.2	57.0	65.0	0.0002	0.0002	20.2	31.8	6.0	6.8	26.2	38.6	5.6	8.3

Notes:

[1] Mercury in unfiltered water, total PCBs on a filtered + particulate basis.

Shaded areas indicate that the concentration is estimated as half of the detection limit.

Model Assumptions:

Common Tern

Food Ingestion =	0.0588	kg/day
Water Ingestion =	0.014	L/day
Body Weight =	0.12	kg

Forster's Tern

Food Ingestion =	0.0714	kg/day
Water Ingestion =	0.017	L/day
Body Weight =	0.158	kg

Double-crested Cormorant

Food Ingestion =	0.318	kg/day
Water Ingestion =	0.084	L/day
Body Weight =	1.68	kg

Bald Eagle

Food Ing. (TL3 Fish) =	0.422	kg/day
Food Ing. (TL4 Fish) =	0.105	kg/day
Water Ingestion =	0.165	L/day
Body Weight =	4.65	kg

Table 6-27 Estimated Exposure Concentrations for Mink in Appleton to Little Rapids Reach

Analyte	Surface Water (µg/L) [1]		Surface Sediment (µg/kg) [2]		Whole Carp (µg/kg)		Mink Ingestion						Total Mink Ingestion (µg/day)		Total Mink Ingestion (µg/kg-BW/day)	
							Surface Water (µg/day) [1]		Surface Sediment (µg/day)		Whole Carp (µg/day)					
	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME
Mercury	0.07	0.09	800	1,700	60	110	0.01	0.01	3.2	6.8	9.2	16.8	12.4	23.6	15.5	29.5
Total PCBs (N)	0.02	0.06	6,751	15,267	2,581	3,606	0.001	0.005	27.0	61.1	395	552	422	613	527	766
Total PCBs (I_0)	0.02	0.06	175	185	2,581	3,606	0.001	0.005	0.7	0.7	395	552	396	553	494	691
Total PCBs (I_d)	0.02	0.06	1,398	1,479	2,581	3,606	0.001	0.005	5.6	5.9	395	552	400	558	501	697
Dieldrin	0.0009	0.0015	4.4	28.3	2.0	2.7	0.0001	0.0001	0.02	0.1	0.3	0.4	0.3	0.5	0.4	0.7
p,p'-DDE	0.0011	0.0015	NE	3.4	47.8	75.2	0.0001	0.0001	—	0.01	7.3	11.5	7.3	11.5	9.1	14.4

Notes:

[1] Mercury in unfiltered water, total PCBs on a filtered + particulate basis.

[2] p,p'-DDT rather than p,p'-DDE was used because this was the predominant form in the sediment.

NE indicates that the calculated mean exceeded the detected maximum and, therefore, the mean was not evaluated.

NA indicates that data was not collected for this reach and this contaminant.

Shaded areas indicate that the concentration is estimated as half of the detection limit.

"—" indicates that the concentration could not be calculated.

Model Assumptions:

Food Ingestion =	0.153	kg/day
Water Ingestion =	0.081	L/day
Sediment Ingestion =	0.004	kg/day
Body Weight =	0.8	kg

Table 6-28 Surface Water Concentrations in Little Rapids to De Pere Reach

Analyte	Number of Samples	Number of Detects	Detection Frequency (%)	Detected Minimum	Detected Maximum	Mean	Data Distribution	95% UCL	90th Percentile	RME	Criteria NOAEC	Criteria LOAEC	Reference	Hazard Quotients Mean NOEL	Hazard Quotients Mean LOEL	Hazard Quotients RME NOEL	Hazard Quotients RME LOEL
Metals (ng/L)																	
Lead (filtered)	2	2	100	118	124	121	Normal	140	0.0	124	49,420		[1]	< 0.1		< 0.1	
Lead (unfiltered)	2	2	100	526	707	617	Normal	1,188	0.0	707	49,420		[1]	< 0.1		< 0.1	
Mercury (filtered)	3	2	67	1,260	2,520	1,273	Normal	3,364	0.0	2,520	440		[1]	2.9		5.7	
Mercury (unfiltered)	3	2	67	4,490	7,120	3,883	Normal	9,917	0.0	7,120	440		[1]	8.8		16	
PCBs (ng/L)																	
Total PCBs (filtered)	98	97	99	0.2	27.6	11.3	Normal	12.3	18.2	12.3	50	500	[2]	0.2	< 0.1	0.2	< 0.1
Total PCBs (particulate)	98	94	96	0.2	96.3	29.9	Normal	33.3	63.4	33.3	50	500	[2]	0.6	0.1	0.7	0.1
Total PCBs (filtered + particulate)				0.4	124	41.1		45.5	81.7	45.5	50	500	[2]	0.8	0.1	0.9	0.1

Notes:

[1] Wisconsin Administrative Code Chapter NR 105.

[2] Niimi, 1996.

Table 6-29 Surface Sediment Concentrations in Little Rapids to De Pere Reach

Analyte	Number of Samples	Number of Detects	Detection Frequency (%)	Detected Minimum	Detected Maximum	Mean	Data Distribution	95% UCL	90th Percentile	RME	Criteria	Reference	Hazard Quotients	
													Mean	RME
Metals (mg/kg)														
Arsenic	20	18	90	3.0	7.6	4.6	Normal	5.1	6.6	5.1	12.1	[1]	0.4	0.4
Lead	20	20	100	6.2	1,400	159	Other	274	282	274	34.2	[1]	4.6	8.0
Mercury	74	74	100	0.01	9.8	3.5	Normal	4.0	8.1	4.0	0.17	[2]	21	24
Dioxins/Furans (µg/kg)														
2,3,7,8-TCDD	2	2	100	3.7E-03	6.8E-03	5.3E-03	Normal	1.5E-02	0.0	6.8E-03	3.90E-03	[3]	1.3	1.7
2,3,7,8-TCDF	2	2	100	4.6E-02	1.2E-01	8.1E-02	Normal	3.1E-01	0.0	1.2E-01				
PCBs (µg/kg)														
Total PCBs (N)	209	203	97	37.0	40,430	4,782	Lognormal	10,543	15,000	10,543	31.6	[1]	151	334
Total PCBs (I_0)	37,490	37,490	100	0.0	40,429	2,054	Other	2,088	6,049	2,088	31.6	[1]	65	66
Total PCBs (I_d)	37,060	37,060	100	37.1	40,429	2,078	Other	2,112	6,133	2,112	31.6	[1]	66	67
Pesticides (µg/kg)														
Dieldrin	19	0												
p,p'-DDD	20	5	25	1.5	2.8	NE	Other	21.7	50.8	2.8	3.54	[2]	NE	0.8
p,p'-DDE	19	4	21	6.6	22.0	12.5	Lognormal	34.7	55.0	22.0	1.42	[2]	8.8	15
p,p'-DDT	14	3	21	5.1	20.0	16.5	Other	27.7	55.0	20.0	7.00	[2]	2.4	2.9

Notes:

NA indicated that the criteria is not available.

N indicates that the data was not interpolated based on depth.

I_0 indicates that interpolated grid areas for which "no values" existed were assumed to equal zero.

I_d indicates that interpolated grid areas for which "no values" existed were deleted from the database.

[1] ARCS SEC (EPA, 1996a).

[2] Environment Canada TEL (Smith *et al.,* 1996); p,p'-DDT TEL based on total DDT TEL.

[3] EPA Sediment Screening Values (EPA, 1997b).

NE indicates that the calculated mean exceeded the detected maximum and, therefore, the mean was not evaluated.

Table 6-30 PCB Congener Concentrations in Surface Sediment in Little Rapids to De Pere Reach

Analyte	Number of Samples	Number of Detects	Detection Frequency (%)	Detected Minimum	Detected Maximum	Mean	Data Distribution	95% UCL	90th Percentile	RME
PCBs (µg/kg)										
PCB Congener 77	23	15	65	2.4	89.1	14.7	Other	57.9	55.4	57.9
PCB Congener 81	22	10	45	0.1	2.4	0.7	Other	2.0	1.5	2.0
PCB Congener 105	23	20	87	0.9	54.4	10.8	Lognormal	21.4	51.6	21.4
PCB Congener 118	40	39	98	3.0	190	33.4	Lognormal	58.4	116	58.4
PCB Congener 126	23	5	22	0.03	0.8	0.6	Other	3.3	1.3	0.8
PCB Congener 169	23	0								

Table 6-31 Little Rapids to De Pere Reach Whole Fish Concentrations

Analyte	Species	Number of Samples	Number of Detects	Detection Frequency (%)	Detected Minimum	Detected Maximum	Mean	Data Distribution	95% UCL	90th Percentile	RME
Metals (mg/kg)											
Mercury	Yellow Perch	1	0								
	Carp	1	1	100	0.15	0.15	NA			0.0	0.15
	Walleye	1	1	100	0.16	0.16	NA			0.0	0.16
PCBs (µg/kg)											
Total PCBs	Gizzard Shad	3	3	100	310	370	347	Normal	401	0.0	370
	Golden Shiner	2	2	100	1,003	1,036	1,020	Normal	1,124	0.0	1,036
	Yellow Perch	1	1	100	627	627	NA			0.0	627
	Carp	20	20	100	604	6,000	3,919	Other	5,800	5,980	5,800
	Walleye	4	4	100	1,490	4,587	3,179	Normal	4,918	0.0	4,587
Pesticides (µg/kg)											
Dieldrin	Yellow Perch	1	0								
	Carp	4	0								
	Walleye	4	1	25	5.8	5.8	3.4	Normal	5.4	0.0	5.4
o,p'-DDD	Carp	4	0								
	Walleye	3	0								
o,p'-DDE	Carp	4	0								
	Walleye	3	3	100	38.0	61.0	45.7	Other	102	0.0	61.0
o,p'-DDT	Carp	4	0								
	Walleye	3	0								
p,p'-DDD	Yellow Perch	1	0								
	Carp	5	3	60	1.6	8.0	NE	Normal	19.0	0.0	8.0
	Walleye	4	0								
p,p'-DDE	Yellow Perch	1	1	100	16.0	16.0	NA			0.0	16.0
	Carp	5	5	100	13.0	140	74.2	Normal	128	0.0	128
	Walleye	4	4	100	75.0	220	129	Normal	208	0.0	208
p,p'-DDT	Yellow Perch	1	0								
	Carp	5	0								
	Walleye	4	0								

Notes:

NA - Not applicable.

NE indicates that the calculated mean exceeded the detected maximum and, therefore, the mean was not evaluated.

Table 6-32 Dioxin and PCB Congener Concentrations in Little Rapids to De Pere Reach Whole Fish

Analyte	Species	Number of Samples	Number of Detects	Detection Frequency (%)	Detected Minimum	Detected Maximum	Mean	Data Distribution	95% UCL	90th Percentile	RME
Dioxins/Furans (µg/kg)											
1,2,3,4,6,7,8,9-OCDD	Carp	3	3	100	1.33E-02	5.47E-02	3.25E-02	Normal	6.77E-02	0.0	5.47E-02
1,2,3,4,6,7,8,9-OCDF	Carp	3	1	33	2.30E-03	2.30E-03	1.65E-03	Normal	2.89E-03	0.0	2.30E-03
1,2,3,4,6,7,8-HPCDD	Carp	3	3	100	5.40E-03	1.11E-02	8.13E-03	Normal	1.30E-02	0.0	1.11E-02
1,2,3,4,6,7,8-HPCDF	Carp	3	3	100	9.30E-04	2.00E-03	1.41E-03	Normal	2.33E-03	0.0	2.00E-03
1,2,3,4,7,8,9-HPCDF	Carp	3	1	33	6.90E-04	6.90E-04	3.13E-04	Normal	8.78E-04	0.0	6.90E-04
1,2,3,4,7,8-HXCDD	Carp	3	3	100	8.40E-04	2.10E-03	1.26E-03	Other	2.39E-02	0.0	2.10E-03
1,2,3,4,7,8-HXCDF	Carp	3	3	100	3.40E-04	8.50E-04	5.80E-04	Normal	1.01E-03	0.0	8.50E-04
1,2,3,6,7,8-HXCDD	Carp	3	3	100	1.60E-03	5.30E-03	2.93E-03	Normal	6.40E-03	0.0	5.30E-03
1,2,3,6,7,8-HXCDF	Carp	3	3	100	4.60E-04	1.60E-03	9.03E-04	Normal	1.93E-03	0.0	1.60E-03
1,2,3,7,8,9-HXCDD	Carp	3	3	100	1.70E-04	5.50E-04	4.20E-04	Normal	7.85E-04	0.0	5.50E-04
1,2,3,7,8,9-HXCDF	Carp	3	1	33	6.40E-04	6.40E-04	2.77E-04	Normal	8.15E-04	0.0	6.40E-04
1,2,3,7,8-PECDD	Carp	3	3	100	5.30E-04	1.10E-03	7.20E-04	Other	5.12E-03	0.0	1.10E-03
1,2,3,7,8-PECDF	Carp	3	3	100	5.00E-04	1.10E-03	7.17E-04	Normal	1.28E-03	0.0	1.10E-03
2,3,4,6,7,8-HXCDF	Carp	3	3	100	2.00E-04	9.30E-04	5.70E-04	Normal	1.19E-03	0.0	9.30E-04
2,3,4,7,8-PECDF	Carp	3	3	100	6.20E-04	1.90E-03	1.09E-03	Normal	2.28E-03	0.0	1.90E-03
2,3,7,8-TCDD	Carp	3	3	100	3.70E-04	8.80E-04	5.50E-04	Normal	1.03E-03	0.0	8.80E-04
2,3,7,8-TCDF	Carp	3	3	100	5.40E-04	1.70E-03	1.15E-03	Normal	2.13E-03	0.0	1.70E-03
1,2,3,4,6,7,8,9-OCDD	Walleye	3	3	100	1.50E-03	2.60E-03	2.17E-03	Normal	3.15E-03	0.0	2.60E-03
1,2,3,4,6,7,8,9-OCDF	Walleye	3	0								
1,2,3,4,6,7,8-HPCDD	Walleye	3	3	100	1.40E-03	2.30E-03	1.73E-03	Normal	2.56E-03	0.0	2.30E-03
1,2,3,4,6,7,8-HPCDF	Walleye	3	2	67	4.80E-04	6.40E-04	4.37E-04	Normal	8.21E-04	0.0	6.40E-04
1,2,3,4,7,8,9-HPCDF	Walleye	3	0								
1,2,3,4,7,8-HXCDD	Walleye	3	2	67	4.30E-04	4.70E-04	3.33E-04	Normal	6.76E-04	0.0	4.70E-04
1,2,3,4,7,8-HXCDF	Walleye	3	3	100	2.60E-04	2.80E-04	2.73E-04	Other	2.97E-04	0.0	2.80E-04
1,2,3,6,7,8-HXCDD	Walleye	3	3	100	1.50E-03	1.80E-03	1.63E-03	Normal	1.89E-03	0.0	1.80E-03
1,2,3,6,7,8-HXCDF	Walleye	3	3	100	3.70E-04	4.50E-04	4.00E-04	Normal	4.73E-04	0.0	4.50E-04
1,2,3,7,8,9-HXCDD	Walleye	3	2	67	2.70E-04	2.70E-04	1.97E-04	Other	1.97E+00	0.0	2.70E-04
1,2,3,7,8,9-HXCDF	Walleye	3	0								
1,2,3,7,8-PECDD	Walleye	3	3	100	8.70E-04	1.20E-03	1.06E-03	Normal	1.34E-03	0.0	1.20E-03
1,2,3,7,8-PECDF	Walleye	3	3	100	9.20E-04	1.20E-03	1.07E-03	Normal	1.31E-03	0.0	1.20E-03
2,3,4,6,7,8-HXCDF	Walleye	3	3	100	4.90E-04	6.10E-04	5.40E-04	Normal	6.45E-04	0.0	6.10E-04
2,3,4,7,8-PECDF	Walleye	3	3	100	9.10E-04	1.60E-03	1.24E-03	Normal	1.82E-03	0.0	1.60E-03
2,3,7,8-TCDD	Walleye	3	3	100	6.70E-04	9.90E-04	7.97E-04	Normal	1.08E-03	0.0	9.90E-04
2,3,7,8-TCDF	Walleye	3	3	100	8.50E-03	1.32E-02	1.01E-02	Lognormal	2.11E-02	0.0	1.32E-02

Table 6-32 Dioxin and PCB Congener Concentrations in Little Rapids to De Pere Reach Whole Fish (Continued)

Analyte	Species	Number of Samples	Number of Detects	Detection Frequency (%)	Detected Minimum	Detected Maximum	Mean	Data Distribution	95% UCL	90th Percentile	RME
PCBs (µg/kg)											
PCB Congener 77	Golden Shiner	2	2	100	1.5	1.8	1.6	Normal	2.3	0.0	1.8
PCB Congener 81/87/115	Golden Shiner	2	2	100	13.0	15.0	14.0	Normal	20.3	0.0	15.0
PCB Congener 105	Golden Shiner	2	2	100	4.6	5.3	5.0	Normal	7.1	0.0	5.3
PCB Congener 118	Golden Shiner	2	2	100	19.3	19.9	19.6	Normal	21.4	0.0	19.9
PCB Congener 126	Golden Shiner	2	2	100	0.04	0.04	0.04	Normal	0.04	0.00	0.040
PCB Congener 169	Golden Shiner	2	0								
PCB Congener 77	Yellow Perch	1	1	100	0.01	0.01	NA			0.00	0.010
PCB Congener 81	Yellow Perch	1	0								
PCB Congener 105	Yellow Perch	1	1	100	3.3	3.3	NA			0.0	3.3
PCB Congener 118	Yellow Perch	1	1	100	13.0	13.0	NA			0.0	13.0
PCB Congener 126	Yellow Perch	1	0								
PCB Congener 169	Yellow Perch	1	0								
PCB Congener 77	Carp	4	4	100	0.2	1.4	0.7	Normal	1.3	0.0	1.3
PCB Congener 81	Carp	1	0								
PCB Congener 105	Carp	4	4	100	2.4	49.0	18.2	Normal	43.0	0.0	43.0
PCB Congener 118	Carp	4	4	100	13.1	197	72.1	Normal	172	0.0	171.8
PCB Congener 126	Carp	4	2	50	0.1	0.1	0.05	Normal	0.1	0.0	0.1
PCB Congener 169	Carp	4	0								
PCB Congener 77	Walleye	4	4	100	2.0	7.2	4.5	Normal	7.0	0.0	7.0
PCB Congener 81	Walleye	1	0								
PCB Congener 105	Walleye	4	4	100	23.0	39.6	29.9	Normal	39.0	0.0	39.0
PCB Congener 118	Walleye	4	4	100	58.0	98.1	77.0	Normal	98.6	0.0	98.1
PCB Congener 126	Walleye	4	4	100	0.2	0.4	0.3	Normal	0.4	0.0	0.4
PCB Congener 169	Walleye	4	1	25	0.1	0.1	0.02	Other	55.4	0.0	0.1

Note:
NA - Not applicable.

Table 6-33 Estimated Exposure Concentrations for Piscivorous Birds in Little Rapids to De Pere Reach

Analyte	Surface Water (µg/L) [1]		TL3 Fish: Yellow Perch and Gizzard Shad (µg/kg)		Common Tern Ingestion				Total Common Tern Ingestion (µg/day)		Total Common Tern Ingestion (µg/kg-BW/day)	
					Surface Water (µg/day) [1]		TL3 Fish (µg/day)					
	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME
Mercury	3.88	7.12	25.0	50.0	0.1	0.1	1.5	2.9	1.5	3.0	12.7	25.3
Total PCBs	0.04	0.05	347	370	0.001	0.001	20.4	21.8	20.4	21.8	170	181
Dieldrin	NA	NA	1.3	2.5	—	—	0.1	0.1	0.1	0.1	0.6	1.2
p,p'-DDE	NA	NA	16.0	16.0	—	—	0.9	0.9	0.9	0.9	7.8	7.8

Analyte	Surface Water (µg/L) [1]		TL3 Fish: Yellow Perch and Gizzard Shad (µg/kg)		Forster's Tern Ingestion				Total Forster's Tern Ingestion (µg/day)		Total Forster's Tern Ingestion (µg/kg-BW/day)	
					Surface Water (µg/day) [1]		TL3 Fish (µg/day)					
	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME
Mercury	3.88	7.12	25.0	50.0	0.1	0.1	1.8	3.6	1.9	3.7	11.7	23.4
Total PCBs	0.04	0.05	347	370	0.001	0.001	24.8	26.4	24.8	26.4	157	167
Dieldrin	NA	NA	1.3	2.5	—	—	0.1	0.2	0.1	0.2	0.6	1.1
p,p'-DDE	NA	NA	16.0	16.0	—	—	1.1	1.1	1.1	1.1	7.2	7.2

Analyte	Surface Water (µg/L) [1]		TL3 Fish: Yellow Perch and Gizzard Shad (µg/kg)		Double-crested Cormorant Ingestion				Double-crested Cormorant Ingestion (µg/day)		Total Double-crested Cormorant Ingestion (µg/kg-BW/day)	
					Surface Water (µg/day) [1]		TL3 Fish (µg/day)					
	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME
Mercury	3.88	7.12	25.0	50.0	0.3	0.6	8.0	15.9	8.3	16.5	4.9	9.8
Total PCBs	0.04	0.05	347	370	0.003	0.004	110	118	110	118	65.6	70.0
Dieldrin	NA	NA	1.3	2.5	—	—	0.4	0.8	0.4	0.8	0.2	0.5
p,p'-DDE	NA	NA	16.0	16.0	—	—	5.1	5.1	5.1	5.1	3.0	3.0

Notes:

[1] Mercury in unfiltered water, total PCBs on a filtered + particulate basis.

NA indicates that data was not collected for this reach and this contaminant.

Shaded areas indicate that the concentration is estimated as half of the detection limit.

"—" indicates that the concentration could not be calculated.

Table 6-33 Estimated Exposure Concentrations for Piscivorous Birds in Little Rapids to De Pere Reach (Continued)

Analyte	Surface Water (µg/L) [1]		TL3 Fish: Carp (µg/kg)		TL4 Fish: Walleye (µg/kg)		Bald Eagle Ingestion						Total Bald Eagle Ingestion (µg/day)		Total Bald Eagle Ingestion (µg/kg-BW/day)	
							Surface Water (µg/day) [1]		TL3 Fish (µg/day)		TL4 Fish (µg/day)					
	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME
Mercury	3.9	7.1	150	150	160	160	0.6	1.2	63.3	63.3	16.8	16.8	80.7	81.3	17.4	17.5
Total PCBs	0.04	0.05	3,919	5,800	3,179	4,587	0.01	0.01	1,654	2,448	334	482	1,987	2,929	427	630
Dieldrin	NA	NA	1.8	12.5	3.4	5.4	—	—	0.8	5.3	0.4	0.6	1.1	5.8	0.2	1.3
p,p'-DDE	NA	NA	74.2	128	129	208	—	—	31.3	53.8	13.5	21.9	44.9	75.7	9.6	16.3

Notes:

[1] Mercury in unfiltered water, total PCBs on a filtered + particulate basis.

NA indicates that data was not collected for this reach and this contaminant.

Shaded areas indicate that the concentration is estimated as half of the detection limit.

"—" indicates that the concentration could not be calculated.

Model Assumptions:

Common Tern

Food Ingestion =	0.0588	kg/day
Water Ingestion =	0.014	L/day
Body Weight =	0.12	kg

Forster's Tern

Food Ingestion =	0.0714	kg/day
Water Ingestion =	0.017	L/day
Body Weight =	0.158	kg

Double-crested Cormorant

Food Ingestion =	0.318	kg/day
Water Ingestion =	0.084	L/day
Body Weight =	1.68	kg

Bald Eagle

Food Ing. (TL3 Fish) =	0.422	kg/day
Food Ing. (TL4 Fish) =	0.105	kg/day
Water Ingestion =	0.165	L/day
Body Weight =	4.65	kg

Table 6-34 Estimated Exposure Concentrations for Mink in Little Rapids to De Pere Reach

Analyte	Surface Water (µg/L) [1]		Surface Sediment (µg/kg) [2]		Whole Carp (µg/kg)		Mink Ingestion						Total Mink Ingestion (µg/day)		Total Mink Ingestion (µg/kg-BW/day)	
							Surface Water (µg/day) [1]		Surface Sediment (µg/day)		Whole Carp (µg/day)					
	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME
Mercury	3.9	7.1	3,500	4,000	150	150	0.3	0.6	14.0	16.0	23.0	23.0	37.3	39.5	46.6	49.4
Total PCBs (N)	0.04	0.05	4,782	10,543	3,919	5,800	0.003	0.004	19.1	42.2	600	887	619	930	773	1,162
Total PCBs (I_0)	0.04	0.05	2,054	2,088	3,919	5,800	0.003	0.004	8.2	8.4	600	887	608	896	760	1,120
Total PCBs (I_d)	0.04	0.05	2,078	2,112	3,919	5,800	0.003	0.004	8.3	8.4	600	887	608	896	760	1,120
Dieldrin	NA	NA	5.0	15.9	1.8	12.5	—	—	0.02	0.1	0.3	1.9	0.3	2.0	0.4	2.5
p,p'-DDE	NA	NA	16.5	20.0	74.2	128	—	—	0.1	0.1	11.4	19.5	11.4	19.6	14.3	24.5

Notes:

[1] Mercury in unfiltered water, total PCBs on a filtered + particulate basis.

[2] p,p'-DDT rather than p,p'-DDE was used because this was the predominant form in the sediment.

NA indicates that data was not collected for this reach and this contaminant.

Shaded areas indicate that the concentration is estimated as half of the detection limit.

"—" indicates that the concentration could not be calculated.

Model Assumptions:

Food Ingestion =	0.153	kg/day
Water Ingestion =	0.081	L/day
Sediment Ingestion =	0.004	kg/day
Body Weight =	0.8	kg

Table 6-35 Surface Water Concentrations in De Pere to Green Bay Reach (Green Bay Zone 1)

Analyte	Number of Samples	Number of Detects	Detection Frequency (%)	Detected Minimum	Detected Maximum	Mean	Data Distribution	95% UCL	90th Percentile	RME	Criteria NOAEC	Criteria LOAEC	Reference	Hazard Quotients Mean NOEL	Hazard Quotients Mean LOEL	Hazard Quotients RME NOEL	Hazard Quotients RME LOEL
Metals (ng/L)																	
Arsenic (unfiltered)	4	1	25	1,500	1,500	NE	Normal	6,655	0.0	1,500	152,200		[1]	NE		< 0.1	
Lead (unfiltered)	4	3	75	1,450	5,300	3,113	Normal	5,205	0.0	5,205	49,420		[1]	0.1		0.1	
Mercury (filtered)	45	43	96	0.5	40.8	4.9	Other	7.6	17.5	7.6	440		[1]	< 0.1		< 0.1	
Mercury (unfiltered)	45	41	91	1.8	191	27.5	Other	40.3	41.6	40.3	440		[1]	0.1		0.1	
Mercury (particulate)	32	32	100	1.8	74.8	23.0	Other	37.0	38.6	37.0	440		[1]	0.1		0.1	
Dioxins/Furans (ng/L)																	
2,3,7,8-TCDD (unfiltered)	3	0															
2,3,7,8-TCDF (unfiltered)	2	0															
PCBs (ng/L)																	
Total PCBs (filtered)	143	142	99	2.4	45.0	16.6	Normal	17.7	25.9	17.7	50	500	[2]	0.3	< 0.1	0.4	< 0.1
Total PCBs (particulate)	143	129	90	1.4	149	44.2	Other	54.7	90.4	54.7	50	500	[2]	0.9	0.1	1.1	0.1
Total PCBs (filtered + particulate)				3.8	194	60.9		72.4	116	72.4	50	500	[2]	1.2	0.1	1.4	0.1
Pesticides (ng/L)																	
Dieldrin (unfiltered)	4	0															
p,p'-DDD (filtered)	7	5	71	0.05	0.07	0.05	Normal	0.06	0.00	0.06	1		[3]	< 0.1		0.1	
p,p'-DDD (unfiltered)	4	0															
p,p'-DDD (particulate)	40	38	95	0.05	0.27	0.11	Lognormal	0.13	0.19	0.13	1		[3]	0.1		0.1	
p,p'-DDE (filtered)	19	19	100	0.03	0.07	0.04	Other	0.04	0.05	0.04	1		[3]	< 0.1		< 0.1	
p,p'-DDE (unfiltered)	4	0															
p,p'-DDE (particulate)	42	41	98	0.03	0.41	0.17	Normal	0.19	0.26	0.19	1		[3]	0.2		0.2	
p,p'-DDT (unfiltered)	4	0															
p,p'-DDT (particulate)	8	7	88	0.05	0.21	0.07	Other	0.13	0.00	0.13	1		[3]	0.1		0.1	

Notes:

[1] Wisconsin Administrative Code Chapter NR 105.

[2] Niimi, 1996.

[3] Chronic National Ambient Water Quality Criteria (EPA, 1980a, 1980b, 1980c).

NE indicates that the calculated mean exceeded the detected maximum and, therefore, the mean was not evaluated.

Table 6-36 Surface Sediment Concentrations in De Pere to Green Bay Reach (Green Bay Zone 1)

Analyte	Number of Samples	Number of Detects	Detection Frequency (%)	Detected Minimum	Detected Maximum	Mean	Data Distribution	95% UCL	90th Percentile	RME	Criteria	Reference	Hazard Quotients	
													Mean	RME
Metals (mg/kg)														
Arsenic	92	66	72	0.8	386	10.1	Other	16.9	9.9	16.9	12.1	1	0.8	1.4
Lead	92	92	100	4.4	350	75.7	Other	91.2	110	91.2	34.2	1	2.2	2.7
Mercury	92	89	97	0.1	7.7	1.0	Other	1.4	1.7	1.4	0.17	2	6.1	8.1
PCBs (µg/kg)														
Total PCBs (N)	290	285	98	19.9	99000	4,184	Other	5,510	8,170	5,510	31.6	1	132	174
Total PCBs (I_0)	52,115	52,115	100	0.0	98,991	2,950	Other	2,976	5,784	2,976	31.6	1	93	94
Total PCBs (I_d)	51,963	51,963	100	51.0	98,991	2,959	Other	2,984	5,789	2,984	31.6	1	94	94
Pesticides (µg/kg)														
Dieldrin	22	0												
p,p'-DDD	22	3	14	1.2	4.5	NE	Other	13.7	11.5	4.5	3.54	2	NE	1.3
p,p'-DDE	22	1	5	1.9	1.9	NE	Other	15.4	12.9	1.9	1.42	2	NE	1.3
p,p'-DDT	22	0												

Notes:

N indicates that the data was not interpolated based on depth.

I_0 indicates that interpolated grid areas for which "no values" existed were assumed to equal zero.

I_d indicates that interpolated grid areas for which "no values" existed were deleted from the database.

[1] ARCS SEC (EPA, 1996a).

[2] Environment Canada TEL (Smith *et al.,* 1996); p,p'-DDT based on total DDT TEL.

NE indicates that the calculated mean exceeded the detected maximum and, therefore, the mean was not evaluated.

Table 6-37 PCB Congener Concentrations in Surface Sediment in De Pere to Green Bay Reach (Green Bay Zone 1)

Analyte	Number of Samples	Number of Detects	Detection Frequency (%)	Detected Minimum	Detected Maximum	Mean	Data Distribution	95% UCL	90th Percentile	RME
PCBs (µg/kg)										
PCB Congener 77	26	24	92	1.9	85.0	13.0	Lognormal	27.0	42.3	27.0
PCB Congener 81	21	16	76	0.04	0.2	NE	Other	0.6	1.2	0.2
PCB Congener 105	26	25	96	0.8	23.0	5.6	Other	10.6	17.5	10.6
PCB Congener 118	26	26	100	1.4	46.0	12.7	Other	24.1	42.5	24.1
PCB Congener 126	26	5	19	0.03	0.3	0.2	Other	0.5	1.2	0.3
PCB Congener 169	26	0								

Note:

NE indicates that the calculated mean exceeded the detected maximum and, therefore, the mean was not evaluated.

Table 6-38 De Pere to Green Bay Reach (Green Bay Zone 1) Bird Tissue Concentrations

Analyte	Species	Tissue	Number of Samples	Number of Detects	Detection Frequency (%)	Detected Minimum	Detected Maximum	Mean	Data Distribution	95% UCL	90th Percentile	RME
PCBs (µg/kg)												
Total PCBs	Tree Swallow	whole	22	22	100	510	17,000	3,118	Lognormal	4,505	7,100	4,505
Pesticides (µg/kg)												
Dieldrin	Tree Swallow	whole	22	0								
o,p'-DDD	Tree Swallow	whole	22	0								
o,p'-DDE	Tree Swallow	whole	22	0								
o,p'-DDT	Tree Swallow	whole	22	0								
p,p'-DDD	Tree Swallow	whole	22	3	14	12	14.0	6.1	Other	7.1	12.7	7.1
p,p'-DDE	Tree Swallow	whole	22	22	100	28	520	218	Lognormal	331	495	331
p,p'-DDT	Tree Swallow	whole	22	0								

Table 6-39 Estimated Exposure Concentrations for Piscivorous Birds in Green Bay Zone 1

Analyte	Surface Water (µg/L) [1]		TL3 Fish: Alewife (µg/kg)		Common Tern Ingestion				Total Common Tern Ingestion (µg/day)		Total Common Tern Ingestion (µg/kg-BW/day)	
					Surface Water (µg/day) [1]		TL3 Fish: (µg/day)					
	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME
Mercury	0.03	0.04	100	250	0.0004	0.001	5.9	14.7	5.9	14.7	49.0	123
Total PCBs	0.06	0.07	2,599	3,182	0.001	0.001	153	187	153	187	1,274	1,559
Dieldrin	0.006	0.013	21.0	57.9	0.0001	0.0002	1.2	3.4	1.2	3.4	10.3	28.4
p,p'-DDE	0.0002	0.0002	104	143	0.000003	0.000003	6.1	8.4	6.1	8.4	51.1	70.0

Analyte	Surface Water (µg/L) [1]		TL3 Fish: Alewife (µg/kg)		Forster's Tern Ingestion				Total Forster's Tern Ingestion (µg/day)		Total Forster's Tern Ingestion (µg/kg-BW/day)	
					Surface Water (µg/day) [1]		TL3 Fish: (µg/day)					
	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME
Mercury	0.03	0.04	100	250	0.0005	0.001	7.1	17.9	7.1	17.9	45.2	113
Total PCBs	0.06	0.07	2,599	3,182	0.001	0.001	186	227	186	227	1,175	1,438
Dieldrin	0.006	0.013	21.0	57.9	0.0001	0.0002	1.5	4.1	1.5	4.1	9.5	26.2
p,p'-DDE	0.0002	0.0002	104	143	0.000003	0.000003	7.4	10.2	7.4	10.2	47.1	64.6

Analyte	Surface Water (µg/L) [1]		TL3 Fish: Alewife (µg/kg)		Double-crested Cormorant Ingestion				Double-crested Cormorant Ingestion (µg/day)		Total Double-crested Cormorant Ingestion (µg/kg-BW/day)	
					Surface Water (µg/day) [1]		TL3 Fish: (µg/day)					
	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME
Mercury	0.03	0.04	100	250	0.002	0.003	31.8	79.5	31.8	79.5	18.9	47.3
Total PCBs	0.06	0.07	2,599	3,182	0.01	0.01	826	1,012	826	1,012	492	602
Dieldrin	0.006	0.013	21.0	57.9	0.0005	0.001	6.7	18.4	6.7	18.4	4.0	11.0
p,p'-DDE	0.0002	0.0002	104	143	0.00002	0.00002	33.1	45.4	33.1	45.4	19.7	27.0

Notes:

[1] Mercury in unfiltered water, total PCBs on a filtered + particulate basis, p,p'-DDE on a particulate basis.

Shaded areas indicate that the concentration is estimated as half of the detection limit.

Table 6-39 Estimated Exposure Concentrations for Piscivorous Birds in Green Bay Zone 1 (Continued)

Analyte	Surface Water (µg/L)[1]		TL3 Fish: Carp (µg/kg)		TL4 Fish: Walleye (µg/kg)		Bald Eagle Ingestion						Total Bald Eagle Ingestion (µg/day)		Total Bald Eagle Ingestion (µg/kg-BW/day)	
							Surface Water (µg/day)[1]		TL3 Fish (µg/day)		TL4 Fish (µg/day)					
	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME
Mercury	0.03	0.04	60.0	70.0	210	270	0.005	0.01	25.3	29.5	22.1	28.4	47.4	57.9	10.2	12.5
Total PCBs	0.06	0.07	6,637	7,369	6,539	7,658	0.01	0.01	2,801	3,110	687	804	3,487	3,914	750	842
Dieldrin	0.005	0.013	20.8	29.4	37.3	52.2	0.001	0.002	8.8	12.4	3.9	5.5	12.7	17.9	2.7	3.8
p,p'-DDE	0.0002	0.0002	196.5	700.0	353.0	462.0	0.00003	0.00003	82.9	295	37.1	48.5	120	344	25.8	74.0

Notes:

[1] Mercury in unfiltered water, total PCBs on a filtered + particulate basis, p,p'-DDE on a particulate basis.

Shaded areas indicate that the concentration is estimated as half of the detection limit.

Model Assumptions:

Common Tern

Food Ingestion = 0.0588 kg/day
Water Ingestion = 0.014 L/day
Body Weight = 0.12 kg

Forster's Tern

Food Ingestion = 0.0714 kg/day
Water Ingestion = 0.017 L/day
Body Weight = 0.158 kg

Double-crested Cormorant

Food Ingestion = 0.318 kg/day
Water Ingestion = 0.084 L/day
Body Weight = 1.68 kg

Bald Eagle

Food Ing. (TL3 Fish) = 0.422 kg/day
Food Ing. (TL4 Fish) = 0.105 kg/day
Water Ingestion = 0.165 L/day
Body Weight = 4.65 kg

Table 6-40 Estimated Exposure Concentrations for Mink in Green Bay Zone 1

Analyte	Surface Water (µg/L) [1]		Surface Sediment (µg/kg) [2]		Whole Carp (µg/kg)		Mink Ingestion						Total Mink Ingestion (µg/day)		Total Mink Ingestion (µg/kg-BW/day)	
							Surface Water (µg/day) [1]		Surface Sediment (µg/day)		Whole Carp (µg/day)					
	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME
Mercury	0.03	0.04	1,000	1,400	60.0	70.0	0.002	0.003	4.0	5.6	9.2	10.7	13.2	16.3	16.5	20.4
Total PCBs (N)	0.06	0.07	4,184	5,510	6,637	7,369	0.005	0.01	16.7	22.0	1,015	1,127	1,032	1,150	1,290	1,437
Total PCBs (I_0)	0.06	0.07	2,950	2,976	6,637	7,369	0.005	0.01	11.8	11.9	1,015	1,127	1,027	1,139	1,284	1,424
Total PCBs (I_d)	0.06	0.07	2,959	2,984	6,637	7,369	0.005	0.01	11.8	11.9	1,015	1,127	1,027	1,139	1,284	1,424
Dieldrin	0.006	0.013	3.8	7.2	20.8	29.4	0.0004	0.001	0.02	0.03	3.2	4.5	3.2	4.5	4.0	5.7
p,p'-DDE	0.0002	0.0002	4.1	6.4	197	700	0.00002	0.00002	0.02	0.03	30.1	107	30.1	107	37.6	134

Notes:

[1] Mercury in unfiltered water, total PCBs on a filtered + particulate basis.

[2] p,p'-DDT rather than p,p'-DDE was used because this was the predominant form in the sediment.

Shaded areas indicate that the concentration is estimated as half of the detection limit.

Model Assumptions:

Food Ingestion =	0.153	kg/day
Water Ingestion =	0.081	L/day
Sediment Ingestion =	0.004	kg/day
Body Weight =	0.8	kg

Table 6-41 Surface Water Concentrations in Green Bay Zone 2 (2A and 2B)

Analyte	Number of Samples	Number of Detects	Detection Frequency (%)	Detected Minimum	Detected Maximum	Mean	Data Distribution	95% UCL	90th Percentile	RME	Criteria NOAEC	Criteria LOAEC	Reference	Hazard Quotients Mean NOEL	Hazard Quotients Mean LOEL	Hazard Quotients RME NOEL	Hazard Quotients RME LOEL
Metals (ng/L)																	
Lead (filtered)	2	2	100	44.0	44.2	44.1	Normal	44.7	0.0	44.2	49,420		[1]	< 0.1		< 0.1	
Lead (unfiltered)	2	2	100	73.3	264	169	Normal	771	0.0	264	49,420		[1]	< 0.1		< 0.1	
Mercury (filtered)	10	2	20	1,150	2,330	391	Other	2,506	2,212	2,300	440		[1]	0.9		5.2	
Mercury (unfiltered)	11	2	18	1,520	5,000	629	Other	7,664	4,304	5,000	440		[1]	1.4		11	
PCBs (ng/L)																	
Total PCBs (filtered)	63	63	100	1.0	13.7	4.8	Normal	5.4	9.9	5.4	50	500	[2]	0.1	< 0.1	0.1	< 0.1
Total PCBs (particulate)	71	71	100	1.3	91.7	13.0	Lognormal	15.2	25.7	15.2	50	500	[2]	0.3	< 0.1	0.3	< 0.1
Total PCBs (filtered + particulate)				2.3	105	17.8		20.7	35.6	20.7	50	500	[2]	0.4	< 0.1	0.4	< 0.1

Notes:

[1] Wisconsin Administrative Code Chapter NR 105.

[2] Niimi, 1996.

Table 6-42 Surface Sediment Concentrations in Green Bay Zone 2

Analyte	Number of Samples	Number of Detects	Detection Frequency (%)	Detected Minimum	Detected Maximum	Mean	Data Distribution	95% UCL	90th Percentile	RME	Criteria	Reference	Hazard Quotients	
													Mean	RME
Metals (mg/kg)														
Arsenic	11	10	91	1.0	3.2	2.1	Normal	2.6	3.1	2.6	12.1	[1]	0.2	0.2
Lead	11	11	100	2.0	42.0	19.7	Normal	28.1	42.0	28.1	34.2	[1]	0.6	0.8
Mercury	11	9	82	0.1	1.5	0.5	Lognormal	3.9	1.5	1.5	0.17	[2]	2.9	8.8
PCBs (µg/kg)														
Total PCBs (N)	15	14	93	26.0	799	251	Lognormal	720	742	720	31.6	[1]	7.9	23
Total PCBs (I_d)	11,566	11,566	100	37.8	10,032	1,132	Other	1,154	3,461	1,154	31.6	[1]	36	37
Pesticides (µg/kg)														
Dieldrin	11	0												
p,p'-DDD	11	0												
p,p'-DDE	11	0												
p,p'-DDT	11	0												

Notes:

N indicates that the data was not interpolated based on depth.

I_0 indicates that interpolated grid areas for which "no values" existed were assumed to equal zero.

I_d indicates that interpolated grid areas for which "no values" existed were deleted from the database.

[1] ARCS SEC (EPA, 1996a).

[2] Environment Canada TEL (Smith *et al.,* 1996); p,p'-DDT based on total DDT TEL.

Table 6-43 PCB Congener Concentrations in Surface Sediment in Green Bay Zone 2

Analyte	Number of Samples	Number of Detects	Detection Frequency (%)	Detected Minimum	Detected Maximum	Mean	Data Distribution	95% UCL	90th Percentile	RME
PCBs (µg/kg)										
PCB Congener 77	11	11	100	0.1	9.2	3.2	Lognormal	40.3	8.9	9.2
PCB Congener 81	15	12	80	0.02	0.6	0.2	Lognormal	0.4	0.5	0.4
PCB Congener 105	11	10	91	0.1	5.2	1.9	Lognormal	13.2	5.1	5.2
PCB Congener 118	15	14	93	0.1	15.9	3.8	Lognormal	20.1	11.9	15.9
PCB Congener 126	11	5	45	0.01	0.1	0.04	Normal	0.1	0.1	0.1
PCB Congener 169	11	0								

Table 6-44 Green Bay Zones 1 and 2 Whole Fish Concentrations

Analyte	Species	Number of Samples	Number of Detects	Detection Frequency (%)	Detected Minimum	Detected Maximum	Mean	Data Distribution	95% UCL	90th Percentile	RME
Metals (mg/kg)											
Mercury	Alewife	5	2	40	0.10	0.25	0.10	Other	0.43	0.0	0.25
	Gizzard Shad	7	0								
	Rainbow Smelt	4	4	100	0.02	0.04	0.03	Normal	0.04	0.0	0.04
	Yellow Perch	9	0								
	Carp	10	1	10	0.12	0.12	0.06	Other	0.07	0.1	0.07
	Walleye	11	10	91	0.11	0.39	0.21	Normal	0.27	0.4	0.27
PCBs (µg/kg)											
Total PCBs	Alewife	51	51	100	990	19,000	2,599	Other	3,182	3,960	3,182
	Gizzard Shad	50	50	100	700	4,100	1,852	Normal	2,005	2,781	2,005
	Rainbow Smelt	33	33	100	280	1,600	1,049	Normal	1,152	1,500	1,152
	Common Shiner	5	5	100	3,100	4,000	3,520	Normal	3,846	0.0	3,846
	Emerald Shiners	5	5	100	3,100	4,000	3,520	Normal	3,846	0.0	3,846
	Golden Shiner	2	2	100	1,326	1,443	1,385	Normal	1,754	0.0	1,443
	Yellow Perch	9	9	100	614	2,151	1,206	Normal	1,567	0.0	1,567
	Carp	115	115	100	202	22,500	6,637	Normal	7,369	13,280	7,369
	Walleye	91	91	100	387	19,000	6,539	Other	7,658	10,923	7,658
Pesticides (µg/kg)											
Dieldrin	Alewife	51	45	88	3.5	140	21.0	Other	57.9	47.8	57.9
	Gizzard Shad	46	22	48	1.7	80.0	10.5	Other	48.4	19.0	48.4
	Rainbow Smelt	33	29	88	0.7	21.0	7.5	Normal	8.7	12.0	8.7
	Yellow Perch	9	0								
	Carp	78	66	85	0.8	91.0	20.8	Lognormal	29.4	67.1	29.4
	Walleye	70	58	83	1.8	190	37.3	Other	52.2	80.7	52.2
o,p'-DDD	Gizzard Shad	15	0								
	Rainbow Smelt	4	0								
	Carp	4	0								
	Walleye	3	0								

Table 6-44 Green Bay Zones 1 and 2 Whole Fish Concentrations (Continued)

Analyte	Species	Number of Samples	Number of Detects	Detection Frequency (%)	Detected Minimum	Detected Maximum	Mean	Data Distribution	95% UCL	90th Percentile	RME
o,p'-DDE	Gizzard Shad	8	0								
	Rainbow Smelt	4	0								
	Carp	4	3	75	15.0	88.0	50.0	Normal	91.8	0.0	88.0
	Walleye	3	3	100	64.0	120	85.0	Normal	136	0.0	120
o,p'-DDT	Gizzard Shad	15	0								
	Rainbow Smelt	4	0								
	Carp	4	0								
	Walleye	3	0								
p,p'-DDD	Alewife	5	1	20	11.0	11.0	7.3	Normal	10.6	0.0	10.6
	Gizzard Shad	22	1	5	26.0	26.0	22.8	Other	28.5	25.7	26.0
	Yellow Perch	9	0								
	Carp	13	3	23	51.0	79.0	31.8	Other	167	79.6	79.0
	Walleye	14	1	7	33.0	33.0	23.5	Lognormal	57.0	62.5	33.0
p,p'-DDE	Alewife	5	5	100	56.0	150	104	Normal	143	0.0	143
	Gizzard Shad	22	8	36	70.0	380	64.2	Other	93.6	144	93.6
	Rainbow Smelt	4	0								
	Yellow Perch	9	9	100	10.0	64.0	32.9	Normal	45.1	0.0	45.1
	Carp	13	13	100	9.0	700	197	Lognormal	1,048	612	700
	Walleye	14	14	100	18.0	760	353	Normal	462	705	462
p,p'-DDT	Alewife	5	0								
	Gizzard Shad	22	0								
	Rainbow Smelt	4	0								
	Yellow Perch	9	0								
	Carp	13	0								
	Walleye	14	0								

Table 6-45 Dioxin and PCB Congener Concentrations in Green Bay Zones 1 and 2 Whole Fish

Analyte	Species	Number of Samples	Number of Detects	Detection Frequency (%)	Detected Minimum	Detected Maximum	Mean	Data Distribution	95% UCL	90th Percentile	RME
Dioxins/Furans (µg/kg)											
1,2,3,4,6,7,8,9-OCDD	Carp	3	3	100	8.80E-03	7.54E-02	3.20E-02	Normal	9.54E-02	0.0	7.54E-02
1,2,3,4,6,7,8,9-OCDF	Carp	3	0								
1,2,3,4,6,7,8-HPCDD	Carp	3	3	100	4.60E-03	4.40E-02	1.90E-02	Normal	5.56E-02	0.0	4.40E-02
1,2,3,4,6,7,8-HPCDF	Carp	3	3	100	7.60E-04	5.60E-03	2.55E-03	Normal	7.02E-03	0.0	5.60E-03
1,2,3,4,7,8,9-HPCDF	Carp	3	0								
1,2,3,4,7,8-HXCDD	Carp	3	3	100	4.40E-04	4.80E-03	2.28E-03	Normal	6.09E-03	0.0	4.80E-03
1,2,3,4,7,8-HXCDF	Carp	3	2	67	5.40E-04	2.40E-03	1.01E-03	Normal	3.07E-03	0.0	2.40E-03
1,2,3,6,7,8-HXCDD	Carp	3	3	100	1.10E-03	1.31E-02	6.17E-03	Normal	1.66E-02	0.0	1.31E-02
1,2,3,6,7,8-HXCDF	Carp	3	3	100	2.70E-04	4.10E-03	1.79E-03	Normal	5.22E-03	0.0	4.10E-03
1,2,3,7,8,9-HXCDD	Carp	3	3	100	2.00E-04	1.00E-03	5.30E-04	Normal	1.23E-03	0.0	1.00E-03
1,2,3,7,8,9-HXCDF	Carp	3	1	33	1.70E-04	1.70E-04	6.67E-05	Other	2.46E+04	0.0	1.70E-04
1,2,3,7,8-PECDD	Carp	3	3	100	4.50E-04	1.60E-03	1.12E-03	Normal	2.12E-03	0.0	1.60E-03
1,2,3,7,8-PECDF	Carp	3	2	67	6.50E-04	2.80E-03	1.20E-03	Normal	4.E-03	0.0	2.80E-03
2,3,4,6,7,8-HXCDF	Carp	3	3	100	4.10E-04	7.80E-04	6.40E-04	Normal	9.78E-04	0.0	7.80E-04
2,3,4,7,8-PECDF	Carp	3	3	100	7.80E-04	3.60E-03	2.19E-03	Normal	4.57E-03	0.0	3.60E-03
2,3,7,8-TCDD	Carp	3	3	100	3.50E-04	1.30E-03	9.83E-04	Other	2.99E-01	0.0	1.30E-03
2,3,7,8-TCDF	Carp	3	3	100	1.60E-03	4.60E-03	2.90E-03	Normal	5.50E-03	0.0	4.60E-03
1,2,3,4,6,7,8,9-OCDD	Walleye	3	3	100	2.60E-03	4.50E-03	3.30E-03	Normal	5.06E-03	0.0	4.50E-03
1,2,3,4,6,7,8,9-OCDF	Walleye	3	0								
1,2,3,4,6,7,8-HPCDD	Walleye	3	3	100	3.00E-03	5.20E-03	3.73E-03	Other	1.22E-02	0.0	5.20E-03
1,2,3,4,6,7,8-HPCDF	Walleye	3	3	100	7.50E-04	1.30E-03	9.70E-04	Normal	1.46E-03	0.0	1.30E-03
1,2,3,4,7,8,9-HPCDF	Walleye	3	1	33	1.70E-04	1.70E-04	8.33E-05	Normal	2.11E-04	0.0	1.70E-04
1,2,3,4,7,8-HXCDD	Walleye	3	3	100	5.60E-04	9.50E-04	7.20E-04	Normal	1.06E-03	0.0	9.50E-04
1,2,3,4,7,8-HXCDF	Walleye	3	3	100	3.70E-04	6.20E-04	4.70E-04	Normal	6.93E-04	0.0	6.20E-04
1,2,3,6,7,8-HXCDD	Walleye	3	3	100	2.70E-03	4.60E-03	3.40E-03	Normal	5.16E-03	0.0	4.60E-03
1,2,3,6,7,8-HXCDF	Walleye	3	3	100	5.80E-04	9.60E-04	7.10E-04	Normal	1.08E-03	0.0	9.60E-04
1,2,3,7,8,9-HXCDD	Walleye	3	3	100	3.80E-04	5.90E-04	4.77E-04	Normal	6.55E-04	0.0	5.90E-04
1,2,3,7,8,9-HXCDF	Walleye	3	1	33	8.00E-05	8.00E-05	5.83E-05	Normal	9.02E-05	0.0	8.00E-05
1,2,3,7,8-PECDD	Walleye	3	3	100	1.90E-03	3.10E-03	2.37E-03	Normal	3.45E-03	0.0	3.10E-03
1,2,3,7,8-PECDF	Walleye	3	3	100	1.40E-03	2.20E-03	1.70E-03	Normal	2.43E-03	0.0	2.20E-03
2,3,4,6,7,8-HXCDF	Walleye	3	3	100	7.10E-04	1.30E-03	9.50E-04	Normal	1.47E-03	0.0	1.30E-03
2,3,4,7,8-PECDF	Walleye	3	3	100	2.80E-03	3.90E-03	3.27E-03	Normal	4.23E-03	0.0	3.90E-03
2,3,7,8-TCDD	Walleye	3	3	100	1.00E-03	2.00E-03	1.40E-03	Normal	2.29E-03	0.0	2.00E-03
2,3,7,8-TCDF	Walleye	3	3	100	1.41E-02	1.94E-02	1.69E-02	Normal	2.14E-02	0.0	1.94E-02
PCBs (µg/kg)											
PCB Congener 77	Alewife	5	5	100	0.3	1.4	0.9	Normal	1.3	0.0	1.3
PCB Congener 81	Alewife	51	44	86	2.8	48.0	6.4	Other	114.8	10.0	48.0
PCB Congener 105	Alewife	5	5	100	10.0	29.0	19.8	Normal	27.6	0.0	27.6
PCB Congener 118	Alewife	51	51	100	21.0	480	66.2	Other	80.7	89.6	80.7
PCB Congener 126	Alewife	5	5	100	0.02	0.7	0.2	Normal	0.5	0.0	0.5
PCB Congener 169	Alewife	5	0								

Table 6-45 Dioxin and PCB Congener Concentrations in Green Bay Zones 1 and 2 Whole Fish (Continued)

Analyte	Species	Number of Samples	Number of Detects	Detection Frequency (%)	Detected Minimum	Detected Maximum	Mean	Data Distribution	95% UCL	90th Percentile	RME
PCB Congener 77	Gizzard Shad	6	6	100	0.1	1.8	1.0	Normal	1.6	0.0	1.6
PCB Congener 81	Gizzard Shad	29	25	86	0.1	4.6	3.0	Other	150	4.5	4.6
PCB Congener 105	Gizzard Shad	6	6	100	8.3	28.0	19.1	Normal	25.1	0.0	25.1
PCB Congener 118	Gizzard Shad	29	28	97	21.0	63.0	32.3	Other	105	61.0	63.0
PCB Congener 126	Gizzard Shad	6	5	83	0.1	0.3	0.2	Other	37.9	0.0	0.3
PCB Congener 169	Gizzard Shad	6	1	17	0.1	0.1	0.02	Other	0.4	0.0	0.1
PCB Congener 77/110	Rainbow Smelt	29	29	100	23.0	73.0	41.6	Normal	45.7	62.0	45.7
PCB Congener 81	Rainbow Smelt	29	29	100	1.8	7.9	3.7	Lognormal	4.2	6.4	4.2
PCB Congener 132/153/105	Rainbow Smelt	29	29	100	16.0	68.0	32.6	Lognormal	37.3	55.0	37.3
PCB Congener 118	Rainbow Smelt	29	29	100	15.0	47.0	26.9	Normal	29.6	42.0	29.6
PCB Congener 77	Golden Shiner	2	2	100	2.3	3.1	2.7	Normal	5.1	0.0	3.1
PCB Congener 81/87/115	Golden Shiner	2	2	100	23.0	25.0	24.0	Normal	30.3	0.0	25.0
PCB Congener 105	Golden Shiner	2	2	100	9.3	11.8	10.6	Normal	18.6	0.0	11.8
PCB Congener 118	Golden Shiner	2	2	100	23.5	32.2	27.8	Normal	55.4	0.0	32.2
PCB Congener 126	Golden Shiner	2	2	100	0.1	0.1	0.1	Normal	0.2	0.0	0.1
PCB Congener 169	Golden Shiner	2	0								
PCB Congener 77	Yellow Perch	9	9	100	0.02	0.8	0.2	Lognormal	1.3	0.0	0.8
PCB Congener 81	Yellow Perch	9	0								
PCB Congener 105	Yellow Perch	9	8	89	5.6	25.0	12.8	Normal	18.0	0.0	18.0
PCB Congener 118	Yellow Perch	9	9	100	12.0	64.0	34.4	Normal	45.4	0.0	45.4
PCB Congener 126	Yellow Perch	9	2	22	0.02	0.04	0.01	Other	0.02	0.0	0.02
PCB Congener 169	Yellow Perch	9	0								
PCB Congener 77	Carp	14	14	100	0.1	6.0	1.4	Lognormal	10.8	4.6	6.0
PCB Congener 81	Carp	77	69	90	0.2	39.0	14.1	Normal	16.2	30.2	16.2
PCB Congener 105	Carp	14	14	100	1.9	138	37.4	Lognormal	182	137	138
PCB Congener 118	Carp	80	80	100	7.2	470	138	Normal	157	288	157
PCB Congener 126	Carp	14	10	71	0.02	2.5	0.3	Lognormal	5.7	1.5	2.5
PCB Congener 169	Carp	16	3	19	0.02	0.8	0.1	Other	0.3	0.8	0.3
PCB Congener 77	Walleye	16	16	100	0.2	11.2	4.9	Normal	6.5	9.6	6.5
PCB Congener 81	Walleye	69	65	94	0.1	61.0	15.7	Other	227	28.0	61.0
PCB Congener 105	Walleye	27	25	93	5.8	251	69.0	Other	1,357	163	251
PCB Congener 118	Walleye	83	83	100	13.0	697	174	Other	199	307	199
PCB Congener 126	Walleye	16	15	94	0.1	5.3	1.1	Lognormal	9.3	3.8	5.3
PCB Congener 169	Walleye	25	16	64	0.04	1.7	0.3	Other	4.0	1.2	1.7

Table 6-46 Green Bay Zone 2 Bird Tissue Concentrations

Analyte	Species	Tissue	Number of Samples	Number of Detects	Detection Frequency (%)	Detected Minimum	Detected Maximum	Mean	Data Distribution	95% UCL	90th Percentile	RME
PCBs (µg/kg)												
Total PCBs	Double-crested Cormorant	brain	5	5	100	1,900	6,000	3,700	Normal	5,307	0.0	5,307
	Double-crested Cormorant	egg	34	34	100	610	74,000	13,944	Other	21,127	25,000	21,127
	Double-crested Cormorant	whole	74	74	100	324	63,000	11,026	Other	13,870	21,500	13,870
	Common Tern	egg	10	10	100	2,266	9,011	4,819	Normal	5,963	8,751	5,963
	Forster's Tern	egg	10	10	100	1,478	8,092	5,077	Normal	6,234	7,992	6,234
	Tree Swallow	whole	15	15	100	1,200	4,500	2,980	Normal	3,495	4,440	3,495
Pesticides (µg/kg)												
Dieldrin	Double-crested Cormorant	brain	5	5	100	30.0	64.0	48.2	Normal	60.5	0.0	60.5
	Double-crested Cormorant	egg	34	32	94	39.0	1,300	224	Other	445	545	445
	Double-crested Cormorant	whole	73	73	100	36.0	1,300	196	Lognormal	243	412	243
	Common Tern	egg	5	5	100	29.8	155	85.0	Normal	139	0.0	139
	Forster's Tern	egg	7	7	100	26.5	84.9	47.6	Normal	62.7	0.0	62.7
	Tree Swallow	whole	15	0								
o,p'-DDD	Double-crested Cormorant	brain	5	0								
	Double-crested Cormorant	egg	34	0								
	Double-crested Cormorant	whole	73	0								
	Common Tern	egg	5	0								
	Forster's Tern	egg	7	0								
	Tree Swallow	whole	15	0								
o,p'-DDE	Double-crested Cormorant	brain	5	0								
	Double-crested Cormorant	egg	34	0								
	Double-crested Cormorant	whole	73	0								
	Common Tern	egg	5	0								
	Forster's Tern	egg	7	0								
	Tree Swallow	whole	15	0								
o,p'-DDT	Double-crested Cormorant	brain	5	0								
	Double-crested Cormorant	egg	34	0								
	Double-crested Cormorant	whole	73	0								
	Common Tern	egg	5	0								
	Forster's Tern	egg	7	0								
	Tree Swallow	whole	15	0								

Table 6-46 Green Bay Zone 2 Bird Tissue Concentrations (Continued)

Analyte	Species	Tissue	Number of Samples	Number of Detects	Detection Frequency (%)	Detected Minimum	Detected Maximum	Mean	Data Distribution	95% UCL	90th Percentile	RME
p,p'-DDD	Double-crested Cormorant	brain	5	0								
	Double-crested Cormorant	egg	34	22	65	10.0	54.0	15.0	Other	20.1	29.0	20.1
	Double-crested Cormorant	whole	73	14	19	10.0	43.0	7.3	Other	8.4	13.6	8.4
	Common Tern	egg	5	5	100	1.0	3.8	2.1	Normal	3.2	0.0	3.2
	Forster's Tern	egg	7	4	57	0.9	2.7	NE	Normal	18.1	0.0	2.7
	Tree Swallow	whole	15	3	20	12.0	13.0	6.5	Other	8.0	13.0	8.0
p,p'-DDE	Double-crested Cormorant	brain	5	5	100	410	670	534	Normal	643	0.0	643
	Double-crested Cormorant	egg	34	34	100	170	11,000	4,132	Other	7,277	8,800	7,277
	Double-crested Cormorant	whole	73	73	100	380	11,000	2,756	Lognormal	3,523	5,060	3,523
	Common Tern	egg	5	5	100	421	942	666	Normal	893	0.0	893
	Forster's Tern	egg	7	7	100	206	735	447	Normal	576	0.0	576
	Tree Swallow	whole	15	15	100	51	380	128	Lognormal	187	326	187
p,p'-DDT	Double-crested Cormorant	brain	5	0								
	Double-crested Cormorant	egg	34	3	9	21.0	47.0	7.6	Other	10.1	13.0	10.1
	Double-crested Cormorant	whole	73	19	26	10.0	41.0	8.1	Other	9.3	18.0	9.3
	Common Tern	egg	5	0								
	Forster's Tern	egg	7	0								
	Tree Swallow	whole	15	0								

Notes:

All tern and tree swallow data are from Kidney Island.

NE indicates that the calculated mean exceeded the detected maximum and, therefore, the mean was not evaluated.

Table 6-47 Estimated Exposure Concentrations for Piscivorous Birds in Green Bay Zone 2

Analyte	Surface Water (µg/L) [1]		TL3 Fish: Alewife (µg/kg)		Common Tern Ingestion				Total Common Tern Ingestion (µg/day)		Total Common Tern Ingestion (µg/kg-BW/day)	
					Surface Water (µg/day) [1]		TL3 Fish: (µg/day)					
	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME
Mercury	0.6	5.0	100	250	0.01	0.1	5.9	14.7	5.9	14.8	49.1	123
Total PCBs	0.02	0.02	2,599	3,182	0.0002	0.0003	153	187	153	187	1,274	1,559
Dieldrin	NA	NA	21.0	57.9	—	—	1.2	3.4	1.2	3.4	10.3	28.4
p,p'-DDE	NA	NA	104	143	—	—	6.1	8.4	6.1	8.4	51.1	70.0

Analyte	Surface Water (µg/L) [1]		TL3 Fish: Alewife (µg/kg)		Forster's Tern Ingestion				Total Forster's Tern Ingestion (µg/day)		Total Forster's Tern Ingestion (µg/kg-BW/day)	
					Surface Water (µg/day) [1]		TL3 Fish: (µg/day)					
	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME
Mercury	0.6	5.0	100	250	0.01	0.1	7.1	17.9	7.2	17.9	45.3	114
Total PCBs	0.02	0.02	2,599	3,182	0.0003	0.0004	186	227	186	227	1,174	1,438
Dieldrin	NA	NA	21.0	57.9	—	—	1.5	4.1	1.5	4.1	9.5	26.2
p,p'-DDE	NA	NA	104	143	—	—	7.4	10.2	7.4	10.2	47.1	64.6

Analyte	Surface Water (µg/L) [1]		TL3 Fish: Alewife (µg/kg)		Double-crested Cormorant Ingestion				Double-crested Cormorant Ingestion (µg/day)		Total Double-crested Cormorant Ingestion (µg/kg-BW/day)	
					Surface Water (µg/day) [1]		TL3 Fish: (µg/day)					
	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME
Mercury	0.6	5.0	100	250	0.1	0.4	31.8	79.5	31.9	79.9	19.0	47.6
Total PCBs	0.02	0.02	2,599	3,182	0.001	0.002	826	1,012	826	1,012	492	602
Dieldrin	NA	NA	21.0	57.9	—	—	6.7	18.4	6.7	18.4	4.0	11.0
p,p'-DDE	NA	NA	104	143	—	—	33.1	45.4	33.1	45.4	19.7	27.0

Notes:

[1] Mercury in unfiltered water, total PCBs on a filtered + particulate basis.

NA indicates that data was not collected for this reach and this contaminant.

"—" indicates that the concentration could not be calculated.

Table 6-47 Estimated Exposure Concentrations for Piscivorous Birds in Green Bay Zone 2 (Continued)

Analyte	Surface Water (µg/L) [1]		TL3 Fish: Carp (µg/kg)		TL4 Fish: Walleye (µg/kg)		Bald Eagle Ingestion						Total Bald Eagle Ingestion (µg/day)		Total Bald Eagle Ingestion (µg/kg-BW/day)	
							Surface Water (µg/day) [1]		TL3 Fish (µg/day)		TL4 Fish (µg/day)					
	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME
Mercury	0.6	5.0	60.0	70.0	210	270	0.1	0.8	25.3	29.5	22.1	28.4	47.5	58.7	10.2	12.6
Total PCBs	0.02	0.02	6,637	7,369	6,539	7,658	0.003	0.003	2,801	3,110	687	804	3,487	3,914	750	842
Dieldrin	NA	NA	20.8	29.4	37.3	52.2	—	—	8.8	12.4	3.9	5.5	12.7	17.9	2.7	3.8
p,p'-DDE	NA	NA	197	700	353	462	—	—	82.9	295	37.1	48.5	120	344	25.8	74.0

Notes:

[1] Mercury in unfiltered water, total PCBs on a filtered + particulate basis.

NA indicates that data was not collected for this reach and this contaminant.

"—" indicates that the concentration could not be calculated.

Model Assumptions:

Common Tern

Food Ingestion =	0.0588	kg/day
Water Ingestion =	0.014	L/day
Body Weight =	0.12	kg

Forster's Tern

Food Ingestion =	0.0714	kg/day
Water Ingestion =	0.017	L/day
Body Weight =	0.158	kg

Double-crested Cormorant

Food Ingestion =	0.318	kg/day
Water Ingestion =	0.084	L/day
Body Weight =	1.68	kg

Bald Eagle

Food Ing. (TL3 Fish) =	0.422	kg/day
Food Ing. (TL4 Fish) =	0.105	kg/day
Water Ingestion =	0.165	L/day
Body Weight =	4.65	kg

Table 6-48 PCB Congeners and Dioxins/Furans in Whole Tree Swallows in Green Bay Zone 2

Analyte	Number of Samples	Number of Detects	Detection Frequency (%)	Detected Minimum	Detected Maximum	Mean	Data Distribution	95% UCL	90th Percentile	RME
PCB Congeners (µg/kg)										
PCB Congener 77	15	1	7	1.3	1.3	0.1	Other	0.3	0.6	0.3
PCB Congener 105	15	15	100	16.0	61	37.8	Normal	44	60	44
PCB Congener 118/106	15	15	100	42.0	120	85.9	Normal	97	120	97
PCB Congener 126	15	8	53	0.3	0.7	0.3	Other	0.8	0.6	0.7
PCB Congener 169	15	0								

Notes:

Tree swallow data are from Kidney Island.

Table 6-49 PCB Congeners and Dioxins/Furans in Double-crested Cormorants in Green Bay Zone 2

Analyte	Tissue	Number of Samples	Number of Detects	Detection Frequency (%)	Detected Minimum	Detected Maximum	Mean	Data Distribution	95% UCL	90th Percentile	RME
PCB Congeners and Dioxins/Furans (µg/kg)											
1,2,3,4,6,7,8,9-OCDD	whole	4	4	100	7.6E-02	2.1E-01	1.4E-01	Normal	2.1E-01	0.0	2.1E-01
1,2,3,4,6,7,8,9-OCDF	whole	4	0								
1,2,3,4,6,7,8-HPCDD	whole	4	0								
1,2,3,4,6,7,8-HPCDF	whole	4	0								
1,2,3,4,7,8,9-HPCDF	whole	4	0								
1,2,3,4,7,8-HXCDD	whole	4	0								
1,2,3,4,7,8-HXCDF	whole	4	0								
1,2,3,6,7,8-HXCDD	whole	4	0								
1,2,3,6,7,8-HXCDF	whole	4	0								
1,2,3,7,8,9-HXCDD	whole	4	0								
1,2,3,7,8,9-HXCDF	whole	4	0								
1,2,3,7,8-PECDD	whole	4	0								
1,2,3,7,8-PECDF	whole	4	0								
2,3,4,6,7,8-HXCDF	whole	4	0								
2,3,4,7,8-PECDF	whole	4	0								
2,3,7,8-TCDD	whole	4	1	25	9.6E-03	9.6E-03	4.7E-03	Other	3.0E-02	0.0	9.6E-03
2,3,7,8-TCDF	whole	4	0								
PCB Congener 77	whole	26	9	35	0.2	2.0	0.3	Other	0.5	1.5	0.5
PCB Congener 105	whole	26	26	100	40.0	530	157	Lognormal	215	429	215
PCB Congener 118/106	whole	26	26	100	88.0	1,200	379	Other	558	1,046	558
PCB Congener 126	whole	26	19	73	0.3	1.5	0.7	Other	2.1	1.4	1.5
PCB Congener 169	whole	26	7	27	0.1	0.2	0.1	Other	0.1	0.2	0.1

Table 6-49 PCB Congeners and Dioxins/Furans in Double-crested Cormorants in Green Bay Zone 2 (Continued)

Analyte	Tissue	Number of Samples	Number of Detects	Detection Frequency (%)	Detected Minimum	Detected Maximum	Mean	Data Distribution	95% UCL	90th Percentile	RME
PCB Congeners and Dioxins/Furans (µg/kg)											
1,2,3,4,6,7,8,9-OCDD	egg	4	0								
1,2,3,4,6,7,8,9-OCDF	egg	4	0								
1,2,3,4,6,7,8-HPCDD	egg	4	0								
1,2,3,4,6,7,8-HPCDF	egg	4	0								
1,2,3,4,7,8,9-HPCDF	egg	4	0								
1,2,3,4,7,8-HXCDD	egg	4	0								
1,2,3,4,7,8-HXCDF	egg	4	0								
1,2,3,6,7,8-HXCDD	egg	4	0								
1,2,3,6,7,8-HXCDF	egg	4	0								
1,2,3,7,8,9-HXCDD	egg	4	0								
1,2,3,7,8,9-HXCDF	egg	4	0								
1,2,3,7,8-PECDD	egg	4	0								
1,2,3,7,8-PECDF	egg	4	0								
2,3,4,6,7,8-HXCDF	egg	4	0								
2,3,4,7,8-PECDF	egg	4	0								
2,3,7,8-TCDD	egg	4	1	25	0.02	0.02	0.01	Other	0.04	0.0	0.02
2,3,7,8-TCDF	egg	4	0								
PCB Congener 77	egg	12	9	75	1.4	2.3	1.3	Other	18.2	2.2	2.3
PCB Congener 105	egg	12	12	100	14.0	630	210	Normal	303	558	303
PCB Congener 118/106	egg	12	12	100	37.0	1,600	551	Normal	783	1,414	783
PCB Congener 126	egg	12	11	92	0.2	2.3	1.1	Normal	1.5	2.3	1.5
PCB Congener 169	egg	12	5	42	0.1	0.4	0.1	Other	0.2	0.4	0.2

Table 6-50 PCB Congeners and Dioxins/Furans in Common Tern Eggs in Green Bay Zone 2

Analyte	Number of Samples	Number of Detects	Detection Frequency (%)	Detected Minimum	Detected Maximum	Mean	Data Distribution	95% UCL	90th Percentile	RME
PCB Congeners and Dioxins/Furans (µg/kg)										
1,2,3,4,6,7,8,9-OCDD	5	5	100	6.4E-02	1.9E-01	1.0E-01	Normal	1.5E-01	0.0	1.5E-01
1,2,3,4,6,7,8,9-OCDF	1	0								
1,2,3,4,6,7,8-HPCDD	5	5	100	7.2E-03	1.6E-02	1.2E-02	Normal	1.5E-02	0.0	1.5E-02
1,2,3,4,6,7,8-HPCDF	5	4	80	8.5E-04	2.5E-03	1.5E-03	Normal	2.4E-03	0.0	2.4E-03
1,2,3,4,7,8-HXCDD	5	2	40	1.2E-03	2.2E-03	1.3E-03	Normal	1.8E-03	0.0	1.8E-03
1,2,3,4,7,8-HXCDF	5	4	80	9.4E-04	2.0E-03	1.3E-03	Normal	1.9E-03	0.0	1.9E-03
1,2,3,6,7,8-HXCDD	5	5	100	6.9E-03	1.7E-02	1.2E-02	Normal	1.6E-02	0.0	1.6E-02
1,2,3,6,7,8-HXCDF	5	2	40	2.0E-03	2.1E-03	1.3E-03	Normal	2.0E-03	0.0	2.0E-03
1,2,3,7,8,9-HXCDD	5	5	100	1.1E-03	2.4E-03	1.9E-03	Normal	2.3E-03	0.0	2.3E-03
1,2,3,7,8-PECDD	5	4	80	5.6E-03	9.3E-03	6.2E-03	Normal	8.8E-03	0.0	8.8E-03
1,2,3,7,8-PECDF	4	2	50	1.2E-03	1.5E-03	9.5E-04	Normal	1.5E-03	0.0	1.5E-03
2,3,4,6,7,8-HXCDF	4	3	75	6.2E-04	1.2E-03	8.0E-04	Normal	1.1E-03	0.0	1.1E-03
2,3,4,7,8-PECDF	5	4	80	4.2E-03	8.6E-03	5.6E-03	Normal	7.8E-03	0.0	7.8E-03
2,3,7,8-TCDD	5	5	100	1.7E-03	4.7E-03	3.2E-03	Normal	4.4E-03	0.0	4.4E-03
2,3,7,8-TCDF	5	5	100	1.1E-02	2.0E-02	1.6E-02	Normal	2.0E-02	0.0	2.0E-02
PCB Congener 77	10	6	60	3.9	10.4	5.2	Normal	7.28	10.2	7.3
PCB Congener 81	10	6	60	0.7	1.4	NE	Lognormal	16.54	15.5	1.4
PCB Congener 105	10	10	100	47.0	177	109	Normal	131.71	174	132
PCB Congener 118	10	10	100	122	689	357	Normal	452.30	668	452
PCB Congener 126	10	6	60	0.8	2.0	NE	Lognormal	6.45	9.3	2.0
PCB Congener 169	10	5	50	0.1	0.2	NE	Lognormal	127.82	9.3	0.2

Notes:

Data is from Kidney Island.

NE indicates that the calculated mean exceeded the detected maximum and, therefore, the mean was not evaluated.

Table 6-51 PCB Congeners and Dioxins/Furans in Forster's Tern Eggs in Green Bay Zone 2

Analyte	Number of Samples	Number of Detects	Detection Frequency (%)	Detected Minimum	Detected Maximum	Mean	Data Distribution	95% UCL	90th Percentile	RME
PCB Congeners and Dioxins/Furans (µg/kg)										
1,2,3,4,6,7,8,9-OCDD	7	7	100	2.8E-01	7.3E-01	5.3E-01	Normal	6.4E-01	0.0	6.4E-01
1,2,3,4,6,7,8,9-OCDF	3	1	33	1.0E-03	1.0E-03	7.6E-04	Normal	1.1E-03	0.0	1.0E-03
1,2,3,4,6,7,8-HPCDD	7	7	100	4.7E-03	1.3E-02	8.7E-03	Normal	1.1E-02	0.0	1.1E-02
1,2,3,4,6,7,8-HPCDF	7	3	43	4.2E-04	6.6E-04	4.2E-04	Normal	5.2E-04	0.0	5.2E-04
1,2,3,4,7,8-HXCDD	5	5	100	6.6E-04	2.0E-03	9.7E-04	Other	1.9E-03	0.0	1.9E-03
1,2,3,4,7,8-HXCDF	3	3	100	3.1E-04	8.1E-04	5.8E-04	Normal	1.0E-03	0.0	8.1E-04
1,2,3,6,7,8-HXCDD	7	6	86	3.7E-03	1.2E-02	6.4E-03	Normal	8.9E-03	0.0	8.9E-03
1,2,3,6,7,8-HXCDF	7	2	29	9.4E-04	9.4E-04	9.6E-04	Lognormal	1.5E-03	0.0	9.4E-04
1,2,3,7,8,9-HXCDD	5	4	80	7.9E-04	1.5E-03	1.1E-03	Normal	1.5E-03	0.0	1.5E-03
1,2,3,7,8-PECDD	7	6	86	1.6E-03	4.4E-03	2.5E-03	Normal	3.3E-03	0.0	3.3E-03
1,2,3,7,8-PECDF	1	1	100	9.3E-04	9.3E-04	—	—	—	0.0	9.3E-04
2,3,4,7,8-PECDF	5	2	40	4.1E-04	8.8E-04	3.9E-04	Normal	6.6E-04	0.0	6.6E-04
2,3,7,8-TCDD	7	7	100	2.0E-03	5.1E-03	3.3E-03	Normal	4.3E-03	0.0	4.3E-03
2,3,7,8-TCDF	7	7	100	9.4E-04	2.3E-03	1.3E-03	Lognormal	1.7E-03	0.0	1.7E-03
PCB Congener 77	9	5	56	1.2	3.3	2.6	Lognormal	9.2	0.0	3.3
PCB Congener 81	9	5	56	0.5	1.3	NE	Lognormal	90.4	0.0	1.3
PCB Congener 105	10	10	100	34.0	158	93.4	Normal	113	154	113
PCB Congener 118	10	10	100	84.1	421	283	Normal	348	419	348
PCB Congener 126	9	5	56	0.3	0.7	NE	Lognormal	52.4	0.0	0.7
PCB Congener 169	10	2	20	0.1	0.8	NE	Lognormal	30.9	14.5	0.8

Notes:

Data is from Kidney Island.

NE indicates that the calculated mean exceeded the detected maximum and, therefore, the mean was not evaluated.

Table 6-52 Estimated Exposure Concentrations for Mink in Green Bay Zone 2

Analyte	Surface Water (µg/L) [1]		Surface Sediment (µg/kg) [2]		Whole Carp (µg/kg)		Mink Ingestion						Total Mink Ingestion (µg/day)		Total Mink Ingestion (µg/kg-BW/day)	
							Surface Water (µg/day) [1]		Surface Sediment (µg/day)		Whole Carp (µg/day)					
	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME
Mercury	0.6	5.0	500	1,500	60.0	70.0	0.1	0.4	2.0	6.0	9.2	10.7	11.2	17.1	14.0	21.4
Total PCBs (N)	0.02	0.02	251	720	6,637	7,369	0.001	0.002	1.0	2.9	1,015	1,127	1,016	1,130	1,271	1,413
Total PCBs (I_d)	0.02	0.02	1,132	1,154	6,637	7,369	0.001	0.002	4.5	4.6	1,015	1,127	1,020	1,132	1,275	1,415
Dieldrin	NA	NA	2.8	3.3	20.8	29.4	—	—	0.01	0.01	3.2	4.5	3.2	4.5	4.0	5.6
p,p'-DDE	NA	NA	2.8	3.3	197	700	—	—	0.01	0.01	30.1	107	30.1	107	37.6	134

Notes:

[1] Mercury in unfiltered water, total PCBs on a filtered + particulate basis.

[2] p,p'-DDT rather than p,p'-DDE was used because this was the predominant form in the sediment.

NA indicates that data was not collected for this reach and this contaminant.

Shaded areas indicate that the concentration is estimated as half of the detection limit.

"—" indicates that the concentration could not be calculated.

Model Assumptions:

Food Ingestion =	0.153	kg/day
Water Ingestion =	0.081	L/day
Sediment Ingestion =	0.004	kg/day
Body Weight =	0.8	kg

Table 6-53 Surface Water Concentrations in Green Bay Zone 3A

Analyte	Number of Samples	Number of Detects	Detection Frequency (%)	Detected Minimum	Detected Maximum	Mean	Data Distribution	95% UCL	90th Percentile	RME	Criteria		Reference	Hazard Quotients			
											NOAEC	LOAEC		Mean NOEL	Mean LOEL	RME NOEL	RME LOEL
Metals (ng/L)																	
Mercury (filtered)	6	0															
Mercury (unfiltered)	6	0															
PCBs (ng/L)																	
Total PCBs (filtered)	60	60	100	0.5	5.1	1.6	Lognormal	1.9	3.2	1.9	50	500	[1]	< 0.1	< 0.1	< 0.1	< 0.1
Total PCBs (particulate)	66	61	92	0.2	16.9	2.8	Lognormal	3.7	6.7	3.7	50	500	[1]	0.1	< 0.1	0.1	< 0.1
Total PCBs (filtered + particulate)				0.7	22.1	4.4		5.6	9.9	5.6	50	500	[1]	0.1	< 0.1	0.1	< 0.1

Note:

[1] Niimi, 1996.

Table 6-54 Surface Sediment Concentrations in Green Bay Zone 3A

Analyte	Number of Samples	Number of Detects	Detection Frequency (%)	Detected Minimum	Detected Maximum	Mean	Data Distribution	95% UCL	90th Percentile	RME	Criteria	Reference	Hazard Quotients	
													Mean	RME
Metals (mg/kg)														
Arsenic	2	2	100	1.4	1.6	1.5	Normal	2.1	0.0	1.6	12.1	[1]	0.1	0.1
Lead	2	2	100	1.1	1.9	1.5	Normal	4.0	0.0	1.9	34.2	[1]	< 0.1	0.1
Mercury	2	0												
PCBs (µg/kg)														
Total PCBs (N)	15	13	87	6.0	993	376	Normal	518	862	518	31.6	[1]	12	16
Total PCBs (I_d)	81,496	81,496	100	5.3	1,026	256	Other	257	621	257	31.6	[1]	8.1	8.1
Pesticides (µg/kg)														
Dieldrin	2	0												
p,p'-DDD	2	0												
p,p'-DDE	2	0												
p,p'-DDT	2	0												

Notes:

N indicates that the data was not interpolated based on depth.

I_0 indicates that interpolated grid areas for which "no values" existed were assumed to equal zero.

I_d indicates that interpolated grid areas for which "no values" existed were deleted from the database.

[1] ARCS SEC (EPA, 1996a).

Table 6-55 PCB Congener Concentrations in Surface Sediment in Green Bay Zone 3A

Analyte	Number of Samples	Number of Detects	Detection Frequency (%)	Detected Minimum	Detected Maximum	Mean	Data Distribution	95% UCL	90th Percentile	RME
PCBs (µg/kg)										
PCB Congener 77	2	2	100	0.02	0.1	0.04	Normal	0.2	0.0	0.1
PCB Congener 81	15	14	93	0.01	1.2	0.4	Normal	0.6	1.1	0.6
PCB Congener 105	2	1	50	1.6	1.6	0.8	Normal	5.8	0.0	1.6
PCB Congener 118	15	11	73	0.03	25.4	6.2	Lognormal	211	20.0	25.4
PCB Congener 126	2	0								
PCB Congener 169	2	0								

Table 6-56 Green Bay Zone 3A Whole Fish Concentrations

Analyte	Species	Number of Samples	Number of Detects	Detection Frequency (%)	Detected Minimum	Detected Maximum	Mean	Data Distribution	95% UCL	90th Percentile	RME
Metals (mg/kg)											
Mercury	Gizzard Shad	1	0								
	Rainbow Smelt	6	4	67	0.02	0.05	0.03	Normal	0.04	0.0	0.04
	Yellow Perch	2	0								
	Carp	1	0								
PCBs (µg/kg)											
Total PCBs	Alewife	18	18	100	280	2,700	907	Lognormal	1,271	1,800	1,271
	Gizzard Shad	1	1	100	3,524	3,524	NA			0.0	3,524
	Rainbow Smelt	32	31	97	210	1,300	570	Lognormal	735	997	735
	Yellow Perch	2	2	100	107	251	179	Normal	634	0.0	251
	Carp	11	11	100	249	7,900	2,642	Normal	3,974	7,180	3,974
	Walleye	14	14	100	980	7,500	4,155	Normal	5,064	7,000	5,064
	Brown Trout	14	14	100	1,800	4,400	3,250	Normal	3,612	4,250	3,612
Pesticides (µg/kg)											
Dieldrin	Alewife	18	18	100	7.3	60.0	21.5	Lognormal	27.5	33.0	27.5
	Gizzard Shad	1	0								
	Rainbow Smelt	32	23	72	3.1	30.0	14.4	Lognormal	17.5	28.4	17.5
	Yellow Perch	2	0								
	Carp	11	10	91	3.8	70.0	17.9	Lognormal	54.6	61.4	54.6
	Walleye	10	10	100	5.3	87.0	43.4	Normal	57.7	84.4	57.7
	Brown Trout	14	14	100	2.3	100	76.0	Other	212	100	100
o,p'-DDD	Rainbow Smelt	12	0								
o,p'-DDE	Rainbow Smelt	11	0								
o,p'-DDT	Rainbow Smelt	12	0								
p,p'-DDD	Gizzard Shad	1	0								
	Rainbow Smelt	12	0								
	Yellow Perch	2	0								
	Carp	1	0								
p,p'-DDE	Gizzard Shad	1	1	100	150	150	NA			0.0	150
	Rainbow Smelt	12	2	17	50.0	60.0	30.0	Other	36.2	57.0	36.2
	Yellow Perch	2	1	50	9.0	9.0	6.0	Normal	24.9	0.0	9.0
	Carp	1	1	100	25.0	25.0	25.0			0.0	25.0
p,p'-DDT	Gizzard Shad	1	0								
	Rainbow Smelt	12	0								
	Yellow Perch	2	0								
	Carp	1	0								

Note:
NA - Not applicable.

Table 6-57 PCB Congener Concentrations in Green Bay Zone 3A Whole Fish

Analyte	Species	Number of Samples	Number of Detects	Detection Frequency (%)	Detected Minimum	Detected Maximum	Mean	Data Distribution	95% UCL	90th Percentile	RME
PCBs (µg/kg)											
PCB Congener 77/110	Alewife	18	18	100	13.0	100	39.5	Lognormal	53.1	75.7	53.1
PCB Congener 81	Alewife	18	18	100	1.4	8.1	3.5	Normal	4.2	6.9	4.2
PCB Congener 132/153/105	Alewife	18	18	100	11.0	93.0	45.7	Normal	56.5	84.9	56.5
PCB Congener 118	Alewife	18	18	100	9.2	68.0	29.5	Normal	36.5	59.0	36.5
PCB Congener 77	Gizzard Shad	1	1	100	1.9	1.9	NA			0.0	1.9
PCB Congener 81	Gizzard Shad	1	0								
PCB Congener 105	Gizzard Shad	1	1	100	54.0	54.0	NA			0.0	54.0
PCB Congener 118	Gizzard Shad	1	1	100	123	123	NA			0.0	123
PCB Congener 126	Gizzard Shad	1	1	100	0.7	0.7	NA			0.0	0.7
PCB Congener 169	Gizzard Shad	1	0								
PCB Congener 77/110	Rainbow Smelt	20	20	100	8.4	65.0	22.7	Lognormal	30.2	43.7	30.2
PCB Congener 81	Rainbow Smelt	20	20	100	1.0	5.4	2.6	Normal	3.1	4.6	3.1
PCB Congener 132/153/105	Rainbow Smelt	20	20	100	11.0	46.0	26.3	Normal	31.0	43.9	31.0
PCB Congener 118	Rainbow Smelt	20	20	100	7.5	38.0	17.7	Lognormal	23.0	34.5	23.0
PCB Congener 77	Yellow Perch	2	1	50	0.1	0.1	0.1	Normal	0.5	0.0	0.1
PCB Congener 81	Yellow Perch	2	0								
PCB Congener 105	Yellow Perch	2	2	100	1.5	4.0	2.8	Normal	10.6	0.0	4.0
PCB Congener 118	Yellow Perch	2	2	100	4.5	12.0	8.3	Normal	31.9	0.0	12.0
PCB Congener 126	Yellow Perch	2	0								
PCB Congener 169	Yellow Perch	2	0								
PCB Congener 77	Carp	1	1	100	0.1	0.1	0.1			0.0	0.1
PCB Congener 81	Carp	11	9	82	1.1	26.0	8.8	Other	7133	25.2	26.0
PCB Congener 105	Carp	1	1	100	4.5	4.5	4.5			0.0	4.5
PCB Congener 118	Carp	11	11	100	8.8	230	94.2	Lognormal	615	224	230
PCB Congener 126	Carp	1	0								
PCB Congener 169	Carp	1	1	100	0.0	0.0	0.0			0.0	0.0
PCB Congener 77	Walleye	1	1	100	8.7	8.7	NA			0.0	8.7
PCB Congener 81	Walleye	11	10	91	0.6	25.0	11.4	Normal	15.5	23.2	15.5
PCB Congener 105	Walleye	4	4	100	48.5	71.7	63.3	Normal	75.6	0.0	71.7
PCB Congener 118	Walleye	14	14	100	34.0	200	125	Normal	150	196	150
PCB Congener 126	Walleye	1	1	100	0.9	0.9	NA			0.0	0.9
PCB Congener 169	Walleye	4	4	100	0.1	2.4	1.6	Normal	2.9	0.0	2.4
PCB Congener 77/110	Brown Trout	14	14	100	46.0	200	134	Normal	153	195	153
PCB Congener 81	Brown Trout	14	14	100	3.8	21.0	12.3	Normal	14.4	20.5	14.4
PCB Congener 132/153/105	Brown Trout	14	14	100	35.0	270	170	Normal	199	265	199
PCB Congener 118	Brown Trout	14	14	100	27.0	160	111	Normal	127	160	127

Note:
NA - Not applicable.

Table 6-58 Green Bay Zone 3A Bird Tissue Concentrations

Analyte	Species	Tissue	Number of Samples	Number of Detects	Detection Frequency (%)	Detected Minimum	Detected Maximum	Mean	Data Distribution	95% UCL	90th Percentile	RME
Metals (mg/kg)												
Mercury	Bald Eagle	egg	3	3	100	0.3	0.3	0.3	Other	0.3	0.0	0.3
PCBs (µg/kg)												
Total PCBs	Bald Eagle	egg	1	1	100	13,000	13,000	NA	—	—	0.0	13,000
Pesticides (µg/kg)												
Dieldrin	Bald Eagle	egg	1	1	100	200	200	NA	—	—	0.0	200
p,p'-DDD	Bald Eagle	egg	1	1	100	120	120	NA	—	—	0.0	120
p,p'-DDE	Bald Eagle	egg	1	1	100	2,400	2,400	NA	—	—	0.0	2,400
p,p'-DDT	Bald Eagle	egg	1	0								

Note:
NA - Not applicable.

Table 6-59 Estimated Exposure Concentrations for Piscivorous Birds in Green Bay Zone 3A

Analyte	Surface Water (µg/L) [1]		TL3 Fish: Alewife and Rainbow Smelt (µg/kg)		Common Tern Ingestion				Total Common Tern Ingestion (µg/day)		Total Common Tern Ingestion (µg/kg-BW/day)	
					Surface Water (µg/day) [1]		TL3 Fish (µg/day)					
	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME
Mercury	0.02	0.03	30.0	40.0	0.0003	0.0004	1.8	2.4	1.8	2.4	14.7	19.6
Total PCBs	0.004	0.006	907	1,271	0.0001	0.0001	53.3	74.8	53.3	74.8	444	623
Dieldrin	NA	NA	21.5	27.5	—	—	1.3	1.6	1.3	1.6	10.5	13.5
p,p'-DDE	NA	NA	30.0	36.2	—	—	1.8	2.1	1.8	2.1	14.7	17.7

Analyte	Surface Water (µg/L) [1]		TL3 Fish: Alewife and Rainbow Smelt (µg/kg)		Forster's Tern Ingestion				Total Forster's Tern Ingestion (µg/day)		Total Forster's Tern Ingestion (µg/kg-BW/day)	
					Surface Water (µg/day) [1]		TL3 Fish (µg/day)					
	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME
Mercury	0.02	0.03	30.0	40.0	0.0003	0.0005	2.1	2.9	2.1	2.9	13.6	18.1
Total PCBs	0.004	0.006	907	1,271	0.0001	0.0001	64.7	90.8	64.7	90.8	410	575
Dieldrin	NA	NA	21.5	27.5	—	—	1.5	2.0	1.5	2.0	9.7	12.4
p,p'-DDE	NA	NA	30.0	36.2	—	—	2.1	2.6	2.1	2.6	13.6	16.3

Analyte	Surface Water (µg/L) [1]		TL3 Fish: Alewife and Rainbow Smelt (µg/kg)		Double-crested Cormorant Ingestion				Double-crested Cormorant Ingestion (µg/day)		Total Double-crested Cormorant Ingestion (µg/kg-BW/day)	
					Surface Water (µg/day) [1]		TL3 Fish (µg/day)					
	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME
Mercury	0.02	0.03	30.0	40.0	0.002	0.002	9.5	12.7	9.5	12.7	5.7	7.6
Total PCBs	0.004	0.006	907	1,271	0.0004	0.0005	288	404	288	404	172	241
Dieldrin	NA	NA	21.5	27.5	—	—	6.8	8.7	6.8	8.7	4.1	5.2
p,p'-DDE	NA	NA	30.0	36.2	—	—	9.5	11.5	9.5	11.5	5.7	6.8

Notes:

[1] Mercury in unfiltered water, total PCBs on a filtered + particulate basis.

NA indicates that data was not collected for this reach and this contaminant.

Shaded areas indicate that the concentration is estimated as half of the detection limit.

"—" indicates that the concentration could not be calculated.

Table 6-59 Estimated Exposure Concentrations for Piscivorous Birds in Green Bay Zone 3A (Continued)

Analyte	Surface Water (µg/L) [1]		TL3 Fish: Carp (µg/kg)		TL4 Fish: Walleye (µg/kg)		Bald Eagle Ingestion						Total Bald Eagle Ingestion (µg/day)		Total Bald Eagle Ingestion (µg/kg-BW/day)	
							Surface Water (µg/day) [1]		TL3 Fish (µg/day)		TL4 Fish (µg/day)					
	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME
Mercury	0.020	0.028	25.0	50.0	NA	NA	0.003	0.005	10.6	21.1	—	—	10.6	21.1	2.3	4.5
Total PCBs	0.004	0.006	2,642	3,974	4,155	5,064	0.001	0.001	1,115	1,677	436	532	1,551	2,209	334	475
Dieldrin	NA	NA	17.9	54.6	43.4	57.7	—	—	7.5	23.0	4.6	6.1	12.1	29.1	2.6	6.3
p,p'-DDE	NA	NA	25.0	25.0	NA	NA	—	—	10.6	10.6	—	—	10.6	10.6	2.3	2.3

Notes:

[1] Mercury in unfiltered water, total PCBs on a filtered + particulate basis.

NA indicates that data was not collected for this reach and this contaminant.

Shaded areas indicate that the concentration is estimated as half of the detection limit.

"—" indicates that the concentration could not be calculated.

Model Assumptions:

Common Tern

Food Ingestion =	0.0588	kg/day
Water Ingestion =	0.014	L/day
Body Weight =	0.12	kg

Forster's Tern

Food Ingestion =	0.0714	kg/day
Water Ingestion =	0.017	L/day
Body Weight =	0.158	kg

Double-crested Cormorant

Food Ingestion =	0.318	kg/day
Water Ingestion =	0.084	L/day
Body Weight =	1.68	kg

Bald Eagle

Food Ing. (TL3 Fish) =	0.422	kg/day
Food Ing. (TL4 Fish) =	0.105	kg/day
Water Ingestion =	0.165	L/day
Body Weight =	4.65	kg

Table 6-60 Estimated Exposure Concentrations for Mink in Green Bay Zone 3A

Analyte	Surface Water (µg/L) [1]		Surface Sediment (µg/kg) [2]		Whole Carp (µg/kg)		Mink Ingestion						Total Mink Ingestion (µg/day)		Total Mink Ingestion (µg/kg-BW/day)	
							Surface Water (µg/day) [1]		Surface Sediment (µg/day)		Whole Carp (µg/day)					
	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME
Mercury	0.02	0.03	15.0	15.0	25.0	50.0	0.002	0.002	0.1	0.1	3.8	7.7	3.9	7.7	4.9	9.6
Total PCBs (N)	0.004	0.006	376	518	2,642	3,974	0.0004	0.0005	1.5	2.1	404	608	406	610	507	763
Total PCBs (I_d)	0.004	0.006	256	257	2,642	3,974	0.0004	0.0005	1.0	1.0	404	608	405	609	507	761
Dieldrin	NA	NA	1.7	1.8	17.9	54.6	—	—	0.01	0.01	2.7	8.4	2.7	8.4	3.4	10.5
p,p'-DDE	NA	NA	1.7	1.8	25.0	25.0	—	—	0.01	0.01	3.8	3.8	3.8	3.8	4.8	4.8

Notes:

[1] Mercury in unfiltered water, total PCBs on a filtered + particulate basis.

[2] p,p'-DDT rather than p,p'-DDE was used because this was the predominant form in the sediment.

NA indicates that data was not collected for this reach and this contaminant.

Shaded areas indicate that the concentration is estimated as half of the detection limit.

"—" indicates that the concentration could not be calculated.

Model Assumptions:

Food Ingestion =	0.153	kg/day
Water Ingestion =	0.081	L/day
Sediment Ingestion =	0.004	kg/day
Body Weight =	0.8	kg

Table 6-61 Surface Water Concentrations in Green Bay Zone 3B

Analyte	Number of Samples	Number of Detects	Detection Frequency (%)	Detected Minimum	Detected Maximum	Mean	Data Distribution	95% UCL	90th Percentile	RME	Criteria NOAEC	Criteria LOAEC	Reference	Hazard Quotients Mean NOEL	Hazard Quotients Mean LOEL	Hazard Quotients RME NOEL	Hazard Quotients RME LOEL
Metals (ng/L)																	
Mercury (filtered)	7	0															
Mercury (unfiltered)	7	1	14	90.0	90.0	47.3	Normal	65.4	0.0	65.4	440		[1]	0.1		0.1	
PCBs (ng/L)																	
Total PCBs (filtered)	40	40	100	0.5	3.9	1.4	Lognormal	1.7	2.6	1.7	50	500	[2]	< 0.1	< 0.1	< 0.1	< 0.1
Total PCBs (particulate)	45	40	89	0.3	9.4	2.2	Other	3.5	4.7	3.5	50	500	[2]	< 0.1	< 0.1	0.1	< 0.1
Total PCBs (filtered + particulate)				0.8	13.4	3.7		5.2	7.3	5.2	50	500	[2]	0.1	< 0.1	0.1	< 0.1

Notes:

[1] Wisconsin Administrative Code Chapter NR 105.

[2] Niimi, 1996.

Table 6-62 Surface Sediment Concentrations in Green Bay Zone 3B

Analyte	Number of Samples	Number of Detects	Detection Frequency (%)	Detected Minimum	Detected Maximum	Mean	Data Distribution	95% UCL	90th Percentile	RME	Criteria	Reference	Hazard Quotients Mean	Hazard Quotients RME
Metals (mg/kg)														
Arsenic	4	4	100	3.6	15.0	8.6	Normal	14.1	0.0	14.1	12.1	[1]	0.7	1.2
Lead	4	4	100	9.6	50.0	29.9	Normal	49.4	0.0	49.4	34.2	[1]	0.9	1.4
Mercury	4	1	25	0.2	0.2	0.1	Normal	0.2	0.0	0.2	0.17	[2]	0.6	1.1
PCBs (µg/kg)														
Total PCBs (N)	40	35	88	50.0	1,056	542	Other	809	963	809	31.6	[1]	17	26
Total PCBs (I_d)	68,378	68,378	100	19.6	964	482	Other	483	722	483	31.6	[1]	15	15
Pesticides (µg/kg)														
Dieldrin	4	0												
p,p'-DDD	4	0												
p,p'-DDE	4	0												
p,p'-DDT	4	0												

Notes:

N indicates that the data was not interpolated based on depth.

I_0 indicates that interpolated grid areas for which "no values" existed were assumed to equal zero.

I_d indicates that interpolated grid areas for which "no values" existed were deleted from the database.

[1] ARCS SEC (EPA, 1996a).

[2] Environment Canada TEL (Smith *et al.,* 1996); p,p'-DDT based on total DDT TEL.

Table 6-63 PCB Congener Concentrations in Surface Sediment in Green Bay Zone 3B

Analyte	Number of Samples	Number of Detects	Detection Frequency (%)	Detected Minimum	Detected Maximum	Mean	Data Distribution	95% UCL	90th Percentile	RME
PCBs (µg/kg)										
PCB Congener 77	4	4	100	0.3	1.4	0.6	Other	11.7	0.0	1.4
PCB Congener 81	37	32	86	0.05	1.5	0.5	Other	0.8	1.0	0.8
PCB Congener 105	4	4	100	0.3	1.1	0.6	Normal	1.0	0.0	1.0
PCB Congener 118	37	33	89	0.4	31.0	12.4	Other	36.6	25.5	31.0
PCB Congener 126	4	0								
PCB Congener 169	4	0								

Table 6-64 Green Bay Zone 3B Whole Fish Concentrations

Analyte	Species	Number of Samples	Number of Detects	Detection Frequency (%)	Detected Minimum	Detected Maximum	Mean	Data Distribution	95% UCL	90th Percentile	RME
Metals (mg/kg)											
Mercury	Alewife	1	0								
	Gizzard Shad	1	0								
	Yellow Perch	2	0								
	Carp	2	1	50	0.2	0.2	0.1	Normal	0.5	0.0	0.2
	Walleye	3	1	33	0.65	0.65	0.25	Other	2.88E+08	0.0	0.65
PCBs (µg/kg)											
Total PCBs	Alewife	8	8	100	536	2,800	1,821	Normal	2,375	0.0	2,375
	Gizzard Shad	1	1	100	635	635	NA			0.0	635
	Rainbow Smelt	20	20	100	250	1,500	733	Normal	861	1,190	861
	Yellow Perch	2	2	100	138	169	154	Normal	251	0.0	169
	Carp	14	14	100	46.0	8,500	4,947	Normal	6,149	8,200	6,149
	Walleye	26	26	100	212	20,031	6,429	Other	11,741	12,421	11,741
	Brown Trout	26	26	100	75.0	6,700	2,223	Normal	2,697	3,690	2,697
Pesticides (µg/kg)											
Dieldrin	Alewife	8	7	88	12.0	46.0	19.1	Normal	27.3	0.0	27.3
	Gizzard Shad	1	0								
	Rainbow Smelt	20	20	100	3.1	42.0	14.7	Normal	18.4	27.8	18.4
	Yellow Perch	2	0								
	Carp	14	12	86	26.0	88.0	43.2	Normal	54.6	85.5	54.6
	Walleye	15	12	80	29.0	110	50.1	Normal	63.3	95.0	63.3
	Brown Trout	12	12	100	25.0	99.0	72.0	Normal	83.1	98.1	83.1
p,p'-DDD	Alewife	1	0								
	Gizzard Shad	1	0								
	Yellow Perch	2	0								
	Carp	2	0								
	Walleye	3	0								
p,p'-DDE	Alewife	1	1	100	80.0	80.0	NA			0.0	80.0
	Gizzard Shad	1	1	100	37.0	37.0	NA			0.0	37.0
	Yellow Perch	2	2	100	19.0	23.0	21.0	Normal	33.6	0.0	23.0
	Carp	2	2	100	12.0	240	126	Normal	846	0.0	240
	Walleye	3	2	67	64.0	540	207	Normal	695	0.0	540
p,p'-DDT	Alewife	1	0								
	Gizzard Shad	1	0								
	Yellow Perch	2	0								
	Carp	2	0								
	Walleye	3	0								

Note:
NA - Not applicable.

Table 6-65 PCB Congener Concentrations in Green Bay Zone 3B Whole Fish

Analyte	Species	Number of Samples	Number of Detects	Detection Frequency (%)	Detected Minimum	Detected Maximum	Mean	Data Distribution	95% UCL	90th Percentile	RME
PCBs (µg/kg)											
PCB Congener 77	Alewife	1	1	100	0.1	0.1	NA				0.1
PCB Congener 81	Alewife	8	7	88	4.4	7.9	5.7	Other	148,425	0.0	7.9
PCB Congener 105	Alewife	1	1	100	13.0	13.0	NA				13.0
PCB Congener 118	Alewife	8	8	100	28.0	68.0	52.0	Normal	62.5	0.0	62.5
PCB Congener 126	Alewife	1	0								
PCB Congener 169	Alewife	1	0								
PCB Congener 77	Gizzard Shad	1	1	100	0.2	0.2	NA			0.0	0.2
PCB Congener 81	Gizzard Shad	1	0								
PCB Congener 105	Gizzard Shad	1	1	100	12.0	12.0	NA			0.0	12.0
PCB Congener 118	Gizzard Shad	1	1	100	32.0	32.0	NA			0.0	32.0
PCB Congener 126	Gizzard Shad	1	0								
PCB Congener 169	Gizzard Shad	1	0								
PCB Congener 77/110	Rainbow Smelt	20	20	100	9.0	66.0	29.8	Normal	35.5	55.6	35.5
PCB Congener 81	Rainbow Smelt	20	19	95	1.2	5.8	2.8	Normal	3.3	4.8	3.3
PCB Congener 132/153/105	Rainbow Smelt	20	20	100	12.0	54.0	31.2	Normal	36.5	52.3	36.5
PCB Congener 118	Rainbow Smelt	20	20	100	7.2	41.0	22.3	Normal	26.2	37.7	26.2
PCB Congener 77	Yellow Perch	2	0								
PCB Congener 81	Yellow Perch	2	0								
PCB Congener 105	Yellow Perch	2	2	100	3.4	4.3	3.9	Normal	6.7	0.0	4.3
PCB Congener 118	Yellow Perch	2	2	100	3.4	10.0	6.7	Normal	27.5	0.0	10.0
PCB Congener 126	Yellow Perch	2	0								
PCB Congener 169	Yellow Perch	2	0								
PCB Congener 77	Carp	1	1	100	1.7	1.7	1.7				1.7
PCB Congener 81	Carp	13	11	85	10.0	31.0	15.7	Normal	20.2	29.8	20.2
PCB Congener 105	Carp	2	2	100	0.2	29.0	14.6	Normal	106	0.0	29.0
PCB Congener 118	Carp	14	14	100	2.6	280	155	Normal	190	260.0	190
PCB Congener 126	Carp	1	0								
PCB Congener 169	Carp	1	0								
PCB Congener 77	Walleye	4	4	100	0.03	4.9	2.5	Lognormal	0.0	0.0	4.9
PCB Congener 81	Walleye	16	13	81	2.2	24.0	11.0	Normal	14.7	23.3	14.7
PCB Congener 105	Walleye	13	13	100	4.8	268	103	Normal	134	222	134
PCB Congener 118	Walleye	25	25	100	13.0	983	227	Lognormal	370	510	370
PCB Congener 126	Walleye	4	2	50	0.4	0.6	0.2	Normal	0.6	0.0	0.6
PCB Congener 169	Walleye	12	9	75	0.1	7.1	3.5	Other	124,930	6.8	7.1
PCB Congener 77	Brown Trout	1	1	100	3.5	3.5	NA			0.0	3.5
PCB Congener 81	Brown Trout	13	13	100	0.6	19.0	11.2	Normal	13.5	18.2	13.5
PCB Congener 105	Brown Trout	5	5	100	32.9	47.9	38.9	Normal	44.9	0.0	44.9
PCB Congener 118	Brown Trout	17	17	100	70.0	163	107	Normal	117	145	117
PCB Congener 126	Brown Trout	1	1	100	0.5	0.5	NA			0.0	0.5
PCB Congener 169	Brown Trout	3	3	100	0.1	1.3	0.9	Normal	2.1	0.0	1.3

Note:
NA - Not applicable.

Table 6-66 Green Bay Zone 3B Bird Tissue Concentrations

Analyte	Species	Tissue	Number of Samples	Number of Detects	Detection Frequency (%)	Detected Minimum	Detected Maximum	Mean	Data Distribution	95% UCL	90th Percentile	RME
PCBs (µg/kg)												
Total PCBs	Double-crested Cormorant	whole	21	20	95	246	15,000	5,384	Other	28,675	13,400	15,000
Pesticides (µg/kg)												
Dieldrin	Double-crested Cormorant	whole	20	19	95	63.0	300	128	Other	239	269	239
o,p'-DDD	Double-crested Cormorant	whole	20	0								
o,p'-DDE	Double-crested Cormorant	whole	20	0								
o,p'-DDT	Double-crested Cormorant	whole	20	0								
p,p'-DDD	Double-crested Cormorant	whole	20	3	15	10.0	20.0	6.3	Other	7.6	10.0	7.6
p,p'-DDE	Double-crested Cormorant	whole	20	20	100	140	6,500	2,010	Lognormal	4,546	5,850	4,546
p,p'-DDT	Double-crested Cormorant	whole	20	11	55	10.0	30.0	10.9	Other	14.8	20.0	14.8

Table 6-67 Estimated Exposure Concentrations for Piscivorous Birds in Green Bay Zone 3B

Analyte	Surface Water (µg/L) [1]		TL3 Fish: Alewife (µg/kg)		Common Tern Ingestion				Total Common Tern Ingestion (µg/day)		Total Common Tern Ingestion (µg/kg-BW/day)	
					Surface Water (µg/day) [1]		TL3 Fish: (µg/day)					
	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME
Mercury	0.05	0.07	25.0	50.0	0.001	0.001	1.5	2.9	1.5	2.9	12.3	24.5
Total PCBs	0.004	0.005	1,821	2,375	0.0001	0.0001	107	140	107	140	892	1,164
Dieldrin	NA	NA	19.1	27.3	—	—	1.1	1.6	1.1	1.6	9.3	13.4
p,p'-DDE	NA	NA	80.0	80.0	—	—	4.7	4.7	4.7	4.7	39.2	39.2

Analyte	Surface Water (µg/L) [1]		TL3 Fish: Alewife (µg/kg)		Forster's Tern Ingestion				Total Forster's Tern Ingestion (µg/day)		Total Forster's Tern Ingestion (µg/kg-BW/day)	
					Surface Water (µg/day) [1]		TL3 Fish: (µg/day)					
	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME
Mercury	0.05	0.07	25.0	50.0	0.001	0.001	1.8	3.6	1.8	3.6	11.3	22.6
Total PCBs	0.004	0.005	1,821	2,375	0.0001	0.0001	130	170	130	170	823	1,073
Dieldrin	NA	NA	19.1	27.3	—	—	1.4	2.0	1.4	2.0	8.6	12.3
p,p'-DDE	NA	NA	80.0	80.0	—	—	5.7	5.7	5.7	5.7	36.2	36.2

Analyte	Surface Water (µg/L) [1]		TL3 Fish: Alewife (µg/kg)		Double-crested Cormorant Ingestion				Double-crested Cormorant Ingestion (µg/day)		Total Double-crested Cormorant Ingestion (µg/kg-BW/day)	
					Surface Water (µg/day) [1]		TL3 Fish: (µg/day)					
	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME
Mercury	0.05	0.07	25.0	50.0	0.004	0.01	8.0	15.9	8.0	15.9	4.7	9.5
Total PCBs	0.004	0.005	1,821	2,375	0.0003	0.0004	579	755	579	755	345	450
Dieldrin	NA	NA	19.1	27.3	—	—	6.1	8.7	6.1	8.7	3.6	5.2
p,p'-DDE	NA	NA	80.0	80.0	—	—	25.4	25.4	25.4	25.4	15.1	15.1

Notes:

[1] Mercury in unfiltered water, total PCBs on a filtered + particulate basis.

NA indicates that data was not collected for this reach and this contaminant.

Shaded areas indicate that the concentration is estimated as half of the detection limit.

"—" indicates that the concentration could not be calculated.

Table 6-67 Estimated Exposure Concentrations for Piscivorous Birds in Green Bay Zone 3B (Continued)

Analyte	Surface Water (µg/L) [1]		TL3 Fish: Carp (µg/kg)		TL4 Fish: Walleye (µg/kg)		Bald Eagle Ingestion						Total Bald Eagle Ingestion (µg/day)		Total Bald Eagle Ingestion (µg/kg-BW/day)	
							Surface Water (µg/day) [1]		TL3 Fish (µg/day)		TL4 Fish (µg/day)					
	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME
Mercury	0.05	0.07	110	170	250	650	0.01	0.01	46.4	71.7	26.3	68.3	72.7	140	15.6	30.1
Total PCBs	0.004	0.005	4,947	6,149	6,429	11,741	0.001	0.001	2,088	2,595	675	1,233	2,763	3,828	594	823
Dieldrin	NA	NA	43.2	54.6	50.1	63.3	—	—	18.2	23.1	5.3	6.6	23.5	29.7	5.1	6.4
p,p'-DDE	NA	NA	126	240	207	540	—	—	53.2	101.3	21.7	56.7	74.9	158	16.1	34.0

Notes:

[1] Mercury in unfiltered water, total PCBs on a filtered + particulate basis.

NA indicates that data was not collected for this reach and this contaminant.

Shaded areas indicate that the concentration is estimated as half of the detection limit.

"—" indicates that the concentration could not be calculated.

Model Assumptions:

Common Tern

Food Ingestion =	0.0588	kg/day
Water Ingestion =	0.014	L/day
Body Weight =	0.12	kg

Forster's Tern

Food Ingestion =	0.0714	kg/day
Water Ingestion =	0.017	L/day
Body Weight =	0.158	kg

Double-crested Cormorant

Food Ingestion =	0.318	kg/day
Water Ingestion =	0.084	L/day
Body Weight =	1.68	kg

Bald Eagle

Food Ing. (TL3 Fish) =	0.422	kg/day
Food Ing. (TL4 Fish) =	0.105	kg/day
Water Ingestion =	0.165	L/day
Body Weight =	4.65	kg

Table 6-68 PCB Congeners in Whole Double-crested Cormorants in Green Bay Zone 3B

Analyte	Tissue	Number of Samples	Number of Detects	Detection Frequency (%)	Detected Minimum	Detected Maximum	Mean	Data Distribution	95% UCL	90th Percentile	RME
PCBs (µg/kg)											
PCB Congener 77	whole	16	6	38	0.2	0.8	0.2	Other	0.4	0.7	0.4
PCB Congener 105	whole	16	16	100	2.0	230	92.2	Normal	122	209	122
PCB Congener 118/106	whole	16	16	100	4.7	650	215	Other	671	594	650
PCB Congener 126	whole	16	13	81	0.3	1.7	0.6	Normal	0.8	1.4	0.8
PCB Congener 169	whole	16	5	31	0.1	0.2	0.1	Other	0.1	0.1	0.1

Table 6-69 Estimated Exposure Concentrations for Mink in Green Bay Zone 3B

Analyte	Surface Water (µg/L) [1]		Surface Sediment (µg/kg) [2]		Whole Carp (µg/kg)		Mink Ingestion						Total Mink Ingestion (µg/day)		Total Mink Ingestion (µg/kg-BW/day)	
							Surface Water (µg/day) [1]		Surface Sediment (µg/day)		Whole Carp (µg/day)					
	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME
Mercury	0.05	0.07	100	200	110	170	0.004	0.01	0.4	0.8	16.8	26.0	17.2	26.8	21.5	33.5
Total PCBs (N)	0.004	0.005	542	809	4,947	6,149	0.0003	0.0004	2.2	3.2	757	941	759	944	949	1,180
Total PCBs (I_d)	0.004	0.005	482	483	4,947	6,149	0.0003	0.0004	1.9	1.9	757	941	759	943	949	1,178
Dieldrin	NA	NA	5.1	8.1	43.2	54.6	—	—	0.02	0.03	6.6	8.4	6.6	8.4	8.3	10.5
p,p'-DDE	NA	NA	5.1	8.1	126	240	—	—	0.02	0.03	19.3	36.7	19.3	36.8	24.1	45.9

Notes:

[1] Mercury in unfiltered water, total PCBs on a filtered + particulate basis.

[2] p,p'-DDT rather than p,p'-DDE was used because this was the predominant form in the sediment.

NA indicates that data was not collected for this reach and this contaminant.

Shaded areas indicate that the concentration is estimated as half of the detection limit.

"—" indicates that the concentration could not be calculated.

Model Assumptions:

Food Ingestion =	0.153	kg/day
Water Ingestion =	0.081	L/day
Sediment Ingestion =	0.004	kg/day
Body Weight =	0.8	kg

Table 6-70 Surface Water Concentrations in Green Bay Zone 4

Analyte	Number of Samples	Number of Detects	Detection Frequency (%)	Detected Minimum	Detected Maximum	Mean	Data Distribution	95% UCL	90th Percentile	RME	Criteria NOAEC	Criteria LOAEC	Reference	Hazard Quotients Mean NOEL	Hazard Quotients Mean LOEL	Hazard Quotients RME NOEL	Hazard Quotients RME LOEL
Metals (ng/L)																	
Mercury (filtered)	20	0															
Mercury (unfiltered)	20	0															
PCBs (ng/L)																	
Total PCBs (filtered)	66	66	100	0.3	1.3	0.6	Lognormal	0.6	0.8	0.6	50	500	[1]	< 0.1	< 0.1	< 0.1	< 0.1
Total PCBs (particulate)	86	66	77	0.1	2.4	0.9	Other	1.1	2.5	1.1	50	500	[1]	< 0.1	< 0.1	< 0.1	< 0.1
Total PCBs (filtered + particulate)				0.4	3.7	1.5		1.8	3.3	1.8	50	500	[1]	< 0.1	< 0.1	< 0.1	< 0.1

Note:
[1] Niimi, 1996.

Table 6-71 Surface Sediment Concentrations in Green Bay Zone 4

Analyte	Number of Samples	Number of Detects	Detection Frequency (%)	Detected Minimum	Detected Maximum	Mean	Data Distribution	95% UCL	90th Percentile	RME	Criteria	Reference	Hazard Quotients Mean	Hazard Quotients RME
Metals (mg/kg)														
Arsenic	4	4	100	1.4	8.9	5.0	Normal	8.6	0.0	8.6	12.1	[1]	0.4	0.7
Lead	4	4	100	2.1	4.5	3.1	Normal	4.5	0.0	4.5	34.2	[1]	0.1	0.1
Mercury	4	1	25	0.11	0.11	0.05	Other	0.62	0.00	0.11	0.17	[2]	0.3	0.6
PCBs (µg/kg)														
Total PCBs (N)	31	27	87	10.0	264	82.9	Lognormal	117	185	117	31.6	[1]	2.6	3.7
Total PCBs (I_d)	197,067	197,067	100	2.5	214	45.7	Other	45.8	81.2	45.8	31.6	[1]	1.4	1.4
Pesticides (µg/kg)														
Dieldrin	4	0												
p,p'-DDD	4	0												
p,p'-DDE	4	0												
p,p'-DDT	4	0												

Notes:

N indicates that the data was not interpolated based on depth.

I_0 indicates that interpolated grid areas for which "no values" existed were assumed to equal zero.

I_d indicates that interpolated grid areas for which "no values" existed were deleted from the database.

[1] ARCS SEC (EPA, 1996a).

[2] Environment Canada TEL (Smith *et al.*, 1996); p,p'-DDT based on total DDT TEL.

Table 6-72 PCB Congener Concentrations in Surface Sediment in Green Bay Zone 4

Analyte	Number of Samples	Number of Detects	Detection Frequency (%)	Detected Minimum	Detected Maximum	Mean	Data Distribution	95% UCL	90th Percentile	RME
PCBs (µg/kg)										
PCB Congener 77	4	2	50	0.01	0.04	NE	Normal	0.1	0.0	0.04
PCB Congener 81	31	27	87	0.04	0.6	0.2	Lognormal	0.4	0.5	0.4
PCB Congener 105	4	2	50	0.02	0.1	0.05	Normal	0.1	0.0	0.1
PCB Congener 118	31	28	90	0.1	9.1	2.8	Other	11.6	7.1	9.1
PCB Congener 126	4	0								
PCB Congener 169	4	0								

Note:

NE indicates that the calculated mean exceeded the detected maximum and, therefore, the mean was not evaluated.

Table 6-73 Green Bay Zone 4 Whole Fish Concentrations

Analyte	Species	Number of Samples	Number of Detects	Detection Frequency (%)	Detected Minimum	Detected Maximum	Mean	Data Distribution	95% UCL	90th Percentile	RME
Metals (mg/kg)											
Mercury	Yellow Perch	5	5	100	0.02	0.04	0.03	Normal	0.03	0.0	0.03
	Carp	10	10	100	0.1	0.2	0.17	Normal	0.2	0.2	0.2
	Walleye	20	20	100	0.12	0.34	0.21	Normal	0.23	0.3	0.23
PCBs (µg/kg)											
Total PCBs	Alewife	8	8	100	110	2,000	1,036	Normal	1,488	0.0	1,488
	Rainbow Smelt	18	18	100	150	1,600	526	Lognormal	764	1,150	764
	Yellow Perch	5	5	100	71.0	85.0	79.8	Normal	84.8	0.0	84.8
	Carp	20	20	100	394	9,265	2,992	Lognormal	4,573	8,621	4,573
	Walleye	36	36	100	620	9,620	2,546	Lognormal	3,294	5,867	3,294
	Brown Trout	18	18	100	1,456	3,900	2,451	Normal	2,714	3,720	2,714
Pesticides (µg/kg)											
Dieldrin	Alewife	8	8	100	4.1	29.0	20.8	Normal	26.1	0.0	26.1
	Rainbow Smelt	18	18	100	7.3	39.0	18.1	Normal	21.9	30.9	21.9
	Carp	20	20	100	10.0	78.0	27.7	Lognormal	36.0	50.2	36.0
	Walleye	33	33	100	11.0	140	46.9	Lognormal	62.0	92.4	62.0
	Brown Trout	13	13	100	68.0	120	88.2	Normal	95.7	112	95.7
p,p'-DDD	Yellow Perch	5	0								
	Carp	10	10	100	24.0	149	75.8	Normal	100	147	100
	Walleye	20	20	100	15.0	46.0	28.7	Normal	32.2	43.4	32.2
p,p-DDE	Yellow Perch	5	5	100	14.0	16.0	14.8	Normal	15.6	0.0	15.6
	Carp	10	10	100	161	1,749	885	Normal	1,160	1,715	1,160
	Walleye	20	20	100	235	1,168	479	Lognormal	593	995	593
p,p-DDT	Yellow Perch	5	0								
	Carp	10	9	90	5.0	15.0	8.7	Normal	10.9	14.7	10.9
	Walleye	20	20	100	14.0	61.0	33.9	Lognormal	42.6	59.6	42.6

Table 6-74 PCB Congener Concentrations in Green Bay Zone 4 Whole Fish

Analyte	Species	Number of Samples	Number of Detects	Detection Frequency (%)	Detected Minimum	Detected Maximum	Mean	Data Distribution	95% UCL	90th Percentile	RME
PCBs (µg/kg)											
PCB Congener 77/110	Alewife	8	8	100	4.2	63.0	40.2	Normal	54.6	0.0	54.6
PCB Congener 81	Alewife	8	8	100	0.8	5.8	3.7	Normal	4.9	0.0	4.9
PCB Congener 132/153/105	Alewife	8	8	100	5.2	78.0	51.7	Normal	69.5	0.0	69.5
PCB Congener 118	Alewife	8	8	100	3.5	48.0	29.9	Normal	40.6	0.0	40.6
PCB Congener 77/110	Rainbow Smelt	18	17	94	1.8	59.0	20.7	Normal	27.6	53.6	27.6
PCB Congener 81	Rainbow Smelt	18	18	100	1.0	5.7	2.8	Lognormal	3.8	5.4	3.8
PCB Congener 132/153/105	Rainbow Smelt	18	18	100	12.0	80.0	29.2	Lognormal	40.8	56.6	40.8
PCB Congener 118	Rainbow Smelt	18	18	100	5.7	45.0	17.9	Lognormal	25.7	41.4	25.7
PCB Congener 77/110	Carp	10	10	100	63.0	210	116	Normal	145	206	145
PCB Congener 81	Carp	10	10	100	6.8	20.0	11.4	Normal	14.0	19.6	14.0
PCB Congener 132/153/105	Carp	10	10	100	100	300	206	Normal	246	297	246
PCB Congener 118	Carp	10	10	100	55.0	190	116	Normal	143	188	143
PCB Congener 77	Walleye	1	1	100	2.1	2.1	NA			0.0	2.1
PCB Congener 81	Walleye	14	13	93	0.6	21	10.4	Normal	13.6	19.5	13.6
PCB Congener 105	Walleye	3	3	100	68.8	111	84.7	Normal	124	0.0	111
PCB Congener 118	Walleye	16	16	100	27.0	308	137	Normal	166	229	166
PCB Congener 126	Walleye	1	1	100	0.3	0.3	NA			0.0	0.3
PCB Congener 169	Walleye	2	2	100	3.2	7.8	5.5	Normal	20.0	0.0	7.8
PCB Congener 77	Brown Trout	1	1	100	1.6	1.6	NA			0.0	1.6
PCB Congener 81	Brown Trout	14	14	100	0.2	17.0	9.2	Normal	10.9	15.0	10.9
PCB Congener 105	Brown Trout	5	5	100	17.8	42.9	36.1	Other	60.6	0.0	42.9
PCB Congener 118	Brown Trout	18	18	100	61.0	130	92.1	Normal	101	123	101
PCB Congener 126	Brown Trout	1	1	100	0.3	0.3	NA			0.0	0.3

Note:
NA - Not applicable.

Table 6-75 Estimated Exposure Concentrations for Piscivorous Birds in Green Bay Zone 4

Analyte	Surface Water (µg/L) [1]		TL3 Fish: Alewife and Yellow Perch (µg/kg)		Common Tern Ingestion				Total Common Tern Ingestion (µg/day)		Total Common Tern Ingestion (µg/kg-BW/day)	
					Surface Water (µg/day) [1]		TL3 Fish (µg/day)					
	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME
Mercury	0.02	0.03	30.0	30.0	0.0003	0.0004	1.8	1.8	1.8	1.8	14.7	14.7
Total PCBs	0.0015	0.0018	1,036	1,488	0.00002	0.00003	60.9	87.5	60.9	87.5	508	729
Dieldrin	NA	NA	20.8	26.1	—	—	1.2	1.5	1.2	1.5	10.2	12.8
p,p'-DDE	NA	NA	14.8	15.6	—	—	0.9	0.9	0.9	0.9	7.3	7.6

Analyte	Surface Water (µg/L) [1]		TL3 Fish: Alewife and Yellow Perch (µg/kg)		Forster's Tern Ingestion				Total Forster's Tern Ingestion (µg/day)		Total Forster's Tern Ingestion (µg/kg-BW/day)	
					Surface Water (µg/day) [1]		TL3 Fish (µg/day)					
	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME
Mercury	0.02	0.03	30.0	30.0	0.0004	0.0005	2.1	2.1	2.1	2.1	13.6	13.6
Total PCBs	0.0015	0.0018	1,036	1,488	0.00003	0.00003	74.0	106	74.0	106	468	672
Dieldrin	NA	NA	20.8	26.1	—	—	1.5	1.9	1.5	1.9	9.4	11.8
p,p'-DDE	NA	NA	14.8	15.6	—	—	1.1	1.1	1.1	1.1	6.7	7.0

Analyte	Surface Water (µg/L) [1]		TL3 Fish: Alewife and Yellow Perch (µg/kg)		Double-crested Cormorant Ingestion				Double-crested Cormorant Ingestion (µg/day)		Total Double-crested Cormorant Ingestion (µg/kg-BW/day)	
					Surface Water (µg/day) [1]		TL3 Fish (µg/day)					
	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME
Mercury	0.02	0.03	30.0	30.0	0.002	0.002	9.5	9.5	9.5	9.5	5.7	5.7
Total PCBs	0.0015	0.0018	1,036	1,488	0.0001	0.0002	330	473	330	473	196	282
Dieldrin	NA	NA	20.8	26.1	—	—	6.6	8.3	6.6	8.3	3.9	4.9
p,p'-DDE	NA	NA	14.8	15.6	—	—	4.7	5.0	4.7	5.0	2.8	3.0

Notes:

[1] Mercury in unfiltered water, total PCBs on a filtered + particulate basis.

NA indicates that data was not collected for this reach and this contaminant.

Shaded areas indicate that the concentration is estimated as half of the detection limit.

"—" indicates that the concentration could not be calculated.

Table 6-75 Estimated Exposure Concentrations for Piscivorous Birds in Green Bay Zone 4 (Continued)

Analyte	Surface Water (µg/L) [1]		TL3 Fish: Carp (µg/kg)		TL4 Fish: Walleye (µg/kg)		Bald Eagle Ingestion						Total Bald Eagle Ingestion (µg/day)		Total Bald Eagle Ingestion (µg/kg-BW/day)	
							Surface Water (µg/day) [1]		TL3 Fish (µg/day)		TL4 Fish (µg/day)					
	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME
Mercury	0.02	0.03	170	200	210	230	0.004	0.005	71.7	84.4	22.1	24.2	93.8	108.6	20.2	23.3
Total PCBs	0.002	0.002	2,992	4,573	2,546	3,294	0.0002	0.0003	1,263	1,930	267	346	1,530	2,275	329	489
Dieldrin	NA	NA	27.7	36.0	46.9	62.0	—	—	11.7	15.2	4.9	6.5	16.6	21.7	3.6	4.7
p,p'-DDE	NA	NA	885	1160	479	593	—	—	374	489	50.3	62.3	424	552	91.2	119

Notes:

[1] Mercury in unfiltered water, total PCBs on a filtered + particulate basis.

NA indicates that data was not collected for this reach and this contaminant.

Shaded areas indicate that the concentration is estimated as half of the detection limit.

"—" indicates that the concentration could not be calculated.

Model Assumptions:

Common Tern

Food Ingestion =	0.0588	kg/day
Water Ingestion =	0.014	L/day
Body Weight =	0.12	kg

Forster's Tern

Food Ingestion =	0.0714	kg/day
Water Ingestion =	0.017	L/day
Body Weight =	0.158	kg

Double-crested Cormorant

Food Ingestion =	0.318	kg/day
Water Ingestion =	0.084	L/day
Body Weight =	1.68	kg

Bald Eagle

Food Ing. (TL3 Fish) =	0.422	kg/day
Food Ing. (TL4 Fish) =	0.105	kg/day
Water Ingestion =	0.165	L/day
Body Weight =	4.65	kg

Table 6-76 Estimated Exposure Concentrations for Mink in Green Bay Zone 4

Analyte	Surface Water (µg/L) [1]		Surface Sediment (µg/kg) [2]		Whole Carp (µg/kg)		Mink Ingestion						Total Mink Ingestion (µg/day)		Total Mink Ingestion (µg/kg-BW/day)	
							Surface Water (µg/day) [1]		Surface Sediment (µg/day)		Whole Carp (µg/day)					
	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME
Mercury	0.02	0.03	50.0	100	170	200	0.002	0.002	0.2	0.4	26.0	30.6	26.2	31.0	32.8	38.8
Total PCBs (N)	0.0015	0.0018	82.9	117	2,992	4,573	0.0001	0.0001	0.3	0.5	458	700	458	700	573	875
Total PCBs (I_d)	0.0015	0.0018	45.7	45.8	2,992	4,573	0.0001	0.0001	0.2	0.2	458	700	458	700	573	875
Dieldrin	NA	NA	1.7	1.9	27.7	36.0	—	—	0.01	0.01	4.2	5.5	4.2	5.5	5.3	6.9
p,p'-DDE	NA	NA	1.7	1.9	885	1160	—	—	0.01	0.01	135	177	135	177	169	222

Notes:

[1] Mercury in unfiltered water, total PCBs on a filtered + particulate basis.

[2] p,p'-DDT rather than p,p'-DDE was used because this was the predominant form in the sediment.

NA indicates that data was not collected for this reach and this contaminant.

Shaded areas indicate that the concentration is estimated as half of the detection limit.

"—" indicates that the concentration could not be calculated.

Model Assumptions:

Food Ingestion =	0.153	kg/day
Water Ingestion =	0.081	L/day
Sediment Ingestion =	0.004	kg/day
Body Weight =	0.8	kg

Table 6-77 Hazard Quotients for Whole Fish in Little Lake Butte des Morts Reach

Analyte	Species	Number of Samples	Number of Detects	Detection Frequency (%)	Mean	RME	Criteria		Hazard Quotients			
							NOAEC	LOAEC	NOAEC		LOAEC	
									Mean	RME	Mean	RME
Metals (mg/kg)												
Arsenic	Carp	2	2	100	0.1	0.2	0.50	5.00	0.3	0.3	< 0.1	< 0.1
Mercury	Yellow Perch	2	0									
	Carp	5	3	60	0.05	0.1	0.25	2.37	0.2	0.2	< 0.1	< 0.1
	Walleye	4	1	25	NE	0.03	0.25	2.37	NE	0.1	NE	< 0.1
PCBs (µg/kg)												
Total PCBs	Gizzard Shad	4	4	100	296	530	760	7,600	0.4	0.7	< 0.1	0.1
	Golden Shiner	2	2	100	993	1,140	760	7,600	1.3	1.5	0.1	0.2
	Yellow Perch	1	1	100	NA	363	760	7,600	NA	0.5	NA	< 0.1
	Carp	30	30	100	1,992	2,957	760	7,600	2.6	3.9	0.3	0.4
	Walleye	13	11	85	1,159	3,800	760	7,600	1.5	5.0	0.2	0.5
Pesticides (µg/kg)												
Dieldrin	Yellow Perch	2	0									
	Carp	6	2	33	NE	1.0	370	3,700	NE	< 0.1	NE	< 0.1
	Walleye	7	0									
o,p'-DDD	Carp	5	0									
	Walleye	4	0									
o,p'-DDE	Carp	4	1	25	NE	5.8	300	2,950	NE	<0.1	NE	< 0.1
	Walleye	4	1	25	12.5	16.0	300	2,950	<0.1	0.1	< 0.1	< 0.1
o,p'-DDT	Carp	5	0									
	Walleye	4	0									
p,p'-DDD	Yellow Perch	2	0									
	Carp	7	3	43	NE	5.2	300	2,950	NE	<0.1	NE	< 0.1
	Walleye	7	1	14	23.5	44.9	300	2,950	0.1	0.1	< 0.1	< 0.1
p,p'-DDE	Yellow Perch	2	2	100	9.5	11.0	300	2,950	< 0.1	< 0.1	< 0.1	< 0.1
	Carp	7	5	71	16.9	23.8	300	2,950	0.1	0.1	< 0.1	< 0.1
	Walleye	7	5	71	47.6	61.7	300	2,950	0.2	0.2	< 0.1	<0.01
p,p'-DDT	Yellow Perch	2	0									
	Carp	7	0									
	Walleye	7	0									

Notes:

NA - Not applicable.

NE indicates that the calculated mean exceeded the detected maximum and, therefore, the mean was not evaluated.

Table 6-78 PCB Congener and Dioxin/Furan Hazard Quotients for Whole Fish in Little Lake Butte des Morts Reach

Analyte	Species	Number of Samples	Number of Detects	Detection Frequency (%)	Mean	RME	TEF	Mean TEC	RME TEC	Hazard Quotients			
										NOAEC		LOAEC	
										Mean	RME	Mean	RME
PCBs and Dioxins/Furans (µg/kg)													
PCB Congener 77	Golden Shiner	2	2	100	2.21	2.24	0.0001	2.2E-04	2.2E-04	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 81/87/115	Golden Shiner	2	2	100	11.40	14.00	0.0005	5.7E-03	7.0E-03	0.1	0.2	0.1	0.1
PCB Congener 105	Golden Shiner	2	2	100	4.68	4.72	0.000005	2.3E-05	2.4E-05	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 118	Golden Shiner	2	2	100	21.20	25.00	0.000005	1.1E-04	1.3E-04	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 126	Golden Shiner	2	1	50	3.01E-02	3.93E-02	0.005	1.5E-04	2.0E-04	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 169	Golden Shiner	2	0										
Total TEQ	**Golden Shiner**							**6.2E-03**	**7.6E-03**	**0.2**	**0.2**	**0.1**	**0.1**
PCB Congener 77	Yellow Perch	1	1	100	NA	0.13	0.0001	NA	1.3E-05	NA	< 0.1	NA	< 0.1
PCB Congener 81	Yellow Perch	1	0										
PCB Congener 105	Yellow Perch	1	1	100	NA	2.00	0.000005	NA	1.0E-05	NA	< 0.1	NA	< 0.1
PCB Congener 118	Yellow Perch	1	1	100	NA	8.60	0.000005	NA	4.3E-05	NA	< 0.1	NA	< 0.1
PCB Congener 126	Yellow Perch	1	0										
PCB Congener 169	Yellow Perch	1	0										
Total TEQ	**Yellow Perch**							**0.0E+00**	**6.6E-05**	**NA**	**< 0.1**	**NA**	**< 0.1**
2,3,7,8-TCDD	Carp	3	3	100	2.5E-04	2.9E-04	1	2.5E-04	2.9E-04	< 0.1	< 0.1	< 0.1	< 0.1
2,3,7,8-TCDF	Carp	4	4	100	2.2E-03	2.5E-03	0.05	1.1E-04	1.3E-04	< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,4,6,7,8,9-OCDD	Carp	3	3	100	1.0E-02	1.5E-02	0.0001	1.0E-06	1.5E-06	< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,4,6,7,8,9-OCDF	Carp	3	0										
1,2,3,4,6,7,8-HPCDD	Carp	3	3	100	6.3E-03	7.3E-03	0.001	6.3E-06	7.3E-06	< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,4,6,7,8-HPCDF	Carp	3	3	100	9.9E-04	1.1E-03	0.01	9.9E-06	1.1E-05	< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,4,7,8,9-HPCDF	Carp	3	0										
1,2,3,4,7,8-HXCDD	Carp	3	3	100	7.8E-04	1.0E-03	0.5	3.9E-04	5.0E-04	< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,4,7,8-HXCDF	Carp	3	2	67	2.6E-04	3.4E-04	0.1	2.6E-05	3.4E-05	< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,6,7,8-HXCDD	Carp	3	3	100	2.2E-03	2.6E-03	0.01	2.2E-05	2.6E-05	< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,6,7,8-HXCDF	Carp	3	3	100	3.6E-04	4.0E-04	0.1	3.6E-05	4.0E-05	< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,7,8,9-HXCDD	Carp	3	2	67	3.5E-04	3.9E-04	0.01	3.5E-06	3.9E-06	< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,7,8,9-HXCDF	Carp	3	0										
1,2,3,7,8-PECDD	Carp	3	3	100	6.4E-04	9.7E-04	1	6.4E-04	9.7E-04	< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,7,8-PECDF	Carp	3	3	100	5.4E-04	5.5E-04	0.05	2.7E-05	2.8E-05	< 0.1	< 0.1	< 0.1	< 0.1
2,3,4,6,7,8-HXCDF	Carp	3	3	100	2.1E-04	2.4E-04	0.1	2.1E-05	2.4E-05	< 0.1	< 0.1	< 0.1	< 0.1
2,3,4,7,8-PECDF	Carp	3	2	67	5.1E-04	6.6E-04	0.5	2.6E-04	3.3E-04	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 77	Carp	7	7	100	1.9E+00	8.5E+00	0.0001	1.9E-04	8.5E-04	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 81	Carp	2	0										
PCB Congener 105	Carp	7	6	86	8.6E+00	3.5E+01	0.000005	4.3E-05	1.8E-04	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 118	Carp	7	7	100	3.5E+01	1.5E+02	0.000005	1.8E-04	7.5E-04	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 126	Carp	7	1	14	NE	2.7E-02	0.005	NE	1.4E-04	NE	< 0.1	NE	< 0.1
PCB Congener 169	Carp	6	0										
Total TEQ	**Carp**							**2.2E-03**	**4.3E-03**	**0.1**	**0.1**	**< 0.1**	**0.1**

Table 6-78 PCB Congener and Dioxin/Furan Hazard Quotients for Whole Fish in Little Lake Butte des Morts Reach (Continued)

Analyte	Species	Number of Samples	Number of Detects	Detection Frequency (%)	Mean	RME	TEF	Mean TEC	RME TEC	Hazard Quotients			
										NOAEC		LOAEC	
										Mean	RME	Mean	RME
2,3,7,8-TCDD	Walleye	2	2	100	4.5E-04	5.4E-04	1	4.5E-04	5.4E-04	< 0.1	< 0.1	< 0.1	< 0.1
2,3,7,8-TCDF	Walleye	3	3	100	5.4E-03	6.2E-03	0.05	2.7E-04	3.1E-04	< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,4,6,7,8,9-OCDD	Walleye	3	2	67	9.3E-04	1.2E-03	0.0001	9.3E-08	1.2E-07	< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,4,6,7,8,9-OCDF	Walleye	3	0										
1,2,3,4,6,7,8-HPCDD	Walleye	3	2	67	9.3E-04	1.2E-03	0.001	9.3E-07	1.2E-06	< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,4,6,7,8-HPCDF	Walleye	3	2	67	3.5E-04	4.5E-04	0.01	3.5E-06	4.5E-06	< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,4,7,8,9-HPCDF	Walleye	3	0										
1,2,3,4,7,8-HXCDD	Walleye	3	2	67	2.2E-04	3.1E-04	0.5	1.1E-04	1.6E-04	< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,4,7,8-HXCDF	Walleye	3	0										
1,2,3,6,7,8-HXCDD	Walleye	3	3	100	1.3E-03	1.3E-03	0.01	1.3E-05	1.3E-05	< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,6,7,8-HXCDF	Walleye	3	3	100	2.4E-04	2.7E-04	0.1	2.4E-05	2.7E-05	< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,7,8,9-HXCDD	Walleye	3	3	100	2.5E-04	2.9E-04	0.01	2.5E-06	2.9E-06	< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,7,8,9-HXCDF	Walleye	3	0										
1,2,3,7,8-PECDD	Walleye	3	3	100	7.3E-04	8.6E-04	1	7.3E-04	8.6E-04	< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,7,8-PECDF	Walleye	3	3	100	8.4E-04	1.0E-03	0.05	4.2E-05	5.0E-05	< 0.1	< 0.1	< 0.1	< 0.1
2,3,4,6,7,8-HXCDF	Walleye	3	3	100	3.8E-04	4.6E-04	0.1	3.8E-05	4.6E-05	< 0.1	< 0.1	< 0.1	< 0.1
2,3,4,7,8-PECDF	Walleye	3	3	100	6.4E-04	7.4E-04	0.5	3.2E-04	3.7E-04	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 77	Walleye	7	6	86	2.14	3.72	0.0001	2.1E-04	3.7E-04	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 81	Walleye	3	0										
PCB Congener 105	Walleye	7	7	100	9.25	13.96	0.000005	4.6E-05	7.0E-05	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 118	Walleye	7	7	100	36.78	55.10	0.000005	1.8E-04	2.8E-04	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 126	Walleye	7	4	57	0.22	1.10	0.005	1.1E-03	5.5E-03	< 0.1	0.1	< 0.1	0.1
PCB Congener 169	Walleye	7	1	14	0.06	0.12	0.00005	2.8E-06	6.0E-06	< 0.1	< 0.1	< 0.1	< 0.1
Total TEQ	**Walleye**							**3.5E-03**	**8.6E-03**	**0.1**	**0.2**	**< 0.1**	**0.1**

Notes:

NA - Not applicable.

NE indicates that the calculated mean exceeded the detected maximum and, therefore, the mean was not evaluated

TEQ Criteria: NOAEC = 0.041 μg/kg; LOAEC = 0.084 μg/kg.

Table 6-79 Hazard Quotients for Bird Tissue in Little Lake Butte des Morts Reach

Analyte	Species	Tissue	Number of Samples	Number of Detects	Detection Frequency (%)	Mean	RME	Reproduction Criteria		Hazard Quotients NOAEC		Hazard Quotients LOAEC		Deformity Criteria		Hazard Quotients NOAEC		Hazard Quotients LOAEC	
								NOAEC	LOAEC	Mean	RME	Mean	RME	NOAEC	LOAEC	Mean	RME	Mean	RME
PCBs (µg/kg)																			
Total PCBs	Tree Swallow	egg	5	5	100	2,924	3,732	4,700	7,600	0.6	0.8	0.4	0.5	800	8,000	3.7	4.7	0.4	0.5
Total PCBs	Tree Swallow	whole body	24	24	100	2,135	5,254	4,700	7,600	0.5	1.1	0.3	0.7	800	8,000	2.7	6.6	0.3	0.7
Pesticides (µg/kg)																			
Dieldrin	Tree Swallow	whole body	18	0															
o,p'-DDD	Tree Swallow	whole body	18	0															
o,p'-DDE	Tree Swallow	whole body	18	0															
o,p'-DDT	Tree Swallow	whole body	18	0															
p,p'-DDD	Tree Swallow	whole body	18	0															
p,p'-DDE	Tree Swallow	whole body	18	18	100	155	239	3,000	5,100	0.1	0.1	< 0.1	< 0.1						
p,p'-DDT	Tree Swallow	whole body	18	0															

Table 6-80 Estimated Hazard Quotients for Piscivorous Birds in Little Lake Butte des Morts Reach

Analyte	Total Estimated Exposure (µg/kg-BW/day)							
	Common Tern		Forster's Tern		Double-crested Cormorant		Bald Eagle	
	Mean	RME	Mean	RME	Mean	RME	Mean	RME
Mercury	12.5	13.1	11.5	12.1	4.8	5.1	5.7	6.4
Total PCBs	145	260	134	240	56.0	100	207	354
Dieldrin	0.6	0.6	0.6	0.6	0.2	0.2	0.4	1.1
p,p'-DDE	4.7	5.4	4.3	5.0	1.8	2.1	2.6	3.6

Analyte	NOAEC HQs							
	Common Tern		Forster's Tern		Double-crested Cormorant		Bald Eagle	
	Mean	RME	Mean	RME	Mean	RME	Mean	RME
Mercury	1.6	1.6	1.4	1.5	0.6	0.6	0.7	0.8
Total PCBs	1.3	2.3	1.2	2.1	0.5	0.9	1.8	3.2
Dieldrin	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1
p,p'-DDE	0.3	0.3	0.2	0.3	0.1	0.1	0.1	0.2

Analyte	LOAEC HQs							
	Common Tern		Forster's Tern		Double-crested Cormorant		Bald Eagle	
	Mean	RME	Mean	RME	Mean	RME	Mean	RME
Mercury	0.2	0.2	0.1	0.2	0.1	0.1	0.1	0.1
Total PCBs	0.1	0.2	0.1	0.2	0.1	0.1	0.2	0.3
Dieldrin	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1
p,p'-DDE	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1

Table 6-81 PCB Congener Hazard Quotients for Tree Swallows in Little Lake Butte des Morts Reach

Analyte	Species	Number of Samples	Number of Detects	Detection Frequency (%)	Mean	RME	Tillitt *et al.*, 1991b			Van den Berg *et al.*, 1998			Reproduction Hazard Quotients (based on Tillitt *et al.*, 1991b TEFs)						Reproduction Hazard Quotients (based on Van den Berg *et al.*, 1998 TEFs)						Deformity Hazard Quotients (based on Tillitt *et al.*, 1991b TEFs)				Deformity Hazard Quotients (based on Van den Berg *et al.*, 1998 TEFs)			
							TEF	Mean TEC	RME TEC	TEF	Mean TEC	RME TEC	NOAEC		LD_{20}		LD_{30}		NOAEC		LD_{20}		LD_{30}		NOAEC		LOAEC		NOAEC		LOAEC	
													Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME
PCBs (µg/kg)																																
PCB Congener 77	egg	5	5	100	0.2	0.4	0.000018	4.0E-06	6.8E-06	0.05	1.1E-02	1.9E-02	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	1.6	2.7	0.1	0.1	< 0.1	0.1	< 0.1	< 0.1	< 0.1	< 0.1	0.3	0.5	< 0.1	< 0.1
PCB Congener 105	egg	5	5	100	20.7	32.8	0.0000076	1.6E-04	2.5E-04	0.0001	2.1E-03	3.3E-03	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	0.3	0.5	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	0.1	0.1	< 0.1	< 0.1
PCB Congener 118/106	egg	5	5	100	85.2	108	0.00000037	3.2E-05	4.0E-05	0.00001	8.5E-04	1.1E-03	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	0.1	0.2	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 126	egg	5	5	100	0.3	0.7	0.022	7.5E-03	1.5E-02	0.1	3.4E-02	7.0E-02	1.1	2.2	< 0.1	0.1	< 0.1	0.1	4.9	10	0.2	0.4	0.1	0.2	0.2	0.4	< 0.1	< 0.1	0.9	1.8	0.1	0.2
PCB Congener 169	egg	5	1	20	0.1	0.2	0.00047	3.8E-05	9.4E-05	0.001	8.0E-05	2.0E-04	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1
Total TEQ								**7.7E-03**	**1.6E-02**		**4.8E-02**	**9.3E-02**	**1.1**	**2.3**	**< 0.1**	**0.1**	**< 0.1**	**0.1**	**6.9**	**13**	**0.3**	**0.5**	**0.2**	**0.3**	**0.2**	**0.4**	**< 0.1**	**< 0.1**	**1.3**	**2.5**	**0.1**	**0.2**
PCBs (µg/kg)																																
PCB Congener 77	whole	15	0																													
PCB Congener 105	whole	15	15	100	16.7	37.1	0.0000076	1.3E-04	2.8E-04	0.0001	1.7E-03	3.7E-03	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	0.2	0.5	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	0.1	< 0.1	< 0.1
PCB Congener 118/106	whole	15	15	100	58.4	129	0.00000037	2.2E-05	4.8E-05	0.00001	5.8E-04	1.3E-03	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	0.1	0.2	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 126	whole	15	6	40	0.1	0.2	0.022	2.8E-03	4.6E-03	0.1	1.3E-02	2.1E-02	0.4	0.7	< 0.1	< 0.1	< 0.1	< 0.1	1.8	3.0	0.1	0.1	< 0.1	0.1	0.1	0.1	< 0.1	< 0.1	0.3	0.6	< 0.1	0.1
PCB Congener 169	whole	15	0																						< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1
Total TEQ								**2.9E-03**	**4.9E-03**		**1.5E-02**	**2.6E-02**	**0.4**	**0.7**	**< 0.1**	**< 0.1**	**< 0.1**	**< 0.1**	**2.1**	**3.7**	**0.1**	**0.1**	**< 0.1**	**0.1**	**0.1**	**0.1**	**< 0.1**	**< 0.1**	**0.4**	**0.7**	**< 0.1**	**0.1**

Table 6-82 Estimated Hazard Quotients for Mink in Little Lake Butte des Morts Reach

Analyte	Total Estimated Exposure (µg/kg-BW/day) Mean	RME
Mercury	14.8	19.2
Total PCBs (N)	435	680
Total PCBs (I_0)	397	582
Total PCBs (I_d)	400	584
Dieldrin	0.6	0.2
p,p'-DDE	3.2	4.8
Analyte	**NOAEC HQs Mean**	**RME**
Mercury	0.2	0.2
Total PCBs	109	170
Total PCBs (I_0)	99	146
Total PCBs (I_d)	100	146
Dieldrin	0.1	< 0.1
p,p'-DDE	< 0.1	< 0.1
Analyte	**LOAEC HQs Mean**	**RME**
Mercury	0.1	0.1
Total PCBs	3.3	5.2
Total PCBs (I_0)	3.1	4.5
Total PCBs (I_d)	3.1	4.5
Dieldrin	< 0.1	< 0.1
p,p'-DDE	< 0.1	< 0.1

Table 6-83 Hazard Quotients for Whole Fish in Appleton to Little Rapids Reach

Analyte	Species	Number of Samples	Number of Detects	Detection Frequency (%)	Mean	RME	Criteria		Hazard Quotients			
							NOAEC	LOAEC	NOAEC		LOAEC	
									Mean	RME	Mean	RME
Metals (mg/kg)												
Mercury	Yellow Perch	4	0									
	Carp	5	1	20	0.1	0.1	0.25	2.37	0.3	0.4	< 0.1	< 0.1
	Walleye	3	2	67	0.1	0.2	0.25	2.37	0.6	0.8	0.1	0.1
PCBs (µg/kg)												
Total PCBs	Yellow Perch	4	4	100	779	1,219	760	7,600	1.0	1.6	0.1	0.2
	Carp	12	12	100	2,581	3,606	760	7,600	3.4	4.7	0.3	0.5
	Walleye	4	4	100	2,737	3,900	760	7,600	3.6	5.1	0.4	0.5
Pesticides (µg/kg)												
Dieldrin	Yellow Perch	1	0									
	Carp	4	0									
	Walleye	3	0									
o,p'-DDD	Carp	2	0									
o,p'-DDT	Carp	2	0									
p,p'-DDD	Yellow Perch	1	0									
	Carp	6	0									
	Walleye	3	1	33	7.5	8.0	300	2,950	< 0.1	< 0.1	< 0.1	< 0.1
p,p'-DDE	Yellow Perch	1	1	100	NA	10.0	300	2,950	NA	< 0.1	NA	< 0.1
	Carp	6	4	67	47.8	75.6	300	2,950	0.2	0.3	< 0.1	< 0.1
	Walleye	3	3	100	57.0	65.0	300	2,950	0.2	0.2	< 0.1	< 0.1
p,p'-DDT	Yellow Perch	1	0									
	Carp	6	0									
	Walleye	3	0									

Note:
NA - Not applicable.

Table 6-84 PCB Congener and Dioxin/Furan Hazard Quotients for Whole Fish in Appleton to Little Rapids Reach

Analyte	Species	Number of Samples	Number of Detects	Detection Frequency (%)	Mean	RME	TEF	Mean TEC	RME TEC	Hazard Quotients			
										NOAEC		LOAEC	
										Mean	RME	Mean	RME
PCBs (µg/kg)													
PCB Congener 77	Yellow Perch	4	4	100	0.6	1.8	0.0001	6.14E-05	1.80E-04	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 81	Yellow Perch	4	1	25	0.1	0.4	0.0005	4.94E-05	1.90E-04	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 105	Yellow Perch	4	4	100	5.9	16.0	0.000005	2.95E-05	8.00E-05	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 118	Yellow Perch	4	4	100	23.3	42.9	0.000005	1.16E-04	2.15E-04	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 126	Yellow Perch	4	2	50	0.02	0.03	0.005	7.63E-05	1.62E-04	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 169	Yellow Perch	4	0										
Total TEQ	**Yellow Perch**							**3.33E-04**	**8.27E-04**	**< 0.1**	**< 0.1**	**< 0.1**	**< 0.1**
PCB Congener 77	Carp	5	4	80	0.7	1.5	0.0001	7.33E-05	1.47E-04	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 81	Carp	5	0										
PCB Congener 105	Carp	5	5	100	18.6	34.4	0.000005	9.28E-05	1.72E-04	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 118	Carp	5	5	100	56.3	98.0	0.000005	2.81E-04	4.90E-04	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 126	Carp	5	3	60	0.2	0.8	0.005	1.10E-03	4.15E-03	< 0.1	0.1	< 0.1	< 0.1
PCB Congener 169	Carp	5	2	40	0.04	0.1	0.00005	1.77E-06	4.50E-06	< 0.1	< 0.1	< 0.1	< 0.1
Total TEQ	**Carp**							**1.54E-03**	**4.96E-03**	**< 0.1**	**0.1**	**< 0.1**	**0.1**
PCB Congener 77	Walleye	3	3	100	3.0	4.5	0.0001	3.03E-04	4.50E-04	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 81	Walleye	3	0										
PCB Congener 105	Walleye	3	3	100	16.3	20.0	0.000005	8.17E-05	1.00E-04	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 118	Walleye	3	3	100	80.3	110.0	0.000005	4.02E-04	5.50E-04	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 126	Walleye	3	2	67	0.2	0.3	0.005	9.75E-04	1.65E-03	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 169	Walleye	3	0										
Total TEQ	**Walleye**							**1.76E-03**	**2.75E-03**	**< 0.1**	**0.1**	**< 0.1**	**< 0.1**

Note:

TEQ Criteria: NOAEC = 0.041 µg/kg; LOAEC = 0.084 µg/kg.

Table 6-85 Hazard Quotients for Bird Tissue in Appleton to Little Rapids Reach

Analyte	Species	Tissue	Number of Samples	Number of Detects	Detection Frequency (%)	Mean	RME	Reproduction Criteria		Hazard Quotients NOAEC		Hazard Quotients LOAEC		Deformity Criteria		Hazard Quotients NOAEC		Hazard Quotients LOAEC	
								NOAEC	LOAEC	Mean	RME	Mean	RME	NOAEC	LOAEC	Mean	RME	Mean	RME
Metals (mg/kg)																			
Mercury	Bald Eagle	liver	1	1	100	NA	1.40	0.2	2	NA	7.0	NA	0.7						
PCBs (µg/kg)																			
Total PCBs	Bald Eagle	egg	1	1	100	NA	36,000	4,700	7,600	NA	7.7	NA	4.7	800	8,000	NA	45	NA	4.5
Pesticides (µg/kg)																			
Dieldrin	Bald Eagle	egg	1	1	100	NA	70.0	100	1,000	NA	0.7	NA	0.1						
p,p'-DDD	Bald Eagle	egg	1	1	100	NA	160	3,000	5,100	NA	0.1	< 0.1	< 0.1						
p,p'-DDE	Bald Eagle	egg	1	1	100	NA	1,100	3,000	5,100	NA	0.4	NA	0.2						
p,p'-DDT	Bald Eagle	egg	1	0															

Table 6-86 Estimated Hazard Quotients for Piscivorous Birds in Appleton to Little Rapids Reach

Analyte	Total Estimated Exposure (µg/kg-BW/day)							
	Common Tern		Forster's Tern		Double-crested Cormorant		Bald Eagle	
	Mean	RME	Mean	RME	Mean	RME	Mean	RME
Mercury	12.3	12.3	11.3	11.3	4.7	4.7	8.6	14.5
Total PCBs	382	597	352	551	148	231	296	415
Dieldrin	0.6	1.2	0.6	1.1	0.2	0.5	0.2	0.3
p,p'-DDE	4.9	4.9	4.5	4.5	1.9	1.9	5.6	8.3

Analyte	NOAEC HQs							
	Common Tern		Forster's Tern		Double-crested Cormorant		Bald Eagle	
	Mean	RME	Mean	RME	Mean	RME	Mean	RME
Mercury	1.5	1.5	1.4	1.4	0.6	0.6	1.1	1.8
Total PCBs	3.4	5.3	3.1	4.9	1.3	2.1	2.6	3.7
Dieldrin	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1
p,p'-DDE	0.3	0.3	0.3	0.3	0.1	0.1	0.3	0.5

Analyte	LOAEC HQs							
	Common Tern		Forster's Tern		Double-crested Cormorant		Bald Eagle	
	Mean	RME	Mean	RME	Mean	RME	Mean	RME
Mercury	0.2	0.2	0.1	0.1	0.1	0.1	0.1	0.2
Total PCBs	0.3	0.5	0.3	0.5	0.1	0.2	0.3	0.4
Dieldrin	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1
p,p'-DDE	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1

Table 6-87 Estimated Hazard Quotients for Mink in Appleton to Little Rapids Reach

Analyte	Total Estimated Exposure (µg/kg-BW/day) Mean	RME
Mercury	15.5	29.5
Total PCBs (N)	527	766
Total PCBs (I_0)	494	691
Total PCBs (I_d)	501	697
Dieldrin	0.4	0.7
p,p'-DDE	9.1	14.4
Analyte	**NOAEC HQs Mean**	**RME**
Mercury	0.2	0.4
Total PCBs	132	192
Total PCBs (I_0)	124	173
Total PCBs (I_d)	125	174
Dieldrin	< 0.1	0.1
p,p'-DDE	< 0.1	< 0.1
Analyte	**LOAEC HQs Mean**	**RME**
Mercury	0.1	0.1
Total PCBs	4.1	5.9
Total PCBs (I_0)	3.8	5.3
Total PCBs (I_d)	3.9	5.4
Dieldrin	< 0.1	< 0.1
p,p'-DDE	< 0.1	< 0.1

Table 6-88 Hazard Quotients for Whole Fish in Little Rapids to De Pere Reach

Analyte	Species	Number of Samples	Number of Detects	Detection Frequency (%)	Mean	RME	Criteria		Hazard Quotients			
							NOAEC	LOAEC	NOAEC		LOAEC	
									Mean	RME	Mean	RME
Metals (mg/kg)												
Mercury	Yellow Perch	1	0									
	Carp	1	1	100	NA	0.2	0.025	2.37	NA	6.0	NA	0.1
	Walleye	1	1	100	NA	0.2	0.025	2.37	NA	6.4	NA	0.1
PCBs (µg/kg)												
Total PCBs	Gizzard Shad	3	3	100	347	370	760	7,600	0.5	0.5	< 0.1	< 0.1
	Golden Shiner	2	2	100	1,020	1,036	760	7,600	1.3	1.4	0.1	0.1
	Yellow Perch	1	1	100	NA	627	760	7,600	NA	0.8	NA	0.1
	Carp	20	20	100	3,919	5,800	760	7,600	5.2	7.6	0.5	0.8
	Walleye	4	4	100	3,179	4,587	760	7,600	4.2	6.0	0.4	0.6
Pesticides (µg/kg)												
Dieldrin	Yellow Perch	1	0									
	Carp	4	0									
	Walleye	4	1	25	3.4	5.4	370	3,700	< 0.1	< 0.1	< 0.1	< 0.1
o,p'-DDD	Carp	4	0									
	Walleye	3	0									
o,p'-DDE	Carp	4	0									
	Walleye	3	3	100	45.7	61.0	300	2,950	0.2	0.2	< 0.1	< 0.1
o,p'-DDT	Carp	4	0									
	Walleye	3	0									
p,p'-DDD	Yellow Perch	1	0									
	Carp	5	3	60	NE	8.0	300	2,950	NE	< 0.1	NE	< 0.1
	Walleye	4	0									
p,p'-DDE	Yellow Perch	1	1	100	NA	16.0	300	2,950	NA	0.1	NA	< 0.1
	Carp	5	5	100	74.2	128	300	2,950	0.2	0.4	< 0.1	< 0.1
	Walleye	4	4	100	129	208	300	2,950	0.4	0.7	< 0.1	0.1
p,p'-DDT	Yellow Perch	1	0									
	Carp	5	0									
	Walleye	4	0									

Notes:

NA - Not applicable.

NE indicates that the calculated mean exceeded the detected maximum and, therefore, the mean was not evaluated.

Table 6-89 PCB Congener and Dioxin/Furan Hazard Quotients for Whole Fish in Little Rapids to De Pere Reach

Analyte	Species	Number of Samples	Number of Detects	Detection Frequency (%)	Mean	RME	TEF	Mean TEC	RME TEC	Hazard Quotients			
										NOAEC		LOAEC	
										Mean	RME	Mean	RME
PCBs and Dioxins/Furans (µg/kg)													
PCB Congener 77	Golden Shiner	2	2	100	1.6	1.8	0.0001	1.6E-04	1.8E-04	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 81/87/115	Golden Shiner	2	2	100	14.0	15.0	0.0005	7.0E-03	7.5E-03	0.17	0.18	0.08	0.09
PCB Congener 105	Golden Shiner	2	2	100	5.0	5.3	0.000005	2.5E-05	2.7E-05	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 118	Golden Shiner	2	2	100	19.6	19.9	0.000005	9.8E-05	9.9E-05	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 126	Golden Shiner	2	2	100	0.04	0.04	0.005	2.0E-04	2.0E-04	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 169	Golden Shiner	2	0										
Total TEQ	**Golden Shiner**							**7.5E-03**	**8.0E-03**	**0.2**	**0.2**	**0.1**	**0.1**
PCB Congener 77	Yellow Perch	1	1	100	NA	0.01	0.0001	NA	1.0E-06	NA	< 0.1	NA	< 0.1
PCB Congener 81	Yellow Perch	1	0										
PCB Congener 105	Yellow Perch	1	1	100	NA	3.3	0.000005	NA	1.7E-05	NA	< 0.1	NA	< 0.1
PCB Congener 118	Yellow Perch	1	1	100	NA	13.0	0.000005	NA	6.5E-05	NA	< 0.1	NA	< 0.1
PCB Congener 126	Yellow Perch	1	0										
PCB Congener 169	Yellow Perch	1	0										
Total TEQ	**Yellow Perch**							**0.0E+00**	**8.3E-05**	**NA**	**< 0.1**	**NA**	**< 0.1**
2,3,7,8-TCDD	Carp	3	3	100	5.5E-04	8.8E-04	1	5.5E-04	8.8E-04	< 0.1	< 0.1	< 0.1	< 0.1
2,3,7,8-TCDF	Carp	3	3	100	1.1E-03	1.7E-03	0.05	5.7E-05	8.5E-05	< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,4,6,7,8,9-OCDD	Carp	3	3	100	3.3E-02	5.5E-02	0.0001	3.3E-06	5.5E-06	< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,4,6,7,8,9-OCDF	Carp	3	1	33	1.7E-03	2.3E-03	0.0001	1.7E-07	2.3E-07	< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,4,6,7,8-HPCDD	Carp	3	3	100	8.1E-03	1.1E-02	0.001	8.1E-06	1.1E-05	< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,4,6,7,8-HPCDF	Carp	3	3	100	1.4E-03	2.0E-03	0.01	1.4E-05	2.0E-05	< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,4,7,8,9-HPCDF	Carp	3	1	33	3.1E-04	6.9E-04	0.01	3.1E-06	6.9E-06	< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,4,7,8-HXCDD	Carp	3	3	100	1.3E-03	2.1E-03	0.5	6.3E-04	1.1E-03	< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,4,7,8-HXCDF	Carp	3	3	100	5.8E-04	8.5E-04	0.1	5.8E-05	8.5E-05	< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,6,7,8-HXCDD	Carp	3	3	100	2.9E-03	5.3E-03	0.01	2.9E-05	5.3E-05	< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,6,7,8-HXCDF	Carp	3	3	100	9.0E-04	1.6E-03	0.1	9.0E-05	1.6E-04	< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,7,8,9-HXCDD	Carp	3	3	100	4.2E-04	5.5E-04	0.01	4.2E-06	5.5E-06	< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,7,8,9-HXCDF	Carp	3	1	33	2.8E-04	6.4E-04	0.1	2.8E-05	6.4E-05	< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,7,8-PECDD	Carp	3	3	100	7.2E-04	1.1E-03	1	7.2E-04	1.1E-03	< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,7,8-PECDF	Carp	3	3	100	7.2E-04	1.1E-03	0.05	3.6E-05	5.5E-05	< 0.1	< 0.1	< 0.1	< 0.1
2,3,4,6,7,8-HXCDF	Carp	3	3	100	5.7E-04	9.3E-04	0.1	5.7E-05	9.3E-05	< 0.1	< 0.1	< 0.1	< 0.1
2,3,4,7,8-PECDF	Carp	3	3	100	1.1E-03	1.9E-03	0.5	5.5E-04	9.5E-04	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 77	Carp	4	4	100	7.4E-01	1.3E+00	0.0001	7.4E-05	1.3E-04	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 81	Carp	1	0										
PCB Congener 105	Carp	4	4	100	1.8E+01	4.3E+01	0.000005	9.1E-05	2.2E-04	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 118	Carp	4	4	100	7.2E+01	1.7E+02	0.000005	3.6E-04	8.6E-04	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 126	Carp	4	2	50	4.8E-02	1.0E-01	0.005	2.4E-04	5.2E-04	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 169	Carp	4	0										
Total TEQ	**Carp**							**3.6E-03**	**6.4E-03**	**0.1**	**0.2**	**< 0.1**	**0.1**

Table 6-89 PCB Congener and Dioxin/Furan Hazard Quotients for Whole Fish in Little Rapids to De Pere Reach (Continued)

Analyte	Species	Number of Samples	Number of Detects	Detection Frequency (%)	Mean	RME	TEF	Mean TEC	RME TEC	Hazard Quotients			
										NOAEC		LOAEC	
										Mean	RME	Mean	RME
2,3,7,8-TCDD	Walleye	3	3	100	8.0E-04	9.9E-04	1	8.0E-04	9.9E-04	< 0.1	< 0.1	< 0.1	< 0.1
2,3,7,8-TCDF	Walleye	3	3	100	1.0E-02	1.3E-02	0.05	5.1E-04	6.6E-04	< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,4,6,7,8,9-OCDD	Walleye	3	3	100	2.2E-03	2.6E-03	0.0001	2.2E-07	2.6E-07	< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,4,6,7,8,9-OCDF	Walleye	3	0										
1,2,3,4,6,7,8-HPCDD	Walleye	3	3	100	1.7E-03	2.3E-03	0.001	1.7E-06	2.3E-06	< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,4,6,7,8-HPCDF	Walleye	3	2	67	4.4E-04	6.4E-04	0.01	4.4E-06	6.4E-06	< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,4,7,8,9-HPCDF	Walleye	3	0										
1,2,3,4,7,8-HXCDD	Walleye	3	2	67	3.3E-04	4.7E-04	0.5	1.7E-04	2.4E-04	< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,4,7,8-HXCDF	Walleye	3	3	100	2.7E-04	2.8E-04	0.1	2.7E-05	2.8E-05	< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,6,7,8-HXCDD	Walleye	3	3	100	1.6E-03	1.8E-03	0.01	1.6E-05	1.8E-05	< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,6,7,8-HXCDF	Walleye	3	3	100	4.0E-04	4.5E-04	0.1	4.0E-05	4.5E-05	< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,7,8,9-HXCDD	Walleye	3	2	67	2.0E-04	2.7E-04	0.01	2.0E-06	2.7E-06	< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,7,8,9-HXCDF	Walleye	3	0										
1,2,3,7,8-PECDD	Walleye	3	3	100	1.1E-03	1.2E-03	1	1.1E-03	1.2E-03	< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,7,8-PECDF	Walleye	3	3	100	1.1E-03	1.2E-03	0.05	5.4E-05	6.0E-05	< 0.1	< 0.1	< 0.1	< 0.1
2,3,4,6,7,8-HXCDF	Walleye	3	3	100	5.4E-04	6.1E-04	0.1	5.4E-05	6.1E-05	< 0.1	< 0.1	< 0.1	< 0.1
2,3,4,7,8-PECDF	Walleye	3	3	100	1.2E-03	1.6E-03	0.5	6.2E-04	8.0E-04	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 77	Walleye	4	4	100	4.5	7.0	0.0001	4.5E-04	7.0E-04	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 81	Walleye	1	0										
PCB Congener 105	Walleye	4	4	100	29.9	39.0	0.000005	1.5E-04	1.9E-04	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 118	Walleye	4	4	100	77.0	98.1	0.000005	3.9E-04	4.9E-04	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 126	Walleye	4	4	100	0.3	0.4	0.005	1.4E-03	1.9E-03	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 169	Walleye	4	1	25	0.02	0.1	0.00005	1.0E-06	3.2E-06	< 0.1	< 0.1	< 0.1	< 0.1
Total TEQ	**Walleye**							**5.7E-03**	**7.4E-03**	**0.1**	**0.2**	**0.1**	**0.1**

Notes:

NA - Not applicable.

TEQ Criteria: NOAEC = 0.041 μg/kg; LOAEC = 0.084 μg/kg.

Table 6-90 Estimated Hazard Quotients for Piscivorous Birds in Little Rapids to De Pere Reach

Analyte	Total Estimated Exposure (µg/kg-BW/day)							
	Common Tern		Forster's Tern		Double-crested Cormorant		Bald Eagle	
	Mean	RME	Mean	RME	Mean	RME	Mean	RME
Mercury	12.7	25.3	11.7	23.4	4.9	9.8	17.4	17.5
Total PCBs	170	181	157	167	65.6	70.0	427	630
Dieldrin	0.6	1.2	0.6	1.1	0.2	0.5	0.2	1.3
p,p'-DDE	7.8	7.8	7.2	7.2	3.0	3.0	9.6	16.3

Analyte	NOAEC HQs							
	Common Tern		Forster's Tern		Double-crested Cormorant		Bald Eagle	
	Mean	RME	Mean	RME	Mean	RME	Mean	RME
Mercury	1.6	3.2	1.5	2.9	0.6	1.2	2.2	2.2
Total PCBs	1.5	1.6	1.4	1.5	0.6	0.6	3.8	5.6
Dieldrin	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1
p,p'-DDE	0.4	0.4	0.4	0.4	0.2	0.2	0.5	0.9

Analyte	LOAEC HQs							
	Common Tern		Forster's Tern		Double-crested Cormorant		Bald Eagle	
	Mean	RME	Mean	RME	Mean	RME	Mean	RME
Mercury	0.2	0.3	0.2	0.3	0.1	0.1	0.2	0.2
Total PCBs	0.2	0.2	0.1	0.1	0.1	0.1	0.4	0.6
Dieldrin	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1
p,p'-DDE	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	0.1	0.1

Table 6-91 Estimated Hazard Quotients for Mink in Little Rapids to De Pere Reach

Analyte	Total Estimated Exposure (µg/kg-BW/day) Mean	RME
Mercury	46.6	49.4
Total PCBs (N)	773	1,162
Total PCBs (I_0)	760	1,120
Total PCBs (I_d)	760	1,120
Dieldrin	0.4	2.5
p,p'-DDE	14.3	24.5
Analyte	**NOAEC HQs Mean**	**RME**
Mercury	0.6	0.6
Total PCBs	193	291
Total PCBs (I_0)	190	280
Total PCBs (I_d)	190	280
Dieldrin	< 0.1	0.3
p,p'-DDE	< 0.1	< 0.1
Analyte	**LOAEC HQs Mean**	**RME**
Mercury	0.2	0.2
Total PCBs	5.9	8.9
Total PCBs (I_0)	5.8	8.6
Total PCBs (I_d)	5.8	8.6
Dieldrin	< 0.1	0.1
p,p'-DDE	< 0.1	< 0.1

Table 6-92 Hazard Quotients for Bird Tissue in Little Rapids to De Pere Reach

Analyte	Species	Tissue	Number of Samples	Number of Detects	Detection Frequency (%)	Mean	RME	Reproduction Criteria		Hazard Quotients NOAEC		Hazard Quotients LOAEC		Deformity Criteria		Hazard Quotients NOAEC		Hazard Quotients LOAEC	
								NOAEC	LOAEC	Mean	RME	Mean	RME	NOAEC	LOAEC	Mean	RME	Mean	RME
PCBs (µg/kg)																			
Total PCBs	Tree Swallow	whole	22	22	100	3,117.73	4,505.22	4,700	7,600	0.66	0.96	0.41	0.59	800	8,000	3.9	5.6	0.4	0.6
Pesticides (µg/kg)																			
Dieldrin	Tree Swallow	whole	22	0															
o,p'-DDD	Tree Swallow	whole	22	0															
o,p'-DDE	Tree Swallow	whole	22	0															
o,p'-DDT	Tree Swallow	whole	22	0															
p,p'-DDD	Tree Swallow	whole	22	3	14	6.1	7.1	3,000	5,100	< 0.1	< 0.1	< 0.1	< 0.1						
p,p'-DDE	Tree Swallow	whole	22	22	100	218	331	3,000	5,100	0.1	0.1	< 0.1	0.1						
p,p'-DDT	Tree Swallow	whole	22	0															

Table 6-93 Estimated Hazard Quotients for Mink in Green Bay Zone 1

Analyte	Total Estimated Exposure (µg/kg-BW/day) Based on Carp Consumption	RME
Mercury	16.5	20.4
Total PCBs (N)	1,290	1,437
Total PCBs (I_0)	1,284	1,424
Total PCBs (I_d)	1,284	1,424
Dieldrin	4.0	5.7
p,p'-DDE	37.6	134
Analyte	**NOAEC HQs** **Mean**	**RME**
Mercury	0.2	0.2
Total PCBs	323	359
Total PCBs (I_0)	321	356
Total PCBs (I_d)	321	356
Dieldrin	0.4	0.6
p,p'-DDE	< 0.1	< 0.1
Analyte	**LOAEC HQs** **Mean**	**RME**
Mercury	0.1	0.1
Total PCBs	9.9	11
Total PCBs (I_0)	9.9	11
Total PCBs (I_d)	9.9	11
Dieldrin	0.2	0.3
p,p'-DDE	< 0.1	< 0.1

Table 6-94 Hazard Quotients for Whole Fish in Green Bay Zones 1 and 2

Analyte	Species	Number of Samples	Number of Detects	Detection Frequency (%)	Mean	RME	Criteria		Hazard Quotients			
							NOAEC	LOAEC	NOAEC		LOAEC	
									Mean	RME	Mean	RME
Metals (mg/kg)												
Mercury	Alewife	5	2	40	0.1	0.3	0.25	2.37	0.4	1.0	< 0.1	0.1
	Gizzard Shad	7	0									
	Rainbow Smelt	4	4	100	0.03	0.04	0.25	2.37	0.1	0.1	< 0.1	< 0.1
	Yellow Perch	9	0				0.25	2.37				
	Carp	10	1	10	0.1	0.2	0.25	2.37	0.3	0.6	< 0.1	0.1
	Walleye	11	10	91	0.2	0.3	0.25	2.37	0.9	1.1	0.1	0.1
PCBs (µg/kg)												
Total PCBs	Alewife	51	51	100	2,599	3,182	760	7,600	3.4	4.2	0.3	0.4
	Gizzard Shad	50	50	100	1,852	2,005	760	7,600	2.4	2.6	0.2	0.3
	Rainbow Smelt	33	33	100	1,049	1,152	760	7,600	1.4	1.5	0.1	0.2
	Common Shiner	5	5	100	3,520	3,846	760	7,600	4.6	5.1	0.5	0.5
	Emerald Shiners	5	5	100	3,520	3,846	760	7,600	4.6	5.1	0.5	0.5
	Golden Shiner	2	2	100	1,385	1,443	760	7,600	1.8	1.9	0.2	0.2
	Yellow Perch	9	9	100	1,206	1,567	760	7,600	1.6	2.1	0.2	0.2
	Carp	115	115	100	6,637	7,369	760	7,600	8.7	9.7	0.9	1.0
	Walleye	91	91	100	6,539	7,658	760	7,600	8.6	10	0.9	1.0
Pesticides (µg/kg)												
Dieldrin	Alewife	51	45	88	21.0	57.9	370	3,700	0.1	0.2	< 0.1	< 0.1
	Gizzard Shad	46	22	48	10.5	48.4	370	3,700	< 0.1	0.1	< 0.1	< 0.1
	Rainbow Smelt	33	29	88	7.5	8.7	370	3,700	< 0.1	< 0.1	< 0.1	< 0.1
	Yellow Perch	9	0									
	Carp	78	66	85	20.8	29.4	370	3,700	0.1	0.1	< 0.1	< 0.1
	Walleye	70	58	83	37.3	52.2	370	3,700	0.1	0.1	< 0.1	< 0.1
o,p'-DDD	Gizzard Shad	15	0									
	Rainbow Smelt	4	0									
	Carp	4	0									
	Walleye	3	0									
o,p'-DDE	Gizzard Shad	8	0									
	Rainbow Smelt	4	0									
	Carp	4	3	75	50.0	88.0	300	2,950	0.2	0.3	< 0.1	< 0.1
	Walleye	3	3	100	85.0	120	300	2,950	0.3	0.4	< 0.1	< 0.1

Table 6-94 Hazard Quotients for Whole Fish in Green Bay Zones 1 and 2 (Continued)

Analyte	Species	Number of Samples	Number of Detects	Detection Frequency (%)	Mean	RME	Criteria		Hazard Quotients			
							NOAEC	LOAEC	NOAEC		LOAEC	
									Mean	RME	Mean	RME
o,p'-DDT	Gizzard Shad	15	0									
	Rainbow Smelt	4	0									
	Carp	4	0									
	Walleye	3	0									
p,p'-DDD	Alewife	5	1	20	7.3	10.6	300	2,950	< 0.1	< 0.1	< 0.1	< 0.1
	Gizzard Shad	22	1	5	22.8	26.0	300	2,950	0.1	0.1	< 0.1	< 0.1
	Yellow Perch	9	0									
	Carp	13	3	23	31.8	79.0	300	2,950	0.1	0.3	< 0.1	< 0.1
	Walleye	14	1	7	23.5	33.0	300	2,950	0.1	0.1	< 0.1	< 0.1
p,p'-DDE	Alewife	5	5	100	104	143	300	2,950	0.3	0.5	< 0.1	< 0.1
	Gizzard Shad	22	8	36	64.2	93.6	300	2,950	0.2	0.3	< 0.1	< 0.1
	Rainbow Smelt	4	0									
	Yellow Perch	9	9	100	32.9	45.1	300	2,950	0.1	0.2	< 0.1	< 0.1
	Carp	13	13	100	197	700	300	2,950	0.7	2.3	0.1	0.2
	Walleye	14	14	100	353	462	300	2,950	1.2	1.5	0.1	0.2
p,p'-DDT	Alewife	5	0									
	Gizzard Shad	22	0									
	Rainbow Smelt	4	0									
	Yellow Perch	9	0									
	Carp	13	0									
	Walleye	14	0									

Note:

Lead TRVs are based on egg concentrations.

Table 6-95 PCB Congener and Dioxin/Furan Hazard Quotients for Whole Fish in Green Bay Zones 1 and 2

Analyte	Species	Number of Samples	Number of Detects	Detection Frequency (%)	Mean	RME	TEF	Mean TEC	RME TEC	Hazard Quotients			
										NOAEC		LOAEC	
										Mean	RME	Mean	RME
PCBs and Dioxins/Furans (µg/kg)													
PCB Congener 77	Alewife	5	5	100	0.9	1.3	0.0001	8.62E-05	1.27E-04	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 81	Alewife	51	44	86	6.4	48.0	0.0005	3.19E-03	2.40E-02	0.1	0.6	< 0.1	0.3
PCB Congener 105	Alewife	5	5	100	19.8	27.6	0.000005	9.90E-05	1.38E-04	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 118	Alewife	51	51	100	66.2	80.7	0.000005	3.31E-04	4.04E-04	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 126	Alewife	5	5	100	0.2	0.5	0.005	1.10E-03	2.38E-03	< 0.1	0.1	< 0.1	< 0.1
PCB Congener 169	Alewife	5	0										
Total TEQ	**Alewife**							**4.80E-03**	**2.70E-02**	**0.1**	**0.7**	**0.1**	**0.3**
PCB Congener 77	Gizzard Shad	6	6	100	1.0	1.6	0.0001	1.05E-04	1.59E-04	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 81	Gizzard Shad	29	25	86	3.0	4.6	0.0005	1.48E-03	2.30E-03	< 0.1	0.1	< 0.1	< 0.1
PCB Congener 105	Gizzard Shad	6	6	100	19.1	25.1	0.000005	9.55E-05	1.25E-04	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 118	Gizzard Shad	29	28	97	32.3	63.0	0.000005	1.62E-04	3.15E-04	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 126	Gizzard Shad	6	5	83	0.2	0.3	0.005	9.00E-04	1.30E-03	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 169	Gizzard Shad	6	1	17	0.02	0.1	0.00005	1.13E-06	5.50E-06	< 0.1	< 0.1	< 0.1	< 0.1
Total TEQ	**Gizzard Shad**							**2.74E-03**	**4.20E-03**	**0.1**	**0.1**	**0.0**	**0.1**
PCB Congener 77/110	Rainbow Smelt	29	29	100	41.6	45.7	0.0001	4.16E-03	4.57E-03	0.1	0.1	< 0.1	0.1
PCB Congener 81	Rainbow Smelt	29	29	100	3.7	4.2	0.0005	1.83E-03	2.10E-03	< 0.1	0.1	< 0.1	< 0.1
PCB Congener 132/153/105	Rainbow Smelt	29	29	100	32.6	37.3	0.000005	1.63E-04	1.87E-04	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 118	Rainbow Smelt	29	29	100	26.9	29.6	0.000005	1.34E-04	1.48E-04	< 0.1	< 0.1	< 0.1	< 0.1
Total TEQ	**Rainbow Smelt**							**6.29E-03**	**7.00E-03**	**0.2**	**0.2**	**0.1**	**0.1**
PCB Congener 77	Golden Shiner	2	2	100	2.7	3.1	0.0001	2.71E-04	3.09E-04	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 81/87/115	Golden Shiner	2	2	100	24.0	25.0	0.0005	1.20E-02	1.25E-02	0.3	0.3	0.1	0.1
PCB Congener 105	Golden Shiner	2	2	100	10.6	11.8	0.000005	5.29E-05	5.92E-05	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 118	Golden Shiner	2	2	100	27.8	32.2	0.000005	1.39E-04	1.61E-04	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 126	Golden Shiner	2	2	100	0.1	0.1	0.005	4.04E-04	4.73E-04	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 169	Golden Shiner	2	0										
Total TEQ	**Golden Shiner**							**1.29E-02**	**1.35E-02**	**0.3**	**0.3**	**0.2**	**0.2**
PCB Congener 77	Yellow Perch	9	9	100	0.2	0.8	0.0001	2.26E-05	7.60E-05	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 81	Yellow Perch	9	0										
PCB Congener 105	Yellow Perch	9	8	89	12.8	18.0	0.000005	6.39E-05	9.01E-05	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 118	Yellow Perch	9	9	100	34.4	45.4	0.000005	1.72E-04	2.27E-04	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 126	Yellow Perch	9	2	22	0.01	0.02	0.005	5.22E-05	1.10E-04	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 169	Yellow Perch	9	0										
Total TEQ	**Yellow Perch**							**3.11E-04**	**5.03E-04**	**< 0.1**	**< 0.1**	**< 0.1**	**< 0.1**

Table 6-95 PCB Congener and Dioxin/Furan Hazard Quotients for Whole Fish in Green Bay Zones 1 and 2 (Continued)

Analyte	Species	Number of Samples	Number of Detects	Detection Frequency (%)	Mean	RME	TEF	Mean TEC	RME TEC	Hazard Quotients			
										NOAEC		LOAEC	
										Mean	RME	Mean	RME
2,3,7,8-TCDD	Carp	3	3	100	9.8E-04	1.3E-03	1	9.83E-04	1.30E-03	< 0.1	< 0.1	< 0.1	< 0.1
2,3,7,8-TCDF	Carp	3	3	100	2.9E-03	4.6E-03	0.05	1.45E-04	2.30E-04	< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,4,6,7,8,9-OCDD	Carp	3	3	100	3.2E-02	7.5E-02	0.0001	3.20E-06	7.54E-06	< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,4,6,7,8,9-OCDF	Carp	3	0										
1,2,3,4,6,7,8-HPCDD	Carp	3	3	100	1.9E-02	4.4E-02	0.001	1.90E-05	4.40E-05	< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,4,6,7,8-HPCDF	Carp	3	3	100	2.6E-03	5.6E-03	0.01	2.55E-05	5.60E-05	< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,4,7,8,9-HPCDF	Carp	3	0										
1,2,3,4,7,8-HXCDD	Carp	3	3	100	2.3E-03	4.8E-03	0.5	1.14E-03	2.40E-03	< 0.1	0.1	< 0.1	< 0.1
1,2,3,4,7,8-HXCDF	Carp	3	2	67	1.0E-03	2.4E-03	0.1	1.01E-04	2.40E-04	< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,6,7,8-HXCDD	Carp	3	3	100	6.2E-03	1.3E-02	0.01	6.17E-05	1.31E-04	< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,6,7,8-HXCDF	Carp	3	3	100	1.8E-03	4.1E-03	0.1	1.79E-04	4.10E-04	< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,7,8,9-HXCDD	Carp	3	3	100	5.3E-04	1.0E-03	0.01	5.30E-06	1.00E-05	< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,7,8,9-HXCDF	Carp	3	1	33	6.7E-05	1.7E-04	0.1	6.67E-06	1.70E-05	< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,7,8-PECDD	Carp	3	3	100	1.1E-03	1.6E-03	1	1.12E-03	1.60E-03	< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,7,8-PECDF	Carp	3	2	67	1.2E-03	2.8E-03	0.05	5.98E-05	1.40E-04	< 0.1	< 0.1	< 0.1	< 0.1
2,3,4,6,7,8-HXCDF	Carp	3	3	100	6.4E-04	7.8E-04	0.1	6.40E-05	7.80E-05	< 0.1	< 0.1	< 0.1	< 0.1
2,3,4,7,8-PECDF	Carp	3	3	100	2.2E-03	3.6E-03	0.5	1.10E-03	1.80E-03	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 77	Carp	14	14	100	1.4E+00	6.0E+00	0.0001	1.39E-04	6.00E-04	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 81	Carp	77	69	90	1.4E+01	1.6E+01	0.0005	7.03E-03	8.08E-03	0.2	0.2	0.1	0.1
PCB Congener 105	Carp	14	14	100	3.7E+01	1.4E+02	0.000005	1.87E-04	6.90E-04	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 118	Carp	80	80	100	1.4E+02	1.6E+02	0.000005	6.89E-04	7.85E-04	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 126	Carp	14	10	71	3.2E-01	2.5E+00	0.005	1.60E-03	1.25E-02	< 0.1	0.3	< 0.1	0.1
PCB Congener 169	Carp	16	3	19	1.0E-01	2.7E-01	0.00005	5.02E-06	1.34E-05	< 0.1	< 0.1	< 0.1	< 0.1
Total TEQ	**Carp**							**1.47E-02**	**3.11E-02**	**0.4**	**0.8**	**0.2**	**0.4**

Table 6-95 PCB Congener and Dioxin/Furan Hazard Quotients for Whole Fish in Green Bay Zones 1 and 2 (Continued)

Analyte	Species	Number of Samples	Number of Detects	Detection Frequency (%)	Mean	RME	TEF	Mean TEC	RME TEC	Hazard Quotients			
										NOAEC		LOAEC	
										Mean	RME	Mean	RME
2,3,7,8-TCDD	Walleye	3	3	100	1.4E-03	2.0E-03	1	1.40E-03	2.00E-03	< 0.1	< 0.1	< 0.1	< 0.1
2,3,7,8-TCDF	Walleye	3	3	100	1.7E-02	1.9E-02	0.05	8.45E-04	9.70E-04	< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,4,6,7,8,9-OCDD	Walleye	3	3	100	3.3E-03	4.5E-03	0.0001	3.30E-07	4.50E-07	< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,4,6,7,8,9-OCDF	Walleye	3	0										
1,2,3,4,6,7,8-HPCDD	Walleye	3	3	100	3.7E-03	5.2E-03	0.001	3.73E-06	5.20E-06	< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,4,6,7,8-HPCDF	Walleye	3	3	100	9.7E-04	1.3E-03	0.01	9.70E-06	1.30E-05	< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,4,7,8,9-HPCDF	Walleye	3	1	33	8.3E-05	1.7E-04	0.01	8.33E-07	1.70E-06	< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,4,7,8-HXCDD	Walleye	3	3	100	7.2E-04	9.5E-04	0.5	3.60E-04	4.75E-04	< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,4,7,8-HXCDF	Walleye	3	3	100	4.7E-04	6.2E-04	0.1	4.70E-05	6.20E-05	< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,6,7,8-HXCDD	Walleye	3	3	100	3.4E-03	4.6E-03	0.01	3.40E-05	4.60E-05	< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,6,7,8-HXCDF	Walleye	3	3	100	7.1E-04	9.6E-04	0.1	7.10E-05	9.60E-05	< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,7,8,9-HXCDD	Walleye	3	3	100	4.8E-04	5.9E-04	0.01	4.77E-06	5.90E-06	< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,7,8,9-HXCDF	Walleye	3	1	33	5.8E-05	8.0E-05	0.1	5.83E-06	8.00E-06	< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,7,8-PECDD	Walleye	3	3	100	2.4E-03	3.1E-03	1	2.37E-03	3.10E-03	0.1	0.1	< 0.1	< 0.1
1,2,3,7,8-PECDF	Walleye	3	3	100	1.7E-03	2.2E-03	0.05	8.50E-05	1.10E-04	< 0.1	< 0.1	< 0.1	< 0.1
2,3,4,6,7,8-HXCDF	Walleye	3	3	100	9.5E-04	1.3E-03	0.1	9.50E-05	1.30E-04	< 0.1	< 0.1	< 0.1	< 0.1
2,3,4,7,8-PECDF	Walleye	3	3	100	3.3E-03	3.9E-03	0.5	1.63E-03	1.95E-03	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 77	Walleye	16	16	100	4.9	6.5	0.0001	4.90E-04	6.48E-04	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 81	Walleye	69	65	94	15.7	61.0	0.0005	7.85E-03	3.05E-02	0.2	0.7	0.1	0.4
PCB Congener 105	Walleye	27	25	93	69.0	251	0.000005	3.45E-04	1.26E-03	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 118	Walleye	83	83	100	174	199	0.000005	8.71E-04	9.97E-04	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 126	Walleye	16	15	94	1.1	5.3	0.005	5.29E-03	2.65E-02	0.1	0.6	0.1	0.3
PCB Congener 169	Walleye	25	16	64	0.3	1.7	0.00005	1.53E-05	8.48E-05	< 0.1	< 0.1	< 0.1	< 0.1
Total TEQ	**Walleye**							**2.18E-02**	**6.90E-02**	**0.5**	**1.7**	**0.3**	**0.8**

Note:

TEQ Criteria: NOAEC = 0.041 μg/kg; LOAEC = 0.084 μg/kg.

Table 6-96 Hazard Quotients for Bird Tissue in Green Bay Zone 2

Analyte	Species	Tissue	Number of Samples	Number of Detects	Detection Frequency (%)	Mean	RME	Reproduction Criteria		Hazard Quotients NOAEC		Hazard Quotients LOAEC		Deformity Criteria		Hazard Quotients NOAEC		Hazard Quotients LOAEC	
								NOAEC	LOAEC	Mean	RME	Mean	RME	NOAEC	LOAEC	Mean	RME	Mean	RME
PCBs (µg/kg)																			
Total PCBs	Double-crested Cormorant	egg	34	34	100	13,944	21,127	4,700	7,600	3.0	4.5	1.8	2.8	800	8,000	17	26	1.7	2.6
	Double-crested Cormorant	whole	74	74	100	11,026	13,870	4,700	7,600	2.3	3.0	1.5	1.8	800	8,000	14	17	1.4	1.7
	Common Tern	egg	10	10	100	4,819	5,963	4,700	7,600	1.0	1.3	0.6	0.8	800	8,000	6.0	7.5	0.6	0.7
	Forster's Tern	egg	10	10	100	5,077	6,234	4,700	7,600	1.1	1.3	0.7	0.8	800	8,000	6.3	7.8	0.6	0.8
	Tree Swallow	whole	15	15	100	2,980	3,495	4,700	7,600	0.6	0.7	0.4	0.5	800	8,000	3.7	4.4	0.4	0.4
Pesticides (µg/kg)																			
Dieldrin	Double-crested Cormorant	brain	5	5	100	48.2	60.5	490	4,900	0.1	0.1	< 0.1	< 0.1						
	Double-crested Cormorant	egg	34	32	94	224	445	100	1,000	2.2	4.4	0.2	0.4						
	Double-crested Cormorant	whole	73	73	100	196	243	100	1,000	2.0	2.4	0.2	0.2						
	Common Tern	egg	5	5	100	85.0	139	100	1,000	0.9	1.4	0.1	0.1						
	Forster's Tern	egg	7	7	100	47.6	62.7	100	1,000	0.5	0.6	< 0.1	0.1						
	Tree Swallow	whole	15	0															
o,p'-DDD	Double-crested Cormorant	brain	5	0															
	Double-crested Cormorant	egg	34	0															
	Double-crested Cormorant	whole	73	0															
	Common Tern	egg	5	0															
	Forster's Tern	egg	7	0															
	Tree Swallow	whole	15	0															
o,p'-DDE	Double-crested Cormorant	brain	5	0															
	Double-crested Cormorant	egg	34	0															
	Double-crested Cormorant	whole	73	0															
	Common Tern	egg	5	0															
	Forster's Tern	egg	7	0															
	Tree Swallow	whole	15	0															
o,p'-DDT	Double-crested Cormorant	brain	5	0															
	Double-crested Cormorant	egg	34	0															
	Double-crested Cormorant	whole	73	0															
	Common Tern	egg	5	0															
	Forster's Tern	egg	7	0															
	Tree Swallow	whole	15	0															
p,p'-DDD	Double-crested Cormorant	brain	5	0															
	Double-crested Cormorant	egg	34	22	65	15.0	20.1	3,000	5,100	< 0.1	< 0.1	< 0.1	< 0.1						
	Double-crested Cormorant	whole	73	14	19	7.3	8.4	3,000	5,100	< 0.1	< 0.1	< 0.1	< 0.1						
	Common Tern	egg	5	5	100	2.1	3.2	3,000	5,100	< 0.1	< 0.1	< 0.1	< 0.1						
	Forster's Tern	egg	7	4	57	NE	2.7	3,000	5,100	NE	NE	< 0.1	< 0.1						
	Tree Swallow	whole	15	3	20	6.5	8.0	3,000	5,100	< 0.1	< 0.1	< 0.1	< 0.1						

Table 6-96 Hazard Quotients for Bird Tissue in Green Bay Zone 2 (Continued)

Analyte	Species	Tissue	Number of Samples	Number of Detects	Detection Frequency (%)	Mean	RME	Reproduction Criteria		Hazard Quotients				Deformity Criteria		Hazard Quotients			
										NOAEC		LOAEC				NOAEC		LOAEC	
								NOAEC	LOAEC	Mean	RME	Mean	RME	NOAEC	LOAEC	Mean	RME	Mean	RME
p,p'-DDE	Double-crested Cormorant	brain	5	5	100	534	643	1,800	18,000	< 0.1	< 0.1	< 0.1	< 0.1						
	Double-crested Cormorant	egg	34	34	100	4,132	7,277	3,000	5,100	1.4	2.4	0.8	1.4						
	Double-crested Cormorant	whole	73	73	100	2,756	3,523	3,000	5,100	0.9	1.2	0.5	0.7						
	Common Tern	egg	5	5	100	666	893	3,000	5,100	0.2	0.3	0.1	0.2						
	Forster's Tern	egg	7	7	100	447	576	3,000	5,100	0.1	0.2	0.1	0.1						
	Tree Swallow	whole	15	15	100	128	187	3,000	5,100	< 0.1	0.1	< 0.1	< 0.1						
p,p'-DDT	Double-crested Cormorant	brain	5	0															
	Double-crested Cormorant	egg	34	3	9	7.6	10.1	3,000	5,100	< 0.1	< 0.1	< 0.1	< 0.1						
	Double-crested Cormorant	whole	73	19	26	8.1	9.3	3,000	5,100	< 0.1	< 0.1	< 0.1	< 0.1						
	Common Tern	egg	5	0															
	Forster's Tern	egg	7	0															
	Tree Swallow	whole	15	0															

Notes:

All tern data is from Kidney Island.

NE indicates that the calculated mean exceeded the detected maximum and, therefore, the mean was not evaluated.

Table 6-97 Estimated Hazard Quotients for Piscivorous Birds in Green Bay Zones 1 and 2

Analyte	Total Estimated Exposure (µg/kg-BW/day)							
	Common Tern		Forster's Tern		Double-crested Cormorant		Bald Eagle	
	Mean	RME	Mean	RME	Mean	RME	Mean	RME
Mercury	49.1	123	45.3	114	19.0	47.6	10.2	12.5
Total PCBs	1,274	1,559	1,174	1,438	492	602	750	842
Dieldrin	10.3	28.4	9.5	26.2	4.0	11.0	2.7	3.8
p,p'-DDE	51.1	70.0	47.1	64.6	19.7	27.0	25.8	74.0

Analyte	NOAEC HQs							
	Common Tern		Forster's Tern		Double-crested Cormorant		Bald Eagle	
	Mean	RME	Mean	RME	Mean	RME	Mean	RME
Mercury	6.1	15	5.7	14	2.4	5.9	1.3	1.6
Total PCBs	11	14	10	13	4.4	5.4	6.7	7.5
Dieldrin	0.1	0.3	0.1	0.2	< 0.1	0.1	< 0.1	< 0.1
p,p'-DDE	2.8	3.9	2.6	3.6	1.1	1.5	1.4	4.1

Analyte	LOAEC HQs							
	Common Tern		Forster's Tern		Double-crested Cormorant		Bald Eagle	
	Mean	RME	Mean	RME	Mean	RME	Mean	RME
Mercury	0.6	1.6	0.6	1.5	0.2	0.6	0.1	0.2
Total PCBs	1.1	1.4	1.0	1.3	0.4	0.5	0.7	0.8
Dieldrin	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1
p,p'-DDE	0.3	0.4	0.3	0.4	0.1	0.2	0.1	0.4

Table 6-98 PCB Congener and Dioxin/Furan Hazard Quotients for Whole Tree Swallows in Green Bay Zone 2

Analyte	Tissue	Number of Samples	Number of Detects	Detection Frequency (%)	Mean	RME	Tillitt *et al.*, 1991b TEF	Tillitt *et al.*, 1991b Mean TEC	Tillitt *et al.*, 1991b RME TEC	Van den Berg *et al.*, 1998 TEF	Van den Berg *et al.*, 1998 Mean TEC	Van den Berg *et al.*, 1998 RME TEC
PCBs (µg/kg)												
PCB Congener 77	whole	15	1	100	0.1	0.3	0.000018	2.4E-06	5.0E-06	0.05	6.7E-03	1.4E-02
PCB Congener 105	whole	15	15	100	37.8	44.2	0.0000076	2.9E-04	3.4E-04	0.0001	3.8E-03	4.4E-03
PCB Congener 118/106	whole	15	15	100	85.9	97	0.00000037	3.2E-05	3.6E-05	1E-05	8.6E-04	9.7E-04
PCB Congener 126	whole	15	8	100	0.3	0.7	0.022	6.1E-03	1.5E-02	0.1	2.8E-02	6.9E-02
PCB Congener 169	whole	15	0									
Total TEQ								**6.5E-03**	**1.6E-02**		**3.9E-02**	**8.8E-02**

Analyte	Tissue	Number of Samples	Number of Detects	Detection Frequency (%)	Mean	RME	Reproduction Hazard Quotients (based on Tillitt *et al.*, 1991b TEFs) NOAEC Mean	NOAEC RME	LD_{20} Mean	LD_{20} RME	LD_{30} Mean	LD_{30} RME	Reproduction Hazard Quotients (based on Van den Berg *et al.*, 1998 TEFs) NOAEC Mean	NOAEC RME	LD_{20} Mean	LD_{20} RME	LD_{30} Mean	LD_{30} RME
PCBs (µg/kg)																		
PCB Congener 77	whole	15	1	100	0.1	0.3	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	1.0	2.0	< 0.1	0.1	< 0.1	< 0.1
PCB Congener 105	whole	15	15	100	37.8	44.2	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	0.5	0.6	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 118/106	whole	15	15	100	85.9	97	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	0.1	0.1	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 126	whole	15	8	100	0.3	0.7	0.9	2.2	< 0.1	0.1	< 0.1	< 0.1	4.0	10	0.1	0.4	0.1	0.2
PCB Congener 169	whole	15	0															
Total TEQ							**0.9**	**2.2**	**< 0.1**	**0.1**	**< 0.1**	**0.1**	**5.6**	**12.6**	**0.2**	**0.5**	**0.1**	**0.3**

Analyte	Tissue	Number of Samples	Number of Detects	Detection Frequency (%)	Mean	RME	Deformity Hazard Quotients (based on Tillitt *et al.*, 1991b TEFs) NOAEC Mean	NOAEC RME	LOAEC Mean	LOAEC RME	Deformity Hazard Quotients (based on Van den Berg *et al.*, 1998 TEFs) NOAEC Mean	NOAEC RME	LOAEC Mean	LOAEC RME
PCBs (µg/kg)														
PCB Congener 77	whole	15	1	100	0.1	0.3	< 0.1	< 0.1	< 0.1	< 0.1	0.2	0.4	< 0.1	< 0.1
PCB Congener 105	whole	15	15	100	37.8	44.2	< 0.1	< 0.1	< 0.1	< 0.1	0.1	0.1	< 0.1	< 0.1
PCB Congener 118/106	whole	15	15	100	85.9	97	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 126	whole	15	8	100	0.3	0.7	0.2	0.4	< 0.1	< 0.1	0.7	1.8	0.1	0.2
PCB Congener 169	whole	15	0											
Total TEQ							**0.2**	**0.4**	**< 0.1**	**< 0.1**	**1.3**	**2.5**	**0.1**	**0.2**

Table 6-99 PCB Congener and Dioxin/Furan Hazard Quotients for Double-crested Cormorant Eggs and Whole Bodies in Green Bay Zone 2

Analyte	Tissue	Number of Samples	Number of Detects	Detection Frequency (%)	Mean	RME	Tillitt *et al.*, 1991b			Van den Berg *et al.*, 1998			Reproduction Hazard Quotients (based on Tillitt *et al.*, 1991b TEFs)						Reproduction Hazard Quotients (based on Van den Berg *et al.*, 1998 TEFs)						Deformity Hazard Quotients (based on Tillitt *et al.*, 1991b TEFs)				Deformity Hazard Quotients (based on Van den Berg *et al.*, 1998 TEFs)			
							TEF	Mean TEC	RME TEC	TEF	Mean TEC	RME TEC	NOAEC		LD_{20}		LD_{30}		NOAEC		LD_{20}		LD_{30}		NOAEC		LOAEC		NOAEC		LOAEC	
													Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME
PCB Congeners and Dioxins/Furans (µg/kg)																																
1,2,3,4,6,7,8,9-OCDD	whole	4	4	100	1.4E-01	2.1E-01				1	1.4E-01	2.1E-01							19	31	0.7	1.1	0.4	0.7					3.6	5.6	0.4	0.6
1,2,3,4,6,7,8,9-OCDF	whole	4	0																													
1,2,3,4,6,7,8-HPCDD	whole	4	0																													
1,2,3,4,6,7,8-HPCDF	whole	4	0																													
1,2,3,4,7,8,9-HPCDF	whole	4	0																													
1,2,3,4,7,8-HXCDD	whole	4	0																													
1,2,3,4,7,8-HXCDF	whole	4	0																													
1,2,3,6,7,8-HXCDD	whole	4	0																													
1,2,3,6,7,8-HXCDF	whole	4	0																													
1,2,3,7,8,9-HXCDD	whole	4	0																													
1,2,3,7,8,9-HXCDF	whole	4	0																													
1,2,3,7,8-PECDD	whole	4	0																													
1,2,3,7,8-PECDF	whole	4	0																													
2,3,4,6,7,8-HXCDF	whole	4	0																													
2,3,4,7,8-PECDF	whole	4	0																													
2,3,7,8-TCDD	whole	4	1	25	4.7E-03	9.6E-03	1	4.7E-03	9.6E-03	1	4.7E-03	9.6E-03	0.7	1.4	< 0.1	0.1	< 0.1	< 0.1	0.7	1.4	< 0.1	0.1	< 0.1	< 0.1	0.1	0.3	< 0.1	< 0.1	0.1	0.3	< 0.1	< 0.1
2,3,7,8-TCDF	whole	4	0																										< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 77	whole	26	9	35	0.3	0.5	0.000018	5.5E-06	9.3E-06	0.05	1.5E-02	2.6E-02	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	2.2	3.7	0.1	0.1	< 0.1	0.1	< 0.1	< 0.1	< 0.1	< 0.1	0.4	0.7	< 0.1	0.1
PCB Congener 105	whole	26	26	100	157	215	0.0000076	1.2E-03	1.6E-03	0.0001	1.6E-02	2.2E-02	0.2	0.2	< 0.1	< 0.1	< 0.1	< 0.1	2.2	3.1	0.1	0.1	0.1	0.1	< 0.1	< 0.1	< 0.1	< 0.1	0.4	0.6	< 0.1	0.1
PCB Congener 118/106	whole	26	26	100	379	558	0.00000037	1.4E-04	2.1E-04	0.00001	3.8E-03	5.6E-03	<0.1	<0.1	< 0.1	< 0.1	< 0.1	< 0.1	0.5	0.8	<0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	0.1	0.1	< 0.1	< 0.1
PCB Congener 126	whole	26	19	73	0.7	1.5	0.022	1.5E-02	3.3E-02	0.1	6.8E-02	1.5E-01	2.1	4.7	0.1	0.2	< 0.1	0.1	9.8	21	0.4	0.8	0.2	0.5	0.4	0.9	< 0.1	0.1	1.8	3.9	0.2	0.4
PCB Congener 169	whole	26	7	27	0.1	0.1	0.00047	3.6E-05	4.4E-05	0.001	7.6E-05	9.3E-05	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1
Total TEQ								**2.1E-02**	**4.4E-02**		**2.4E-01**	**4.3E-01**	**3.0**	**6.4**	**0.1**	**0.2**	**0.1**	**0.1**	**35**	**61**	**1.3**	**2.2**	**0.8**	**1.4**	**0.6**	**1.2**	**0.1**	**0.1**	**6.4**	**11**	**0.6**	**1.1**
PCB Congeners and Dioxins/Furans (µg/kg)																																
1,2,3,4,6,7,8,9-OCDD	egg	4	0																													
1,2,3,4,6,7,8,9-OCDF	egg	4	0																													
1,2,3,4,6,7,8-HPCDD	egg	4	0																													
1,2,3,4,6,7,8-HPCDF	egg	4	0																													
1,2,3,4,7,8,9-HPCDF	egg	4	0																													
1,2,3,4,7,8-HXCDD	egg	4	0																													
1,2,3,4,7,8-HXCDF	egg	4	0																													
1,2,3,6,7,8-HXCDD	egg	4	0																													
1,2,3,6,7,8-HXCDF	egg	4	0																													
1,2,3,7,8,9-HXCDD	egg	4	0																													
1,2,3,7,8,9-HXCDF	egg	4	0																													
1,2,3,7,8-PECDD	egg	4	0																													
1,2,3,7,8-PECDF	egg	4	0																													
2,3,4,6,7,8-HXCDF	egg	4	0																													
2,3,4,7,8-PECDF	egg	4	0																													
2,3,7,8-TCDD	egg	4	1	25	0.01	0.02	1	1.2E-02	2.0E-02	1	1.2E-02	2.0E-02	1.7	2.9	0.1	0.1	< 0.1	0.1	1.7	2.9	0.1	0.1	< 0.1	0.1	0.3	0.5	< 0.1	0.1	0.3	0.5	< 0.1	0.1
2,3,7,8-TCDF	egg	4	0																													
PCB Congener 77	egg	12	9	75	1.3	2.3	0.000018	2.3E-05	4.1E-05	0.05	6.5E-02	1.2E-01	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	9.3	16	0.3	0.6	0.2	0.4	< 0.1	< 0.1	< 0.1	< 0.1	1.7	3.0	0.2	0.3
PCB Congener 105	egg	12	12	100	210	303	0.0000076	1.6E-03	2.3E-03	0.0001	2.1E-02	3.0E-02	0.2	0.3	< 0.1	< 0.1	< 0.1	< 0.1	3.0	4.3	0.1	0.2	0.1	0.1	< 0.1	0.1	< 0.1	< 0.1	0.6	0.8	0.1	0.1
PCB Congener 118/106	egg	12	12	100	551	783	0.00000037	2.0E-04	2.9E-04	0.00001	5.5E-03	7.8E-03	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	0.8	1.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	0.1	0.2	< 0.1	< 0.1
PCB Congener 126	egg	12	11	92	1.1	1.5	0.022	2.5E-02	3.3E-02	0.1	1.1E-01	1.5E-01	3.6	4.7	0.1	0.2	0.1	0.1	16	22	0.6	0.8	0.4	0.5	0.7	0.9	0.1	0.1	3.0	4.0	0.3	0.4
PCB Congener 169	egg	12	5	42	0.1	0.2	0.00047	5.4E-05	9.5E-05	0.001	1.2E-04	2.0E-04	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1
Total TEQ								**2.7E-02**	**3.6E-02**		**2.2E-01**	**3.2E-01**	**3.8**	**5.1**	**0.1**	**0.2**	**0.1**	**0.1**	**31**	**46**	**1.1**	**1.7**	**0.7**	**1.1**	**1.0**	**1.5**	**0.1**	**0.1**	**5.7**	**8.5**	**0.6**	**0.9**

Table 6-100 PCB Congener and Dioxin/Furan Hazard Quotients for Common Tern Eggs in Green Bay Zone 2

Analyte	Number of Samples	Number of Detects	Detection Frequency (%)	Mean	RME	Tillitt *et al.*, 1991b			Van den Berg *et al.*, 1998			Reproduction Hazard Quotients (based on Tillitt *et al.*, 1991b TEFs)						Reproduction Hazard Quotients (based on Van den Berg *et al.*, 1998 TEFs)						Deformity Hazard Quotients (based on Tillitt *et al.*, 1991b TEFs)				Deformity Hazard Quotients (based on Van den Berg *et al.*, 1998 TEFs)			
						TEF	Mean TEC	RME TEC	TEF	Mean TEC	RME TEC	NOAEC		LD_{20}		LD_{30}		NOAEC		LD_{20}		LD_{30}		NOAEC		LOAEC		NOAEC		LOAEC	
												Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME
Dioxins/Furans (µg/kg)																															
1,2,3,4,6,7,8,9-OCDD	5	5	100	1.0E-01	1.5E-01				0.0001	1.0E-05	1.5E-05							< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1					< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,4,6,7,8,9-OCDF	1	0																										< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,4,6,7,8-HPCDD	5	5	100	1.2E-02	1.5E-02				0.001	1.2E-05	1.5E-05							< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1					< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,4,6,7,8-HPCDF	5	4	80	1.5E-03	2.4E-03				0.01	1.5E-05	2.4E-05							< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1					< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,4,7,8-HXCDD	5	2	40	1.3E-03	1.8E-03				0.05	6.5E-05	9.0E-05							< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1					< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,4,7,8-HXCDF	5	4	80	1.3E-03	1.9E-03				0.1	1.3E-04	1.9E-04							< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1					< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,6,7,8-HXCDD	5	5	100	1.2E-02	1.6E-02				0.01	1.2E-04	1.6E-04							< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1					< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,6,7,8-HXCDF	5	2	40	1.3E-03	2.0E-03				0.1	1.3E-04	2.0E-04							< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1					< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,7,8,9-HXCDD	5	5	100	1.9E-03	2.3E-03				0.1	1.9E-04	2.3E-04							< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1					< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,7,8-PECDD	5	4	80	6.2E-03	8.8E-03				1	6.2E-03	8.8E-03							0.9	1.3	< 0.1	< 0.1	< 0.1	< 0.1					0.2	0.2	< 0.1	< 0.1
1,2,3,7,8-PECDF	4	2	50	9.5E-04	1.5E-03				0.1	9.5E-05	1.5E-04							< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1					< 0.1	< 0.1	< 0.1	< 0.1
2,3,4,6,7,8-HXCDF	4	3	75	8.0E-04	1.1E-03				0.1	8.0E-05	1.1E-04							< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1					< 0.1	< 0.1	< 0.1	< 0.1
2,3,4,7,8-PECDF	5	4	80	5.6E-03	7.8E-03				1	5.6E-03	7.8E-03							0.8	1.1	< 0.1	< 0.1	< 0.1	< 0.1					0.1	0.2	< 0.1	< 0.1
2,3,7,8-TCDD	5	5	100	3.2E-03	4.4E-03	1	3.2E-03	4.4E-03	1	3.2E-03	4.4E-03	0.5	0.6	< 0.1	< 0.1	< 0.1	< 0.1	0.5	0.6	< 0.1	< 0.1	< 0.1	< 0.1	0.1	0.1	< 0.1	< 0.1	0.1	0.1	< 0.1	< 0.1
2,3,7,8-TCDF	5	5	100	1.6E-02	2.0E-02	0.0064	1.0E-04	1.3E-04	1	1.6E-02	2.0E-02	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	2.3	2.8	0.1	0.1	0.1	0.1	< 0.1	< 0.1	< 0.1	< 0.1	0.4	0.5	< 0.1	0.1
PCB Congener 77	10	6	60	5.2	7.3	0.000018	9.4E-05	1.3E-04	0.05	2.6E-01	3.6E-01	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	37	52	1.4	1.9	0.8	1.2	< 0.1	< 0.1	< 0.1	< 0.1	6.9	9.6	0.7	1.0
PCB Congener 81	10	6	60	NE	1.4	0.0019	NA	2.7E-03	0.1	NA	1.4E-01	NA	0.4	NA	< 0.1	NA	< 0.1	NA	21	NA	0.8	NA	0.5	NA	0.1	NA	0.0	NA	3.8	NA	0.4
PCB Congener 105	10	10	100	109	132	0.0000076	8.3E-04	1.0E-03	0.0001	1.1E-02	1.3E-02	0.1	0.1	< 0.1	< 0.1	< 0.1	< 0.1	1.6	1.9	0.1	0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	0.3	0.3	< 0.1	< 0.1
PCB Congener 118	10	10	100	357	452	3.7E-07	1.3E-04	1.7E-04	0.00001	3.6E-03	4.5E-03	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	0.5	0.6	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	0.1	0.1	< 0.1	< 0.1
PCB Congener 126	10	6	60	NE	2.0	0.022	NA	4.4E-02	0.1	NA	2.0E-01	NA	6.3	NA	0.2	NA	0.1	NA	28	NA	1.0	NA	0.6	NA	1.2	NA	0.1	NA	5.2	NA	0.5
PCB Congener 169	10	5	50	NE	0.2	0.00047	NA	7.7E-05	0.001	NA	1.6E-04	NA	< 0.1	NA	< 0.1	NA	< 0.1	NA	< 0.1	NA	< 0.1	NA	< 0.1	NA	< 0.1	NA	< 0.1	NA	< 0.1	NA	< 0.1
Total TEQ							**4.4E-03**	**5.2E-02**		**3.1E-01**	**7.7E-01**	**0.6**	**7.5**	**< 0.1**	**0.3**	**< 0.1**	**0.2**	**44**	**110**	**1.6**	**4.0**	**1.0**	**2.5**	**0.1**	**1.4**	**< 0.1**	**0.1**	**8.1**	**20**	**0.8**	**2.0**

Notes:

Data is from Kidney Island.

NA - Not applicable.

NE indicates that the calculated mean exceeded the detected maximum and, therefore, the mean was not evaluated.

Table 6-101 PCB Congener and Dioxin/Furan Hazard Quotients for Forster's Tern Eggs in Green Bay Zone 2

Analyte	Number of Samples	Number of Detects	Detection Frequency (%)	Mean	RME	Tillitt *et al.*, 1991b			Van den Berg *et al.*, 1998			Reproduction Hazard Quotients (based on Tillitt *et al.*, 1991b TEFs)						Reproduction Hazard Quotients (based on Van den Berg *et al.*, 1998 TEFs)						Deformity Hazard Quotients (based on Tillitt *et al.*, 1991b TEFs)				Deformity Hazard Quotients (based on Van den Berg *et al.*, 1998 TEFs)			
						TEF	Mean TEC	RME TEC	TEF	Mean TEC	RME TEC	NOAEC		LD_{20}		LD_{30}		NOAEC		LD_{20}		LD_{30}		NOAEC		LOAEC		NOAEC		LOAEC	
												Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME
Dioxins/Furans (µg/kg)																															
1,2,3,4,6,7,8,9-OCDD	7	7	100	5.3E-01	6.4E-01				0.0001	5.3E-05	6.4E-05							< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1					< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,4,6,7,8,9-OCDF	3	1	33	7.6E-04	1.0E-03				0.0001	7.6E-08	1.0E-07							< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1					< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,4,6,7,8-HPCDD	7	7	100	8.7E-03	1.1E-02				0.001	8.7E-06	1.1E-05							< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1					< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,4,6,7,8-HPCDF	7	3	43	4.2E-04	5.2E-04				0.01	4.2E-06	5.2E-06							< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1					< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,4,7,8-HXCDD	5	5	100	9.7E-04	1.9E-03				0.05	4.9E-05	9.6E-05							< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1					< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,4,7,8-HXCDF	3	3	100	5.8E-04	8.1E-04				0.1	5.8E-05	8.1E-05							< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1					< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,6,7,8-HXCDD	7	6	86	6.4E-03	8.9E-03				0.01	6.4E-05	8.9E-05							< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1					< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,6,7,8-HXCDF	7	2	29	9.6E-04	9.4E-04				0.1	9.6E-05	9.4E-05							< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1					< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,7,8,9-HXCDD	5	4	80	1.1E-03	1.5E-03				0.1	1.1E-04	1.5E-04							< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1					< 0.1	< 0.1	< 0.1	< 0.1
1,2,3,7,8-PECDD	7	6	86	2.5E-03	3.3E-03				1	2.5E-03	3.3E-03							0.4	0.5	< 0.1	< 0.1	< 0.1	< 0.1					0.1	0.1	< 0.1	< 0.1
1,2,3,7,8-PECDF	1	1	100	NA	9.3E-04				0.1	NA	9.3E-05							NA	< 0.1	NA	< 0.1	NA	< 0.1					NA	< 0.1	NA	< 0.1
2,3,4,7,8-PECDF	5	2	40	3.9E-04	6.6E-04				1	3.9E-04	6.6E-04							0.1	0.1	< 0.1	< 0.1	< 0.1	< 0.1					< 0.1	< 0.1	< 0.1	< 0.1
2,3,7,8-TCDD	7	7	100	3.3E-03	4.3E-03	1	3.3E-03	4.3E-03	1	3.3E-03	4.3E-03	0.5	0.6	< 0.1	< 0.1	< 0.1	< 0.1	0.5	0.6	< 0.1	< 0.1	< 0.1	< 0.1	0.1	0.1	< 0.1	< 0.1	0.1	0.1	< 0.1	< 0.1
2,3,7,8-TCDF	7	7	100	1.3E-03	1.7E-03	0.0064	8.6E-06	1.1E-05	1	1.3E-03	1.7E-03	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	0.2	0.2	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 77	9	5	56	2.6	3.3	0.000018	4.6E-05	5.9E-05	0.05	1.3E-01	1.7E-01	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	18	24	0.7	0.9	0.4	0.5	< 0.1	< 0.1	< 0.1	< 0.1	3.4	4.3	0.3	0.4
PCB Congener 81	9	5	56	NE	1.3	0.0019	NA	2.4E-03	0.1	NA	1.3E-01	NA	0.3	NA	< 0.1	NA	< 0.1	NA	18	NA	0.7	NA	0.4	NA	0.1	NA	0.0	NA	3.3	NA	0.3
PCB Congener 105	10	10	100	93.4	113	0.0000076	7.1E-04	8.6E-04	0.0001	9.3E-03	1.1E-02	0.1	0.1	< 0.1	< 0.1	< 0.1	< 0.1	1.3	1.6	< 0.1	0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	0.2	0.3	< 0.1	< 0.1
PCB Congener 118	10	10	100	283	348	0.00000037	1.0E-04	1.3E-04	0.00001	2.8E-03	3.5E-03	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	0.4	0.5	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	0.1	0.1	< 0.1	< 0.1
PCB Congener 126	9	5	56	NE	0.7	0.022	NA	1.5E-02	0.1	NA	6.8E-02	NA	2.1	NA	0.1	NA	< 0.1	NA	9.7	NA	0.4	NA	0.2	NA	0.4	NA	< 0.1	NA	1.8	NA	0.2
PCB Congener 169	10	2	20	NE	0.8	0.00047	NA	3.7E-04	0.001	NA	8.0E-04	NA	0.1	NA	< 0.1	NA	< 0.1	NA	0.1	NA	< 0.1	NA	< 0.1	NA	< 0.1	NA	< 0.1	NA	< 0.1	NA	< 0.1
Total TEQ							**4.1E-03**	**2.3E-02**		**1.5E-01**	**3.9E-01**	**0.6**	**3.3**	**< 0.1**	**0.1**	**< 0.1**	**0.1**	**21**	**55**	**0.8**	**2.0**	**0.5**	**1.3**	**0.1**	**0.6**	**< 0.1**	**0.1**	**3.9**	**10**	**0.4**	**1.0**

Notes:

Data is from Kidney Island.

NA - Not applicable.

NE indicates that the calculated mean exceeded the detected maximum and, therefore, the mean was not evaluated.

Table 6-102 Estimated Hazard Quotients for Mink in Green Bay Zone 2

Analyte	Total Estimated Exposure (µg/kg-BW/day) Based on Carp Consumption	RME
Mercury	14.0	21.4
Total PCBs (N)	1,271	1,413
Total PCBs (I_d)	1,275	1,415
Dieldrin	4.0	5.6
p,p'-DDE	37.6	134
Analyte	**NOAEC HQs** **Mean**	**RME**
Mercury	0.2	0.3
Total PCBs	318	353
Total PCBs (I_d)	319	354
Dieldrin	0.4	0.6
p,p'-DDE	< 0.1	< 0.1
Analyte	**LOAEC HQs** **Mean**	**RME**
Mercury	0.1	0.1
Total PCBs	9.8	11
Total PCBs (I_d)	9.8	11
Dieldrin	0.2	0.3
p,p'-DDE	< 0.1	< 0.1

Table 6-103 Hazard Quotients for Whole Fish in Green Bay Zone 3A

Analyte	Species	Number of Samples	Number of Detects	Detection Frequency (%)	Mean	RME	Criteria		Hazard Quotients			
							NOAEC	LOAEC	NOAEC		LOAEC	
									Mean	RME	Mean	RME
Metals (mg/kg)												
Mercury	Gizzard Shad	1	0									
	Rainbow Smelt	6	4	67	0.03	0.04	0.25	2.37	0.1	0.2	< 0.1	< 0.1
PCBs (µg/kg)												
Total PCBs	Alewife	18	18	100	907	1,271	760	7,600	1.2	1.7	0.1	0.2
	Gizzard Shad	1	1	100	NA	3,524	760	7,600	NA	4.6	NA	0.5
	Rainbow Smelt	32	31	97	570	735	760	7,600	0.7	1.0	0.1	0.1
	Walleye	14	14	100	4,155	5,064	760	7,600	5.5	6.7	0.5	0.7
	Brown Trout	14	14	100	3,250	3,612	760	7,600	4.3	4.8	0.4	0.5
Pesticides (µg/kg)												
Dieldrin	Alewife	18	18	100	21.5	27.5	370	3,700	0.1	0.1	< 0.1	< 0.1
	Gizzard Shad	1	0									
	Rainbow Smelt	32	23	72	14.4	17.5	370	3,700	< 0.1	< 0.1	< 0.1	< 0.1
	Walleye	10	10	100	43.4	57.7	370	3,700	0.1	0.2	< 0.1	< 0.1
	Brown Trout	14	14	100	76.0	100	370	3,700	0.2	0.3	< 0.1	< 0.1
o,p'-DDD	Rainbow Smelt	12	0									
o,p'-DDE	Rainbow Smelt	11	0									
o,p'-DDT	Rainbow Smelt	12	0									
p,p'-DDD	Gizzard Shad	1	0									
	Rainbow Smelt	12	0									
p,p'-DDE	Gizzard Shad	1	1	100	NA	150	300	2,950	NA	0.5	NA	0.1
	Rainbow Smelt	12	2	17	30.0	36.2	300	2,950	0.1	0.1	< 0.1	< 0.1
p,p'-DDT	Gizzard Shad	1	0									
	Rainbow Smelt	12	0									

Note:
NA - Not applicable.

Table 6-104 PCB Congener Hazard Quotients for Whole Fish in Green Bay Zone 3A

Analyte	Species	Number of Samples	Number of Detects	Detection Frequency (%)	Mean	RME	TEF	Mean TEC	RME TEC	Hazard Quotients			
										NOAEC		LOAEC	
										Mean	RME	Mean	RME
PCBs (µg/kg)													
PCB Congener 77/110	Alewife	18	18	100	39.5	53.1	0.0001	4.0E-03	5.3E-03	0.1	0.1	< 0.1	0.1
PCB Congener 81	Alewife	18	18	100	3.5	4.2	0.0005	1.7E-03	2.1E-03	< 0.1	0.1	< 0.1	< 0.1
PCB Congener 132/153/105	Alewife	18	18	100	45.7	56.5	0.000005	2.3E-04	2.8E-04	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 118	Alewife	18	18	100	29.5	36.5	0.000005	1.5E-04	1.8E-04	< 0.1	< 0.1	< 0.1	< 0.1
Total TEQ	**Alewife**							**6.1E-03**	**7.9E-03**	**0.1**	**0.2**	**0.1**	**0.1**
PCB Congener 77	Gizzard Shad	1	1	100	NA	1.9	0.0001	NA	1.9E-04	NA	< 0.1	NA	< 0.1
PCB Congener 81	Gizzard Shad	1	0										
PCB Congener 105	Gizzard Shad	1	1	100	NA	54.0	0.000005	NA	2.7E-04	NA	< 0.1	NA	< 0.1
PCB Congener 118	Gizzard Shad	1	1	100	NA	123	0.000005	NA	6.2E-04	NA	< 0.1	NA	< 0.1
PCB Congener 126	Gizzard Shad	1	1	100	NA	0.7	0.005	NA	3.5E-03	NA	< 0.1	NA	< 0.1
PCB Congener 169	Gizzard Shad	1	0										
Total TEQ	**Gizzard Shad**							**0.0E+00**	**4.6E-03**	NA	**< 0.1**	NA	**< 0.1**
PCB Congener 77/110	Rainbow Smelt	20	20	100	22.7	30.2	0.0001	2.3E-03	3.0E-03	0.1	0.1	< 0.1	< 0.1
PCB Congener 81	Rainbow Smelt	20	20	100	2.6	3.1	0.0005	1.3E-03	1.6E-03	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 132/153/105	Rainbow Smelt	20	20	100	26.3	31.0	0.000005	1.3E-04	1.5E-04	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 118	Rainbow Smelt	20	20	100	17.7	23.0	0.000005	8.8E-05	1.1E-04	< 0.1	< 0.1	< 0.1	< 0.1
Total TEQ	**Rainbow Smelt**							**3.8E-03**	**4.8E-03**	**0.1**	**0.1**	**< 0.1**	**0.1**
PCB Congener 77	Walleye	1	1	100	NA	8.7	0.0001	NA	8.7E-04	NA	< 0.1	NA	< 0.1
PCB Congener 81	Walleye	11	10	91	11.4	15.5	0.0005	5.7E-03	7.7E-03	0.1	0.2	0.1	0.1
PCB Congener 105	Walleye	4	4	100	63.3	71.7	0.000005	3.2E-04	3.6E-04	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 118	Walleye	14	14	100	125	150	0.000005	6.3E-04	7.5E-04	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 126	Walleye	1	1	100	NA	0.9	0.005	NA	4.7E-03	NA	0.1	NA	0.1
PCB Congener 169	Walleye	4	4	100	1.6	2.4	0.00005	8.1E-05	1.2E-04	< 0.1	< 0.1	< 0.1	< 0.1
Total TEQ	**Walleye**							**6.7E-03**	**1.5E-02**	**0.3**	**0.4**	**0.1**	**0.2**
PCB Congener 77/110	Brown Trout	14	14	100	134	153	0.0001	1.3E-02	1.5E-02	0.3	0.4	0.2	0.2
PCB Congener 81	Brown Trout	14	14	100	12.3	14.4	0.0005	6.1E-03	7.2E-03	0.1	0.2	0.1	0.1
PCB Congener 132/153/105	Brown Trout	14	14	100	170	199	0.000005	8.5E-04	1.0E-03	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 118	Brown Trout	14	14	100	111	127	0.000005	5.5E-04	6.4E-04	< 0.1	< 0.1	< 0.1	< 0.1
Total TEQ	**Brown Trout**							**2.1E-02**	**2.4E-02**	**0.5**	**0.6**	**0.2**	**0.3**

Notes:

NA - Not applicable.

TEQ Criteria: NOAEC = 0.041 µg/kg; LOAEC = 0.084 µg/kg.

Table 6-105 Hazard Quotients for Bird Tissue in Green Bay Zone 3A

Analyte	Species	Tissue	Number of Samples	Number of Detects	Detection Frequency (%)	Mean	RME	Reproduction Criteria		Hazard Quotients NOAEC		Hazard Quotients LOAEC		Deformity Criteria		Hazard Quotients NOAEC		Hazard Quotients LOAEC	
								NOAEC	LOAEC	Mean	RME	Mean	RME	NOAEC	LOAEC	Mean	RME	Mean	RME
Metals (mg/kg)																			
Mercury	Bald Eagle	egg	3	3	100	0.3	0.3	0.5	5	0.5	0.6	0.1	0.1						
PCBs (µg/kg)																			
Total PCBs	Bald Eagle	egg	1	1	100	NA	13,000	4,700	7,600	NA	2.8	NA	1.7	800	8,000	NA	16	NA	1.6
Pesticides (µg/kg)																			
Dieldrin	Bald Eagle	egg	1	1	100	NA	200	100	1,000	NA	2.0	NA	0.2						
p,p'-DDD	Bald Eagle	egg	1	1	100	NA	120	3,000	5,100	NA	< 0.1	NA	< 0.1						
p,p'-DDE	Bald Eagle	egg	1	1	100	NA	2,400	3,000	5,100	NA	0.8	NA	0.5						
p,p'-DDT	Bald Eagle	egg	1	0															

Note:
NA - Not applicable.

Table 6-106 Estimated Hazard Quotients for Piscivorous Birds in Green Bay Zone 3A

Analyte	Total Estimated Exposure (µg/kg-BW/day)							
	Common Tern		Forster's Tern		Double-crested Cormorant		Bald Eagle	
	Mean	RME	Mean	RME	Mean	RME	Mean	RME
Mercury	14.7	19.6	13.6	18.1	5.7	7.6	2.3	4.5
Total PCBs	444	623	410	575	172	241	334	475
Dieldrin	10.5	13.5	9.7	12.4	4.1	5.2	2.6	6.3
p,p'-DDE	14.7	17.7	13.6	16.3	5.7	6.8	2.3	2.3

Analyte	NOAEC HQs							
	Common Tern		Forster's Tern		Double-crested Cormorant		Bald Eagle	
	Mean	RME	Mean	RME	Mean	RME	Mean	RME
Mercury	1.8	2.5	1.7	2.3	0.7	0.9	0.3	0.6
Total PCBs	4.0	5.6	3.7	5.1	1.5	2.1	3.0	4.2
Dieldrin	0.1	0.1	0.1	0.1	< 0.1	< 0.1	< 0.1	0.1
p,p'-DDE	0.8	1.0	0.8	0.9	0.3	0.4	0.1	0.1

Analyte	LOAEC HQs							
	Common Tern		Forster's Tern		Double-crested Cormorant		Bald Eagle	
	Mean	RME	Mean	RME	Mean	RME	Mean	RME
Mercury	0.2	0.3	0.2	0.2	0.1	0.1	< 0.1	0.1
Total PCBs	0.4	0.6	0.4	0.5	0.2	0.2	0.3	0.4
Dieldrin	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1
p,p'-DDE	0.1	0.1	0.1	0.1	< 0.1	< 0.1	< 0.1	< 0.1

Table 6-107 Estimated Hazard Quotients for Mink in Green Bay Zone 3A

Analyte	Total Estimated Exposure (µg/kg-BW/day) Mean	RME
Mercury	4.9	9.6
Total PCBs (N)	507	763
Total PCBs (I_d)	507	761
Dieldrin	3.4	10.5
pip'-DDE	4.8	4.8
Analyte	**NOAEC HQs Mean**	**RME**
Mercury	0.1	0.1
Total PCBs	127	191
Total PCBs (I_d)	127	190
Dieldrin	0.4	1.2
p,p'-DDE	< 0.1	< 0.1
Analyte	**LOAEC HQs Mean**	**RME**
Mercury	< 0.1	< 0.1
Total PCBs	3.9	5.9
Total PCBs (I_d)	3.9	5.9
Dieldrin	0.2	0.6
p,p'-DDE	< 0.1	< 0.1

Table 6-108 Hazard Quotients for Whole Fish in Green Bay Zone 3B

Analyte	Species	Number of Samples	Number of Detects	Detection Frequency (%)	Mean	RME	Criteria		Hazard Quotients			
							NOAEC	LOAEC	NOAEC		LOAEC	
									Mean	RME	Mean	RME
Metals (mg/kg)												
Mercury	Alewife	1	0									
	Gizzard Shad	1	0									
	Walleye	3	1	33	0.3	0.7	0.25	2.37	1.0	2.6	0.1	0.3
PCBs (µg/kg)												
Total PCBs	Alewife	8	8	100	1,821	2,375	760	7,600	2.4	3.1	0.2	0.3
	Gizzard Shad	1	1	100	NA	635	760	7,600	NA	0.8	NA	0.1
	Rainbow Smelt	20	20	100	733	861	760	7,600	1.0	1.1	0.1	0.1
	Walleye	26	26	100	6,429	11,741	760	7,600	8.5	15	0.8	1.5
	Brown Trout	26	26	100	2,223	2,697	760	7,600	2.9	3.5	0.3	0.4
Pesticides (µg/kg)												
Dieldrin	Alewife	8	7	88	19.1	27.3	370	3,700	0.1	0.1	< 0.1	< 0.1
	Gizzard Shad	1	0									
	Rainbow Smelt	20	20	100	14.7	18.4	370	3,700	< 0.1	< 0.1	< 0.1	< 0.1
	Walleye	15	12	80	50.1	63.3	370	3,700	0.1	0.2	< 0.1	< 0.1
	Brown Trout	12	12	100	72.0	83.1	370	3,700	0.2	0.2	< 0.1	< 0.1
p,p'-DDD	Alewife	1	0									
	Gizzard Shad	1	0									
	Walleye	3	0									
p,p'-DDE	Alewife	1	1	100	NA	80.0	300	2,950	NA	0.3	NA	< 0.1
	Gizzard Shad	1	1	100	NA	37.0	300	2,950	NA	0.1	NA	< 0.1
	Walleye	3	2	67	207	540	300	2,950	0.7	1.8	0.1	0.2
p,p'-DDT	Alewife	1	0									
	Gizzard Shad	1	0									
	Walleye	3	0									

Note:
NA - Not applicable.

Table 6-109 PCB Congener Hazard Quotients for Whole Fish in Green Bay Zone 3B

Analyte	Species	Number of Samples	Number of Detects	Detection Frequency (%)	Mean	RME	TEF	Mean TEC	RME TEC	Hazard Quotients			
										NOAEC		LOAEC	
										Mean	RME	Mean	RME
PCBs (µg/kg)													
PCB Congener 77	Alewife	1	1	100	NA	0.1	0.0001	NA	5.4E-06	NA	< 0.1	NA	< 0.1
PCB Congener 81	Alewife	8	7	88	5.7	7.9	0.0005	2.8E-03	4.0E-03	0.1	0.1	< 0.1	< 0.1
PCB Congener 105	Alewife	1	1	100	NA	13.0	0.000005	NA	6.5E-05	NA	< 0.1	NA	< 0.1
PCB Congener 118	Alewife	8	8	100	52.0	62.5	0.000005	2.6E-04	3.1E-04	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 126	Alewife	1	0										
PCB Congener 169	Alewife	1	0										
Total TEQ	**Alewife**							**3.1E-03**	**4.3E-03**	**NA**	**0.1**	**NA**	**0.1**
PCB Congener 77	Gizzard Shad	1	1	100	NA	0.2	0.0001	NA	2.3E-05	NA	< 0.1	NA	< 0.1
PCB Congener 81	Gizzard Shad	1	0										
PCB Congener 105	Gizzard Shad	1	1	100	NA	12.0	0.000005	NA	6.0E-05	NA	< 0.1	NA	< 0.1
PCB Congener 118	Gizzard Shad	1	1	100	NA	32.0	0.000005	NA	1.6E-04	NA	< 0.1	NA	< 0.1
PCB Congener 126	Gizzard Shad	1	0										
PCB Congener 169	Gizzard Shad	1	0										
Total TEQ	**Gizzard Shad**							**0.0E+00**	**2.4E-04**	**NA**	**< 0.1**	**NA**	**< 0.1**
PCB Congener 77/110	Rainbow Smelt	20	20	100	29.8	35.5	0.0001	3.0E-03	3.6E-03	0.1	0.1	< 0.1	< 0.1
PCB Congener 81	Rainbow Smelt	20	19	95	2.8	3.3	0.0005	1.4E-03	1.6E-03	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 132/153/105	Rainbow Smelt	20	20	100	31.2	36.5	0.000005	1.6E-04	1.8E-04	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 118	Rainbow Smelt	20	20	100	22.3	26.2	0.000005	1.1E-04	1.3E-04	< 0.1	< 0.1	< 0.1	< 0.1
Total TEQ	**Rainbow Smelt**							**4.6E-03**	**5.5E-03**	**0.1**	**0.1**	**0.1**	**0.1**
PCB Congener 77	Walleye	4	4	100	2.5	4.9	0.0001	2.5E-04	4.9E-04	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 81	Walleye	16	13	81	11.0	14.7	0.0005	5.5E-03	7.3E-03	0.1	0.2	0.1	0.1
PCB Congener 105	Walleye	13	13	100	103	134	0.000005	5.1E-04	6.7E-04	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 118	Walleye	25	25	100	227	370	0.000005	1.1E-03	1.8E-03	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 126	Walleye	4	2	50	0.2	0.6	0.005	1.2E-03	2.9E-03	< 0.1	0.1	< 0.1	< 0.1
PCB Congener 169	Walleye	12	9	75	3.5	7.1	0.00005	1.7E-04	3.5E-04	< 0.1	< 0.1	< 0.1	< 0.1
Total TEQ	**Walleye**							**8.8E-03**	**1.4E-02**	**0.2**	**0.3**	**0.1**	**0.2**
PCB Congener 77	Brown Trout	1	1	100	NA	3.5	0.0001	NA	3.5E-04	NA	< 0.1	NA	< 0.1
PCB Congener 81	Brown Trout	13	13	100	11.2	13.5	0.0005	5.6E-03	6.8E-03	0.1	0.2	0.1	0.1
PCB Congener 105	Brown Trout	5	5	100	38.9	44.9	0.000005	1.9E-04	2.2E-04	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 118	Brown Trout	17	17	100	107	117	0.000005	5.3E-04	5.8E-04	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 126	Brown Trout	1	1	100	NA	0.5	0.005	NA	2.7E-03	NA	0.1	NA	< 0.1
PCB Congener 169	Brown Trout	3	3	100	0.9	1.3	0.00005	4.4E-05	6.7E-05	< 0.1	< 0.1	< 0.1	< 0.1
Total TEQ	**Brown Trout**							**6.4E-03**	**1.1E-02**	**0.2**	**0.3**	**0.1**	**0.1**

Note:

NA - Not applicable.

TEQ Criteria: NOAEC = 0.041 µg/kg; LOAEC = 0.084 µg/kg.

Table 6-110 Hazard Quotients for Bird Tissue in Green Bay Zone 3B

Analyte	Species	Tissue	Number of Samples	Number of Detects	Detection Frequency (%)	Mean	RME	Reproduction Criteria		Hazard Quotients NOAEC		Hazard Quotients LOAEC		Deformity Criteria		Hazard Quotients NOAEC		Hazard Quotients LOAEC	
								NOAEC	LOAEC	Mean	RME	Mean	RME	NOAEC	LOAEC	Mean	RME	Mean	RME
PCBs (µg/kg)																			
Total PCBs	Double-crested Cormorant	whole	21	20	95	5,384	15,000	4,700	7,600	1.1	3.2	0.7	2.0	800	8,000	6.7	19	0.7	1.9
Pesticides (µg/kg)																			
Dieldrin	Double-crested Cormorant	whole	20	19	95	128	239	100	1,000	1.3	2.4	0.1	0.2						
o,p'-DDD	Double-crested Cormorant	whole	20	0															
o,p'-DDE	Double-crested Cormorant	whole	20	0															
o,p'-DDT	Double-crested Cormorant	whole	20	0															
p,p'-DDD	Double-crested Cormorant	whole	20	3	15	6.3	7.6	3,000	5,100	< 0.1	< 0.1	< 0.1	< 0.1						
p,p'-DDE	Double-crested Cormorant	whole	20	20	100	2,010	4,546	3,000	5,100	0.7	1.5	0.4	0.9						
p,p'-DDT	Double-crested Cormorant	whole	20	11	55	10.9	14.8	3,000	5,100	< 0.1	< 0.1	< 0.1	< 0.1						

Table 6-111 Estimated Hazard Quotients for Piscivorous Birds in Green Bay Zone 3B

Analyte	Total Estimated Exposure (µg/kg-BW/day)							
	Common Tern		Forster's Tern		Double-crested Cormorant		Bald Eagle	
	Mean	RME	Mean	RME	Mean	RME	Mean	RME
Mercury	12.3	24.5	11.3	22.6	4.7	9.5	15.6	30.1
Total PCBs	892	1,164	823	1,073	345	450	594	823
Dieldrin	9.3	13.4	8.6	12.3	3.6	5.2	5.1	6.4
p,p'-DDE	39.2	39.2	36.2	36.2	15.1	15.1	16.1	34.0

Analyte	NOAEC HQs							
	Common Tern		Forster's Tern		Double-crested Cormorant		Bald Eagle	
	Mean	RME	Mean	RME	Mean	RME	Mean	RME
Mercury	1.5	3.1	1.4	2.8	0.6	1.2	2.0	3.8
Total PCBs	8.0	10	7.3	9.6	3.1	4.0	5.3	7.3
Dieldrin	< 0.1	< 0.1	0.1	0.1	< 0.1	< 0.1	< 0.1	0.1
p,p'-DDE	2.2	2.2	2.0	2.0	0.8	0.8	0.9	1.9

Analyte	LOAEC HQs							
	Common Tern		Forster's Tern		Double-crested Cormorant		Bald Eagle	
	Mean	RME	Mean	RME	Mean	RME	Mean	RME
Mercury	0.2	0.3	0.1	0.3	0.1	0.1	0.2	0.4
Total PCBs	0.8	1.0	0.7	1.0	0.3	0.4	0.5	0.7
Dieldrin	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1
p,p'-DDE	0.2	0.2	0.2	0.2	0.1	0.1	0.1	0.2

Table 6-112 PCB Congener Hazard Quotients for Double-crested Cormorants in Green Bay Zone 3B

Analyte	Number of Samples	Number of Detects	Detection Frequency (%)	Mean	RME	Tillitt *et al.*, 1991b			Van den Berg *et al.*, 1998			Reproduction Hazard Quotients (based on Tillitt *et al.*, 1991b TEFs)						Reproduction Hazard Quotients (based on Van den Berg *et al.*, 1998 TEFs)						Deformity Hazard Quotients (based on Tillitt *et al.*, 1991b TEFs)				Deformity Hazard Quotients (based on Van den Berg *et al.*, 1998 TEFs)			
						TEF	Mean TEC	RME TEC	TEF	Mean TEC	RME TEC	NOAEC		LD_{20}		LD_{30}		NOAEC		LD_{20}		LD_{30}		NOAEC		LOAEC		NOAEC		LOAEC	
												Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME	Mean	RME
PCBs (µg/kg)																															
PCB Congener 77	16	6	38	0.2	0.4	0.000018	3.5E-06	7.6E-06	0.05	9.7E-03	2.1E-02	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	1.4	3.0	0.1	0.1	< 0.1	0.1	< 0.1	< 0.1	< 0.1	< 0.1	0.3	0.6	< 0.1	0.1
PCB Congener 105	16	16	100	92.2	122	0.0000076	7.0E-04	9.3E-04	0.0001	9.2E-03	1.2E-02	0.1	0.1	< 0.1	< 0.1	< 0.1	< 0.1	1.3	1.7	< 0.1	0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	0.2	0.3	< 0.1	< 0.1
PCB Congener 118/106	16	16	100	215	650	0.00000037	7.9E-05	2.4E-04	0.00001	2.1E-03	6.5E-03	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	0.3	0.9	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	0.1	0.2	< 0.1	< 0.1
PCB Congener 126	16	13	81	0.6	0.8	0.022	1.4E-02	1.8E-02	0.1	6.2E-02	8.3E-02	2.0	2.6	0.1	0.1	< 0.1	0.1	8.9	12	0.3	0.4	0.2	0.3	0.4	0.5	< 0.1	< 0.1	1.6	2.2	0.2	0.2
PCB Congener 169	16	5	31	0.1	0.1	0.00047	3.4E-05	4.2E-05	0.001	7.2E-05	9.0E-05	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1
Total TEQ							**1.5E-02**	**2.0E-02**		**0.1**	**0.1**	**2.1**	**2.8**	**0.1**	**0.1**	**< 0.1**	**0.1**	**12**	**18**	**0.4**	**0.6**	**0.3**	**0.4**	**0.4**	**0.5**	**< 0.1**	**0.1**	**2.2**	**3.2**	**0.2**	**0.3**

Note:

Data is from Kidney Island.

Table 6-113 Estimated Hazard Quotients for Mink in Green Bay Zone 3B

Analyte	Total Estimated Exposure (µg/kg-BW/day) Mean	RME
Mercury	21.5	33.5
Total PCBs (N)	949	1,180
Total PCBs (I_d)	949	1,178
Dieldrin	8.3	10.5
p,p'-DDE	24.1	45.9
Analyte	**NOAEC HQs Mean**	**RME**
Mercury	0.3	0.4
Total PCBs	237	295
Total PCBs (I_d)	237	295
Dieldrin	0.9	1.2
p,p'-DDE	< 0.1	< 0.1
Analyte	**LOAEC HQs Mean**	**RME**
Mercury	0.1	0.2
Total PCBs	7.3	9.1
Total PCBs (I_d)	7.3	9.1
Dieldrin	0.5	0.6
p,p'-DDE	< 0.1	< 0.1

Table 6-114 Hazard Quotients for Whole Fish in Green Bay Zone 4

Analyte	Species	Number of Samples	Number of Detects	Detection Frequency (%)	Mean	RME	Criteria		Hazard Quotients			
							NOAEC	LOAEC	NOAEC		LOAEC	
									Mean	RME	Mean	RME
Metals (mg/kg)												
Mercury	Walleye	20	20	100	0.2	0.2	0.25	2.37	0.8	0.9	0.1	0.1
PCBs (µg/kg)												
Total PCBs	Alewife	8	8	100	1,036	1,488	760	7,600	1.4	2.0	0.1	0.2
	Rainbow Smelt	18	18	100	526	764	760	7,600	0.7	1.0	0.1	0.1
	Walleye	36	36	100	2,546	3,294	760	7,600	3.4	4.3	0.3	0.4
	Brown Trout	18	18	100	2,451	2,714	760	7,600	3.2	3.6	0.3	0.4
Pesticides (µg/kg)												
Dieldrin	Alewife	8	8	100	20.8	26.1	370	3,700	0.1	0.1	< 0.1	< 0.1
	Rainbow Smelt	18	18	100	18.1	21.9	370	3,700	0.0	0.1	< 0.1	< 0.1
	Walleye	33	33	100	46.9	62.0	370	3,700	0.1	0.2	< 0.1	< 0.1
	Brown Trout	13	13	100	88.2	95.7	370	3,700	0.2	0.3	< 0.1	< 0.1
p,p'-DDD	Walleye	20	20	100	28.7	32.2	300	2,950	0.1	0.1	< 0.1	< 0.1
p,p'-DDE	Walleye	20	20	100	479	593	300	2,950	1.6	2.0	0.2	0.2
p,p'-DDT	Walleye	20	20	100	33.9	42.6	300	2,950	0.1	0.1	0.01	0.01

Table 6-115 PCB Congener Hazard Quotients for Whole Fish in Green Bay Zone 4

Analyte	Species	Number of Samples	Number of Detects	Detection Frequency (%)	Mean	RME	TEF	Mean TEC	RME TEC	Hazard Quotients			
										NOAEC		LOAEC	
										Mean	RME	Mean	RME
PCBs (µg/kg)													
PCB Congener 77/110	Alewife	8	8	100	40.2	54.6	0.0001	4.0E-03	5.5E-03	0.1	0.1	< 0.1	0.1
PCB Congener 81	Alewife	8	8	100	3.7	4.9	0.0005	1.8E-03	2.5E-03	< 0.1	0.1	< 0.1	< 0.1
PCB Congener 132/153/105	Alewife	8	8	100	51.7	69.5	0.000005	2.6E-04	3.5E-04	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 118	Alewife	8	8	100	29.9	40.6	0.000005	1.5E-04	2.0E-04	< 0.1	< 0.1	< 0.1	< 0.1
Total TEQ	**Alewife**							**6.3E-03**	**8.5E-03**	**0.2**	**0.2**	**0.1**	**0.1**
PCB Congener 77/110	Rainbow Smelt	18	17	94	20.7	27.6	0.0001	2.1E-03	2.8E-03	0.1	0.1	< 0.1	< 0.1
PCB Congener 81	Rainbow Smelt	18	18	100	2.8	3.8	0.0005	1.4E-03	1.9E-03	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 132/153/105	Rainbow Smelt	18	18	100	29.2	40.8	0.000005	1.5E-04	2.0E-04	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 118	Rainbow Smelt	18	18	100	17.9	25.7	0.000005	9.0E-05	1.3E-04	< 0.1	< 0.1	< 0.1	< 0.1
Total TEQ	**Rainbow Smelt**							**3.7E-03**	**5.0E-03**	**0.1**	**0.1**	**< 0.1**	**0.1**
PCB Congener 77	Walleye	1	1	100	NA	2.1	0.0001	NA	2.1E-04	NA	< 0.1	NA	< 0.1
PCB Congener 81	Walleye	14	13	93	10.4	13.6	0.0005	5.2E-03	6.8E-03	0.1	0.2	0.1	0.1
PCB Congener 105	Walleye	3	3	100	84.7	111	0.000005	4.2E-04	5.6E-04	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 118	Walleye	16	16	100	137	166	0.000005	6.8E-04	8.3E-04	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 126	Walleye	1	1	100	NA	0.3	0.005	NA	1.3E-03	NA	< 0.1	NA	< 0.1
PCB Congener 169	Walleye	2	2	100	5.5	7.8	0.00005	2.7E-04	3.9E-04	< 0.1	< 0.1	< 0.1	< 0.1
Total TEQ	**Walleye**							**6.6E-03**	**1.0E-02**	**0.2**	**0.2**	**0.1**	**0.1**
PCB Congener 77	Brown Trout	1	1	100	NA	1.6	0.0001	NA	1.6E-04	NA	< 0.1	NA	< 0.1
PCB Congener 81	Brown Trout	14	14	100	9.2	10.9	0.0005	4.6E-03	5.5E-03	0.1	0.1	0.1	0.1
PCB Congener 105	Brown Trout	5	5	100	36.1	42.9	0.000005	1.8E-04	2.1E-04	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 118	Brown Trout	18	18	100	92.1	101	0.000005	4.6E-04	5.1E-04	< 0.1	< 0.1	< 0.1	< 0.1
PCB Congener 126	Brown Trout	1	1	100	NA	0.3	0.005	NA	1.6E-03	NA	< 0.1	NA	< 0.1
Total TEQ	**Brown Trout**							**5.2E-03**	**7.9E-03**	**0.1**	**0.2**	**0.1**	**0.1**

Notes:

NA - Not applicable.

TEQ Criteria: NOAEC = 0.041 µg/kg; LOAEC = 0.084 µg/kg.

Table 6-116 Estimated Hazard Quotients for Piscivorous Birds in Green Bay Zone 4

Analyte	Total Estimated Exposure (µg/kg-BW/day)							
	Common Tern		Forster's Tern		Double-crested Cormorant		Bald Eagle	
	Mean	RME	Mean	RME	Mean	RME	Mean	RME
Mercury	14.7	14.7	13.6	13.6	5.7	5.7	20.2	23.3
Total PCBs	508	729	468	672	196	282	329	489
Dieldrin	10.2	12.8	9.4	11.8	3.9	4.9	3.6	4.7
p,p'-DDE	7.3	7.6	6.7	7.0	2.8	3.0	91.2	119

Analyte	NOAEC HQs							
	Common Tern		Forster's Tern		Double-crested Cormorant		Bald Eagle	
	Mean	RME	Mean	RME	Mean	RME	Mean	RME
Mercury	1.8	1.8	1.7	1.7	0.7	0.7	2.5	2.9
Total PCBs	4.5	6.5	4.2	6.0	1.8	2.5	2.9	4.4
Dieldrin	0.1	0.1	0.1	0.1	< 0.1	< 0.1	< 0.1	< 0.1
p,p'-DDE	0.4	0.4	0.4	0.4	0.2	0.2	5.1	6.6

Analyte	LOAEC HQs							
	Common Tern		Forster's Tern		Double-crested Cormorant		Bald Eagle	
	Mean	RME	Mean	RME	Mean	RME	Mean	RME
Mercury	0.2	0.2	0.2	0.2	0.1	0.1	0.3	0.3
Total PCBs	0.5	0.7	0.4	0.6	0.2	0.3	0.3	0.4
Dieldrin	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1
p,p'-DDE	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1

Table 6-117 Estimated Hazard Quotients for Mink in Green Bay Zone 4

Analyte	Total Estimated Exposure (µg/kg-BW/day) Mean	RME
Mercury	32.8	38.8
Total PCBs (N)	573	875
Total PCBs (I_d)	573	875
Dieldrin	5.3	6.9
p,p'-DDE	169	222
Analyte	**NOAEC HQs Mean**	**RME**
Mercury	0.4	0.5
Total PCBs	143	219
Total PCBs (I_d)	143	219
Dieldrin	0.6	0.8
p,p'-DDE	< 0.1	< 0.1
Analyte	**LOAEC HQs Mean**	**RME**
Mercury	0.2	0.2
Total PCBs	4.4	6.7
Total PCBs (I_d)	4.4	6.7
Dieldrin	0.3	0.4
p,p'-DDE	< 0.1	< 0.1

Table 6-118 Bald Eagle Productivity in the Lower Fox River and Green Bay (from Dykstra and Meyer, 1996)

Location	State	Nest Number	Nest Name	Productivity (young per occupied territory)								
				1987	1988	1989	1990	1991	1992	1993	1994	1995
Little Lake Butte des Morts												
	WI	WI-1	Mud Creek								2	3
Appleton/Little Rapids												
	WI	OU-1a	Kaukauna		2	1	0	3	3	3	1	3
Zone 1												
	WI	BR-2	East River								0	
Zone 2												
	WI	BR-01/OC-08	Little Tail Point							0	0	1
Zone 3A												
	WI	MT-07	Peshtigo River	1	0	0	0	0	0	0		0
	WI	MT-16	Peshtigo River, North						1	0	0	
	WI	MT-17	Blueberry Island							0	2	2
	WI	OC-4	Oconto River		0	0	1	0	0	0	1	0
Zone 4												
	MI	De-09/De-15	Moss Lake/Boutlier Lake	1	0	0	1	0	0	0	1	1
	MI	De-13	Granskog Lake	1	0	0	0	0	0	0	0	1
	MI	De-16	No-se-um Creek/North Lake	2	2		1	0	1	1	2	1
	MI	De-17	Fish Dam River	1	0	0	0	0	0	0	0	2
	MI	De-18/De-07	Squaw Point/Squaw Creek		0		0		0	0	0	0
	MI	De-20	St. Vital's Point							1	0	1
	MI	Mm-03	Deer Creek					2	2	0		0

Table 6-119 Summary of Bald Eagle Productivity in the Lower Fox River and Green Bay (from Dykstra and Meyer, 1996)

Location	Productivity Measurement	Year 1987	1988	1989	1990	1991	1992	1993	1994	1995	Total
Lower Fox River											
	Number of Occupied Territories		1	1	1	1	1	1	3	2	**11**
	Number of Young		2	1	0	3	3	3	3	6	**21**
	Number of Young per Occupied Territory		2	1	0	3	3	3	1	3	
	Average Number of Young per Occupied Territory										**1.9**
Green Bay											
	Number of Occupied Territories	5	7	5	7	7	9	12	11	12	**75**
	Number of Young	6	2	0	3	2	4	2	7	10	**36**
	Number of Young per Occupied Territory	1.2	0.29	0	0.43	0.29	0.44	0.17	0.64	0.83	
	Average Number of Young per Occupied Territory										**0.48**

Table 6-120 Organochlorine Concentrations in Addled Eggs (from Dykstra and Meyer, 1996)

Location	Year	State	Nest Number	Nest Name	Eggs		Blood Plasma	
					Total PCBs (ppm ww)	DDE (ppm ww)	Total PCBs (ppb ww)	DDE (ppb ww)
Appleton/Little Rapids								
	1990	WI	OU-1a	Kaukauna	36	1.1		
	1991	WI	OU-1a	Kaukauna			120	ND
	1992	WI	OU-1a	Kaukauna			318	ND
	1993	WI	OU-1a	Kaukauna			226	ND
	1994	WI	OU-1a	Kaukauna			547	54
	1995	WI	OU-1a	Kaukauna			290	9
Zone 3A								
	1987	WI	MT-07	Peshtigo River	19.0	2.4		
	1991	WI	MT-07	Peshtigo River	56.5	12		
	1992	WI	MT-16	Peshtigo River, North	66.6	14.7	901	361
	1994	WI	OC-4	Oconto River			393	95
	1994	WI	MT-17	Blueberry Island			83	4
	1995	WI	MT-17	Blueberry Island			87	13
Zone 4								
	1986	MI	De-15	Boutlier Lake	55.1	29.9		
	1987	MI	De-13	Granskog Lake			229	111
	1987	MI	De-15	Boutlier Lake			319	235
	1990	MI	De-17	Fish Dam River	26.4	10		
	1991	MI	De-17	Fish Dam River	27.2	7.4		
	1992	MI	De-17	Fish Dam River	27.8	10.7		
	1992	MI	De-18	Squaw Point	28.7	12.3		
Lake Michigan/Door Peninsula								
	1994	WI	DO-01	Toft Point			121	46
	1995	WI	DO-01	Toft Point			150	29
			Mean		**35.0**	**10.3**	**207**	**53.0**

Note:

ND - Non-detect.

Table 6-121 Summary of BLRA Risk Assessment Results for Little Lake Butte des Morts Reach

Assessment Endpoint (What is being protected?)	Risk Questions	Constituent	Range of RME HQ Values		Risk Potential
			NOAEC	LOAEC	
1. Functioning water column invertebrate communities.	Are levels of site contaminants in surface water sufficient to cause adverse alterations to the functioning of water column invertebrate communities?	Lead	< 0.1		No
		Mercury	16		Yes
		estimated Total PCBs	1.1	0.1	Potential
2. Functioning benthic invertebrate communities.	Are levels of site contaminants in surface sediment sufficient to cause adverse alterations to the functioning of benthic invertebrate communities?	Arsenic	0.4		No
		Lead	15		Yes
		Mercury	8.5		Yes
		2,3,7,8-TCDD	1.1		Yes
		Total PCBs (N)	723		Yes
		Total PCBs (I_0)	105		Yes
		Total PCBs (I_d)	119		Yes
		Dieldrin	< 0.1		No
		p,p'-DDD	5.4		Yes
		p,p'-DDE	—		No
		p,p'-DDT	7.1		Yes
3. Benthic fish survival and reproduction.	Are levels of site contaminants sufficient to cause survival or reproductive impairment in benthic fish?	Arsenic	0.3	< 0.1	No
		Mercury	0.2	< 0.1	No
		Total PCBs	3.9	0.4	Potential
		PCB Congeners	0.1	0.1	No
		Dieldrin	< 0.1	< 0.1	No
		o,p'-DDD	—	—	No
		o,p'-DDE	< 0.1	< 0.1	No
		o,p'-DDT	—	—	No
		p,p'-DDD	< 0.1	< 0.1	No
		p,p'-DDE	0.1	< 0.1	No
		p,p'-DDT	—	—	No
4. Pelagial fish survival and reproduction.	Are levels of site contaminants sufficient to cause survival or reproductive impairment in pelagial fish?	Mercury	0.1	< 0.1	No
		Total PCBs	5.0	0.5	Potential
		PCB Congeners	0.2	0.1	No
		Dieldrin	—	—	No
		o,p'-DDD	—	—	No
		o,p'-DDE	0.1	< 0.1	No
		o,p'-DDT	—	—	No
		p,p'-DDD	0.1	< 0.1	No
		p,p'-DDE	0.2	< 0.1	No
		p,p'-DDT	—	—	No
5. Insectivorous bird survival, physiology, and reproduction.	Are levels of site contaminants sufficient to cause survival or reproductive impairment, or deformity in insectivorous birds?	Total PCBs	6.6	0.7	Potential
		PCB Congeners	13	0.5	Potential
		Dieldrin	—	—	No
		o,p'-DDD	—	—	No
		o,p'-DDE	—	—	No
		o,p'-DDT	—	—	No
		p,p'-DDD	—	—	No
		p,p'-DDE	0.1	< 0.1	No
		p,p'-DDT	—	—	No

Table 6-121 Summary of BLRA Risk Assessment Results for Little Lake Butte des Morts Reach (Continued)

Assessment Endpoint (What is being protected?)	Risk Questions	Constituent	Range of RME HQ Values		Risk Potential
			NOAEC	LOAEC	
6. Piscivorous bird survival, physiology, and reproduction.	Are levels of site contaminants sufficient to cause survival or reproductive impairment, or deformity in piscivorous birds?	Mercury *	1.6	0.1	Potential
		Total PCBs *	2.3	0.1	Potential
		Dieldrin *	< 0.1	< 0.1	No
		p,p'-DDE *	0.3	< 0.1	No
7. Carnivorous bird survival, physiology, and reproduction.	Are levels of site contaminants sufficient to cause survival or reproductive impairment, or deformity in carnivorous birds?	Mercury *	0.8	0.1	No
		Total PCBs *	3.2	0.3	Potential
		Dieldrin *	< 0.1	< 0.1	No
		p,p'-DDE *	0.2	< 0.1	No
8. Piscivorous mammal survival and reproduction.	Are levels of site contaminants sufficient to cause survival or reproductive impairment in piscivorous mammals?	Mercury *	0.2	0.1	No
		Total PCBs (N) *	14	6.8	Yes
		Total PCBs (I_0) *	12	5.8	Yes
		Total PCBs (I_d) *	12	5.8	Yes
		Dieldrin *	< 0.1	< 0.1	No
		p,p'-DDE *	< 0.1	< 0.1	No

Notes:

"—" indicates that the compound was not detected and, therefore, there is assumed to be no risk.

* Estimated total dietary exposure.

Table 6-122 Summary of BLRA Risk Assessment Results for Appleton to Little Rapids Reach

Assessment Endpoint (What is being protected?)	Risk Questions	Constituent	Range of RME HQ Values		Risk Potential
			NOAEC	LOAEC	
1. Functioning water column invertebrate communities.	Are levels of site contaminants in surface water sufficient to cause adverse alterations to the functioning of water column invertebrate communities?	Arsenic	—	—	No
		Lead	< 0.1		No
		Mercury	0.2		No
		estimated Total PCBs	1.2	0.1	Potential
		Dieldrin	—	—	No
		p,p'-DDD	—	—	No
		p,p'-DDE	—	—	No
		p,p'-DDT	—	—	No
2. Functioning benthic invertebrate communities.	Are levels of site contaminants in surface sediment sufficient to cause adverse alterations to the functioning of benthic invertebrate communities?	Arsenic	0.5		No
		Lead	2.6		Yes
		Mercury	10		Yes
		Total PCBs (N)	483		Yes
		Total PCBs (I_0)	5.9		Yes
		Total PCBs (I_d)	47		Yes
		Dieldrin	—		No
		p,p'-DDD	0.5		No
		p,p'-DDE	—		No
		p,p'-DDT	0.5		No
3. Benthic fish survival and reproduction.	Are levels of site contaminants sufficient to cause survival or reproductive impairment in benthic fish?	Mercury	0.4	< 0.1	No
		Total PCBs	4.7	0.5	Potential
		PCB Congeners	0.1	0.1	No
		Dieldrin	—	—	No
		o,p'-DDD	—	—	No
		o,p'-DDT	—	—	No
		p,p'-DDD	—	—	No
		p,p'-DDE	0.3	< 0.1	No
		p,p'-DDT	—	—	No
4. Pelagial fish survival and reproduction.	Are levels of site contaminants sufficient to cause survival or reproductive impairment in pelagial fish?	Mercury	0.8	0.1	No
		Total PCBs	5.1	0.5	Potential
		PCB Congeners	0.1	< 0.1	No
		Dieldrin	—	—	No
		p,p'-DDD	< 0.1	< 0.1	No
		p,p'-DDE	0.2	< 0.1	No
		p,p'-DDT	—	—	No
5. Piscivorous bird survival, physiology, and reproduction.	Are levels of site contaminants sufficient to cause survival or reproductive impairment, or deformity in piscivorous birds?	Mercury *	1.5	0.1	Potential
		Total PCBs *	5.3	0.2	Potential
		Dieldrin *	< 0.1	< 0.1	No
		p,p'-DDE *	0.3	< 0.1	No

Table 6-122 Summary of BLRA Risk Assessment Results for Appleton to Little Rapids Reach (Continued)

Assessment Endpoint (What is being protected?)	Risk Questions	Constituent	Range of RME HQ Values		Risk Potential
			NOAEC	LOAEC	
6. Carnivorous bird survival, physiology, and reproduction.	Are levels of site contaminants sufficient to cause survival or reproductive impairment, or deformity in carnivorous birds?	Mercury	7.0	0.7	Potential
		Mercury *	1.8	0.2	Potential
		Total PCBs	45	4.7	Yes
		Total PCBs *	3.7	0.4	Potential
		Dieldrin	0.7	0.1	No
		Dieldrin *	< 0.1	< 0.1	No
		p,p'-DDD	0.1	< 0.1	No
		p,p'-DDE	0.4	0.2	No
		p,p'-DDE *	0.5	< 0.1	No
		p,p'-DDT	—	—	No
7. Piscivorous mammal survival and reproduction.	Are levels of site contaminants sufficient to cause survival or reproductive impairment in piscivorous mammals?	Mercury *	0.4	0.1	No
		Total PCBs (N) *	15	7.7	Yes
		Total PCBs (I_0) *	14	6.9	Yes
		Total PCBs (I_d) *	14	7.0	Yes
		Dieldrin *	0.1	< 0.1	No
		p,p'-DDE *	< 0.1	< 0.1	No

Notes:

"—" indicates that the compound was not detected and, therefore, there is assumed to be no risk.

Tissue data was not available for insectivorous birds, therefore, risk could not be estimated.

* Estimated total dietary exposure.

Table 6-123 Summary of BLRA Risk Assessment Results for Little Rapids to De Pere Reach

Assessment Endpoint (What is being protected?)	Risk Questions	Constituent	Range of RME HQ Values		Risk Potential
			NOAEC	LOAEC	
1. Functioning water column invertebrate communities.	Are levels of site contaminants in surface water sufficient to cause adverse alterations to the functioning of water column invertebrate communities?	Lead	< 0.1		No
		Mercury	16		Yes
		estimated Total PCBs	0.9	0.1	No
2. Functioning benthic invertebrate communities.	Are levels of site contaminants in surface sediment sufficient to cause adverse alterations to the functioning of benthic invertebrate communities?	Arsenic	0.4		No
		Lead	8.0		Yes
		Mercury	24		Yes
		2,3,7,8-TCDD	1.7		Yes
		Total PCBs (N)	334		Yes
		Total PCBs (I_0)	66		Yes
		Total PCBs (I_d)	67		Yes
		Dieldrin	—		No
		p,p'-DDD	0.8		No
		p,p'-DDE	15		Yes
		p,p'-DDT	2.9		Yes
3. Benthic fish survival and reproduction.	Are levels of site contaminants sufficient to cause survival or reproductive impairment in benthic fish?	Mercury	6.0	0.1	Potential
		Total PCBs	7.6	0.8	Potential
		PCB Congeners	0.2	0.1	No
		Dieldrin	—	—	No
		o,p'-DDD	—	—	No
		o,p'-DDE	—	—	No
		o,p'-DDT	—	—	No
		p,p'-DDD	< 0.1	< 0.1	No
		p,p'-DDE	0.4	< 0.1	No
		p,p'-DDT	—	—	No
4. Pelagial fish survival and reproduction.	Are levels of site contaminants sufficient to cause survival or reproductive impairment in pelagial fish?	Mercury	6.4	0.1	Potential
		Total PCBs	6.0	0.6	Potential
		PCB Congeners	0.2	0.1	No
		Dieldrin	< 0.1	< 0.1	No
		o,p'-DDD	—	—	No
		o,p'-DDE	0.2	< 0.1	No
		o,p'-DDT	—	—	No
		p,p'-DDD	—	—	No
		p,p'-DDE	0.7	0.1	No
		p,p'-DDT	—	—	No
5. Piscivorous bird survival, physiology, and reproduction.	Are levels of site contaminants sufficient to cause survival or reproductive impairment, or deformity in piscivorous birds?	Mercury *	3.2	0.1	Potential
		Total PCBs *	1.6	0.1	Potential
		Dieldrin *	< 0.1	< 0.1	No
		p,p'-DDE *	0.4	< 0.1	No
6. Carnivorous bird survival, physiology, and reproduction.	Are levels of site contaminants sufficient to cause survival or reproductive impairment, or deformity in carnivorous birds?	Mercury *	2.2	0.2	Potential
		Total PCBs *	5.6	0.6	Potential
		Dieldrin *	< 0.1	< 0.1	No
		p,p'-DDE *	0.9	0.1	No

Table 6-123 Summary of BLRA Risk Assessment Results for Little Rapids to De Pere Reach (Continued)

Assessment Endpoint (What is being protected?)	Risk Questions	Constituent	Range of RME HQ Values		Risk Potential
			NOAEC	LOAEC	
7. Piscivorous mammal survival and reproduction.	Are levels of site contaminants sufficient to cause survival or reproductive impairment in piscivorous mammals?	Mercury *	0.6	0.2	No
		Total PCBs (N) *	23	12	Yes
		Total PCBs (I_0) *	22	11	Yes
		Total PCBs (I_d) *	22	11	Yes
		Dieldrin *	0.3	0.1	No
		p,p'-DDE *	< 0.1	< 0.1	No

Notes:

"—" indicates that the compound was not detected and, therefore, there is assumed to be no risk.

Tissue data was not available for insectivorous birds, therefore, risk could not be estimated.

* Estimated total dietary exposure.

Table 6-124 Summary of BLRA Risk Assessment Results for Green Bay Zone 1

Assessment Endpoint (What is being protected?)	Risk Questions	Constituent	Range of RME HQ Values		Risk Potential
			NOAEC	LOAEC	
1. Functioning water column invertebrate communities.	Are levels of site contaminants in surface water sufficient to cause adverse alterations to the functioning of water column invertebrate communities?	Arsenic	< 0.1		No
		Lead	0.1		No
		Mercury	0.1		No
		2,3,7,8-TCDD	—	—	No
		2,3,7,8-TCDF	—	—	No
		estimated Total PCBs	1.4	0.1	Potential
		Dieldrin	—	—	No
		p,p'-DDD	0.1		No
		p,p'-DDE	0.2		No
		p,p'-DDT	0.1		No
2. Functioning benthic invertebrate communities.	Are levels of site contaminants in surface sediment sufficient to cause adverse alterations to the functioning of benthic invertebrate communities?	Arsenic	1.4		Yes
		Lead	2.7		Yes
		Mercury	8.1		Yes
		Total PCBs (N)	174		Yes
		Total PCBs (I_0)	94		Yes
		Total PCBs (I_d)	94		Yes
		Dieldrin	—		No
		p,p'-DDD	1.3		Yes
		p,p'-DDE	1.3		Yes
		p,p'-DDT	—		No
3. Insectivorous bird survival, physiology, and reproduction.	Are levels of site contaminants sufficient to cause survival or reproductive impairment, or deformity in insectivorous birds?	Total PCBs	5.6	0.6	Potential
		Dieldrin	—	—	No
		o,p'-DDD	—	—	No
		o,p'-DDE	—	—	No
		o,p'-DDT	—	—	No
		p,p'-DDD	< 0.1	< 0.1	No
		p,p'-DDE	0.1	0.1	No
		p,p'-DDT	—	—	No
4. Piscivorous mammal survival and reproduction.	Are levels of site contaminants sufficient to cause survival or reproductive impairment in piscivorous mammals?	Mercury *	0.2	0.1	No
		Total PCBs (N) *	29	14	Yes
		Total PCBs (I_0) *	28	14	Yes
		Total PCBs (I_d) *	28	14	Yes
		Dieldrin *	0.6	0.3	No
		p,p'-DDE *	< 0.1	< 0.1	No

Notes:

"—" indicates that the compound was not detected and, therefore, there is assumed to be no risk.

Benthic and pelagial fish survival and reproduction for Green Bay Zone 1 are summarized with the Green Bay Zone 2 fish.

Estimated risks to piscivorous and carnivorous birds based on dietary modeling were the same as the risks for Green Bay Zone 2.

* Estimated total dietary exposure.

Table 6-125 Summary of BLRA Risk Assessment Results for Green Bay Zone 2

Assessment Endpoint (What is being	Risk Questions	Constituent	Range of RME HQ Values		Risk Potential
			NOAEC	LOAEC	
1. Functioning water column invertebrate communities.	Are levels of site contaminants in surface water sufficient to cause adverse alterations to the functioning of water column invertebrate communities?	Lead	< 0.1		No
		Mercury	11		Yes
		estimated Total PCBs	0.4	< 0.1	No
2. Functioning benthic invertebrate communities.	Are levels of site contaminants in surface sediment sufficient to cause adverse alterations to the functioning of benthic invertebrate communities?	Arsenic	0.2		No
		Lead	0.8		No
		Mercury	8.8		Yes
		Total PCBs (N)	23		Yes
		Total PCBs (I_d)	37		Yes
		Dieldrin	—		No
		p,p'-DDD	—		No
		p,p'-DDE	—		No
		p,p'-DDT	—		No
3. Benthic fish survival and reproduction.	Are levels of site contaminants sufficient to cause survival or reproductive impairment in benthic fish?	Mercury	0.6	0.1	No
		Total PCBs	9.7	1.0	Potential
		PCB Congeners	0.8	0.4	No
		Dieldrin	0.1	< 0.1	No
		o,p'-DDD	—	—	No
		o,p'-DDE	0.3	< 0.1	No
		o,p'-DDT	—	—	No
		p,p'-DDD	0.3	< 0.1	No
		p,p'-DDE	2.3	0.2	Potential
		p,p'-DDT	—	—	No
4. Pelagial fish survival and reproduction.	Are levels of site contaminants sufficient to cause survival or reproductive impairment in pelagial fish?	Mercury	1.1	0.1	Potential
		Total PCBs	10	1.0	Potential
		PCB Congeners	1.7	0.8	Potential
		Dieldrin	0.2	< 0.1	No
		o,p'-DDD	—	—	No
		o,p'-DDE	0.4	< 0.1	No
		o,p'-DDT	—	—	No
		p,p'-DDD	0.1	< 0.1	No
		p,p'-DDE	1.5	0.2	Potential
		p,p'-DDT	—	—	No
5. Insectivorous bird survival, physiology, and reproduction.	Are levels of site contaminants sufficient to cause survival or reproductive impairment, or deformity in insectivorous birds?	Total PCBs	4	0.4	Potential
		PCB Congeners	12.6	0.1	Potential
		Dieldrin	—	—	No
		o,p'-DDD	—	—	No
		o,p'-DDE	—	—	No
		o,p'-DDT	—	—	No
		p,p'-DDD	< 0.1	< 0.1	No
		p,p'-DDE	2.4	0.1	Potential
		p,p'-DDT	—	—	No

Table 6-125 Summary of BLRA Risk Assessment Results for Green Bay Zone 2 (Continued)

Assessment Endpoint (What is being	Risk Questions	Constituent	Range of RME HQ Values		Risk Potential
			NOAEC	LOAEC	
6. Piscivorous bird survival, physiology, and reproduction.	Are levels of site contaminants sufficient to cause survival or reproductive impairment, or deformity in piscivorous birds?	Mercury *	15	0.6	Potential
		Total PCBs	26	0.7	Potential
		Total PCBs *	14	0.5	Potential
		PCB Congeners	110	0.1	Potential
		Dieldrin	4.4	0.1	Potential
		Dieldrin *	0.3	< 0.1	No
		o,p'-DDD	—	—	No
		o,p'-DDE	—	—	No
		o,p'-DDT	—	—	No
		p,p'-DDD	< 0.1	< 0.1	No
		p,p'-DDE	2.4	0.1	Potential
		p,p'-DDE *	3.9	0.2	Potential
		p,p'-DDT	< 0.1	< 0.1	No
7. Carnivorous bird survival, physiology, and reproduction.	Are levels of site contaminants sufficient to cause survival or reproductive impairment, or deformity in carnivorous birds?	Mercury *	1.6	0.2	Potential
		Total PCBs *	7.5	0.8	Potential
		Dieldrin *	< 0.1	< 0.1	No
		p,p'-DDE *	4.1	0.4	Potential
8. Piscivorous mammal survival and reproduction.	Are levels of site contaminants sufficient to cause survival or reproductive impairment in piscivorous mammals?	Mercury *	0.3	0.1	No
		Total PCBs (N) *	28	14	Yes
		Total PCBs (I_d) *	28	14	Yes
		Dieldrin *	0.6	0.3	No
		p,p'-DDE *	< 0.1	< 0.1	No

Notes:

"—" indicates that the compound was not detected and, therefore, there is assumed to be no risk.

Tissue data was not available for insectivorous birds, therefore, risk could not be estimated.

* Estimated total dietary exposure.

Table 6-126 Summary of BLRA Risk Assessment Results for Green Bay Zone 3A

Assessment Endpoint (What is being	Risk Questions	Constituent	Range of RME HQ Values		Risk Potential
			NOAEC	LOAEC	
1. Functioning water column invertebrate communities.	Are levels of site contaminants in surface water sufficient to cause adverse alterations to the functioning of water column invertebrate communities?	Mercury	—		No
		estimated Total PCBs	0.1	< 0.1	No
2. Functioning benthic invertebrate communities.	Are levels of site contaminants in surface sediment sufficient to cause adverse alterations to the functioning of benthic invertebrate communities?	Arsenic	0.1		No
		Lead	0.1		No
		Mercury	—		No
		Total PCBs (N)	16		Yes
		Total PCBs (I_d)	8.1		Yes
		Dieldrin	—		No
		p,p'-DDD	—		No
		p,p'-DDE	—		No
		p,p'-DDT	—		No
3. Benthic fish survival and reproduction.	Are levels of site contaminants sufficient to cause survival or reproductive impairment in benthic fish?	Mercury	—	—	No
		Total PCBs	4.6	0.5	Potential
		PCB Congeners	< 0.1	< 0.1	No
		Dieldrin	—	—	No
		p,p'-DDD	—	—	No
		p,p'-DDE	0.5	0.1	No
		p,p'-DDT	—	—	No
4. Pelagial fish survival and reproduction.	Are levels of site contaminants sufficient to cause survival or reproductive impairment in pelagial fish?	Mercury	0.2	< 0.1	No
		Total PCBs	6.7	0.7	Potential
		PCB Congeners	0.6	0.3	No
		Dieldrin	0.3	< 0.1	No
		o,p'-DDD	—	—	No
		o,p'-DDE	—	—	No
		o,p'-DDT	—	—	No
		p,p'-DDD	—	—	No
		p,p'-DDE	0.1	< 0.1	No
		p,p'-DDT	—	—	No
5. Piscivorous bird survival, physiology, and reproduction.	Are levels of site contaminants sufficient to cause survival or reproductive impairment, or deformity in piscivorous birds?	Mercury *	2.5	0.1	Potential
		Total PCBs *	5.6	0.2	Potential
		Dieldrin *	0.1	< 0.1	No
		p,p'-DDE *	1.0	< 0.1	No
6. Carnivorous bird survival, physiology, and reproduction.	Are levels of site contaminants sufficient to cause survival or reproductive impairment, or deformity in carnivorous birds?	Mercury	0.6	0.1	No
		Mercury *	0.6	0.1	No
		Total PCBs	16	1.6	Yes
		Total PCBs *	4.2	0.4	Potential
		Dieldrin	2	0.2	Potential
		Dieldrin *	0.1	< 0.1	No
		p,p'-DDD	< 0.1	< 0.1	No
		p,p'-DDE	0.8	0.5	No
		p,p'-DDE *	0.1	< 0.1	No
		p,p'-DDT	—	—	No

Table 6-126 Summary of BLRA Risk Assessment Results for Green Bay Zone 3A (Continued)

Assessment Endpoint (What is being	Risk Questions	Constituent	Range of RME HQ Values		Risk Potential
			NOAEC	LOAEC	
7. Piscivorous mammal survival and reproduction.	Are levels of site contaminants sufficient to cause survival or reproductive impairment in piscivorous mammals?	Mercury *	0.1	< 0.1	No
		Total PCBs (N) *	15	7.6	Yes
		Total PCBs (I_d) *	15	7.6	Yes
		Dieldrin *	1.2	0.6	Potential
		p,p'-DDE *	< 0.1	< 0.1	No

Notes:

"—" indicates that the compound was not detected and, therefore, there is assumed to be no risk.

Tissue data was not available for insectivorous birds, therefore, risk could not be estimated.

* Estimated total dietary exposure.

Table 6-127 Summary of BLRA Risk Assessment Results for Green Bay Zone 3B

Assessment Endpoint (What is being protected?)	Risk Questions	Constituent	Range of RME HQ Values		Risk Potential
			NOAEC	LOAE	
1. Functioning water column invertebrate communities.	Are levels of site contaminants in surface water sufficient to cause adverse alterations to the functioning of water column invertebrate communities?	Mercury	0.1		No
		estimated Total PCBs	0.1	< 0.1	No
2. Functioning benthic invertebrate communities.	Are levels of site contaminants in surface sediment sufficient to cause adverse alterations to the functioning of benthic invertebrate communities?	Arsenic	1.2		Yes
		Lead	1.4		Yes
		Mercury	1.1		Yes
		Total PCBs (N)	26		Yes
		Total PCBs (I_d)	15		Yes
		Dieldrin	—		No
		p,p'-DDD	—		No
		p,p'-DDE	—		No
		p,p'-DDT	—		No
3. Benthic fish survival and reproduction.	Are levels of site contaminants sufficient to cause survival or reproductive impairment in benthic fish?	Mercury	—	—	No
		Total PCBs	0.8	0.1	No
		PCB Congeners	< 0.1	< 0.1	No
		Dieldrin	—	—	No
		p,p'-DDD	—	—	No
		p,p'-DDE	0.1	0.1	No
		p,p'-DDT	—	—	No
4. Pelagial fish survival and reproduction.	Are levels of site contaminants sufficient to cause survival or reproductive impairment in pelagial fish?	Mercury	2.6	0.3	Potential
		Total PCBs	15	1.5	Yes
		PCB Congeners	0.3	0.2	No
		Dieldrin	0.2	< 0.1	No
		p,p'-DDD	—	—	No
		p,p'-DDE	1.8	0.2	Potential
		p,p'-DDT	—	—	No
5. Piscivorous bird survival, physiology, and reproduction.	Are levels of site contaminants sufficient to cause survival or reproductive impairment, or deformity in piscivorous birds?	Mercury *	3.1	0.1	Potential
		Total PCBs	19	1.9	Yes
		Total PCBs *	10	0.4	Potential
		PCB Congeners	18	0.1	Potential
		Dieldrin	2.4	0.2	Potential
		Dieldrin *	0.1	< 0.1	No
		o,p'-DDD	—	—	No
		o,p'-DDE	—	—	No
		o,p'-DDT	—	—	No
		p,p'-DDD	< 0.1	< 0.1	No
		p,p'-DDE	1.5	0.9	Potential
		p,p'-DDE *	2.2	0.1	Potential
		p,p'-DDT	< 0.1	< 0.1	No
6. Carnivorous bird survival, physiology, and reproduction.	Are levels of site contaminants sufficient to cause survival or reproductive impairment, or deformity in carnivorous birds?	Mercury *	3.8	0.4	Potential
		Total PCBs *	7.3	0.7	Potential
		Dieldrin *	0.1	< 0.1	No
		p,p'-DDE *	1.9	0.2	Potential
7. Piscivorous mammal survival and reproduction.	Are levels of site contaminants sufficient to cause survival or reproductive impairment in piscivorous mammals?	Mercury *	0.4	0.2	No
		Total PCBs (N) *	24	12	Yes
		Total PCBs (I_d) *	24	12	Yes
		Dieldrin *	1.2	0.6	Potential
		p,p'-DDE *	< 0.1	< 0.1	No

Notes:

"—" indicates that the compound was not detected and, therefore, there is assumed to be no risk.

Tissue data was not available for insectivorous birds, therefore, risk could not be estimated.

* Estimated total dietary exposure.

Table 6-128 Summary of BLRA Risk Assessment Results for Green Bay Zone 4

Assessment Endpoint (What is being	Risk Questions	Constituent	Range of RME HQ Values		Risk Potential
			NOAEC	LOAEC	
1. Functioning water column invertebrate communities.	Are levels of site contaminants in surface water sufficient to cause adverse alterations to the functioning of water column invertebrate communities?	Mercury	—		No
		estimated Total PCBs	< 0.1	< 0.1	No
2. Functioning benthic invertebrate communities.	Are levels of site contaminants in surface sediment sufficient to cause adverse alterations to the functioning of benthic invertebrate communities?	Arsenic	0.7		No
		Lead	0.1		No
		Mercury	0.6		No
		Total PCBs (N)	3.7		Yes
		Total PCBs (I_d)	1.4		Yes
		Dieldrin	—		No
		p,p'-DDD	—		No
		p,p'-DDE	—		No
		p,p'-DDT	—		No
3. Pelagial fish survival and reproduction.	Are levels of site contaminants sufficient to cause survival or reproductive impairment in pelagial fish?	Mercury	0.9	0.1	No
		Total PCBs	4.3	0.4	Potential
		PCB Congeners	0.2	0.1	No
		Dieldrin	0.3	< 0.1	No
		p,p'-DDD	0.1	< 0.1	No
		p,p'-DDE	2.0	0.2	Potential
		p,p'-DDT	0.1	< 0.1	No
4. Piscivorous bird survival, physiology, and reproduction.	Are levels of site contaminants sufficient to cause survival or reproductive impairment, or deformity in piscivorous birds?	Mercury *	1.8	0.1	Potential
		Total PCBs *	6.5	0.3	Potential
		Dieldrin *	0.1	< 0.1	No
		p,p'-DDE *	0.4	< 0.1	No
5. Carnivorous bird survival, physiology, and reproduction.	Are levels of site contaminants sufficient to cause survival or reproductive impairment, or deformity in carnivorous birds?	Mercury *	2.9	0.3	Potential
		Total PCBs *	4.4	0.4	Potential
		Dieldrin *	< 0.1	< 0.1	No
		p,p'-DDE *	6.6	< 0.1	Potential
6. Piscivorous mammal survival and reproduction.	Are levels of site contaminants sufficient to cause survival or reproductive impairment in piscivorous mammals?	Mercury *	0.5	0.2	No
		Total PCBs (N) *	18	8.8	Yes
		Total PCBs (I_d) *	17	8.7	Yes
		Dieldrin *	0.8	0.4	No
		p,p'-DDE *	< 0.1	< 0.1	No

Notes:

"—" indicates that the compound was not detected and, therefore, there is assumed to be no risk.

Risk to benthic fish survival and reproduction could not be evaluated because benthic fish were not sampled in this Zone.

Tissue data was not available for insectivorous birds, therefore, risk could not be estimated.

* Estimated total dietary exposure.

Table 6-129 Summary of Published Sediment Quality Criteria

Contaminant Type	Agency Program	Effect Level	Effect Concentration in µg/kg	Reference
Total PCBs	**No Effect Concentrations**			
	EPA ARCS - Assessment and Remediation of Contaminated Sediments (ARCS) Program (EPA, 1996a)	NEC	194	EPA (1997d)
	Threshold Effect Concentrations			
	EPA ARCS - Assessment and Remediation of Contaminated Sediments (ARCS) Program (EPA, 1996a)	TEC	31.62	Jones *et al.* (1997)
		TEL	32	EPA (1997d) ***ARCS values for the HA28 assay from Ingersoll *et al.* (1996) and Smith *et al.* (1996)
	Environment Canada	TEL	34.1	Smith *et al.* (1996)
	Ontario Ministry of the Environment	LEL	70	Persuad *et al.* (1993)
	Washington State	LAET	21	Cubbage *et al.* (1997)
		FSQV	21	
	EPA OSWER	ET	23	EPA (1996g)
		SLC	3	Neff *et al.* (1986)
		LAET (Microtox)	21	Cubbage *et al.* (1997)
		TEL	22	MacDonald *et al.* (1996)
		ERL	23	Long *et al.* (1995)
		TEL-HA28	32	Ingersoll *et al.* (1996)
		TEL	34	Smith *et al.* (1996)
		SLC	43	Neff *et al.* (1986)
		ERL	50	Long and Morgan (1990)
		ERL-HA28	50	Ingersoll *et al.* (1996)
		LEL	70	Persuad *et al.* (1993)
		LAET-C (bivalve)	88	Becker *et al.* (1990)
		LAET-PS (Microtox)	130	
		MET	200	Environment Canada and Ministere de l'Environnement du Quebec (1992)

Table 6-129 Summary of Published Sediment Quality Criteria (Continued)

Contaminant Type	Agency Program	Effect Level	Effect Concentration in µg/kg	Reference
Total PCBs	**Midrange Effect Concentrations**			
	EPA ARCS - Assessment and Remediation of Contaminated Sediments (ARCS) Program (EPA, 1996a)	PEC	244.7	Jones *et al.* (1997)
		PEL	240	EPA (1997d) ***ARCS values for the HA28 assay from Ingersoll *et al.* (1996) and Smith *et al.* (1996)
	Environment Canada	PEL	227	Environment Canada Smith *et al.* (1996)
		ERM	180	Long *et al.* (1995)
		PEL	189	MacDonald *et al.* (1996)
		NEC	190	Ingersoll *et al.* (1996)
		PEL-HA28	240	
		PEL	277	Smith *et al.* (1996)
		MAET-C (benthic)	260	Becker *et al.* (1990)
		ERM	400	Long and Morgan (1990)
		PAET (amphipod)	450	Cubbage *et al.* (1997)
		ERM-HA28	730	Ingersoll *et al.* (1996)
		SEC	835	MacDonald *et al.* (1996)
		MAET-PS (benthic)	1,000	Becker *et al.* (1990)
		MAET-PS (oyster)	1,100	
	Extreme Effect Concentrations			
	Ontario Ministry of Environment	SEL	**5,300**	Persuad *et al.* (1993)
		HAET-C (amphipod)	820	Becker *et al.* (1990)
		HAET (amphipod)	960	Cubbage *et al.* (1997)
		TET	1,000	Environment Canada and Ministere de l'Environnement du Quebec (1992)
		HAET-PS	3,100	Becker *et al.* (1990)
		SEL	5,300	Persuad *et al.* (1993)

Table 6-129 Summary of Published Sediment Quality Criteria (Continued)

Contaminant Type	Agency Program	Effect Level	Effect Concentration in µg/kg	Reference
Aroclor 1016	Ontario Ministry of the Environment	LEL	7	Persuad *et al.* (1993)
		SEL	**530**	
Aroclor 1248	Ontario Ministry of the Environment	LEL	30	Persuad *et al.* (1993)
		SEL	**1,500**	
	Washington State	FSQV	21	Cubbage *et al.* (1997)
		LAET	21	
Aroclor 1254	Ontario Ministry of the Environment	LEL	60	Persuad *et al.* (1993)
		SEL	**340**	
	Washington State	LAET	7.3	Cubbage *et al.* (1997)
		FSQV	7.3	
Aroclor 1260	Ontario Ministry of the Environment	LEL	5	Persuad *et al.* (1993)
		SEL	**240**	
Biphenyl	EPA OSWER	ET	1,100	EPA (1996e)

Notes:

C - California
ERL - Effects Range Low
ERM - Effects Range Median
ET - Ecotox Threshold
FSQV - Freshwater Sediment Quality Values
HA28 - *Hyalella azteca* 28-day test
HAET - Highest Apparent Effects Threshold
LAET - Lowest Apparent Effects Threshold
LEL - Lowest Effect Level
MAET - Moderate Apparent Effects Threshold
MET - Moderate Effect Threshold
- µg/% organic carbon
NEC - No Effect Concentration
PAET - Probable Apparent Effects Threshold
PEC - Probable Effect Concentration
PEL - Probable Effect Level
PS - Puget Sound
SEC - Sediment Effect Concentration
SEL - Severe Effect Level
SLC - Screening Level Concentration
TEC - Threshold Effect Concentration
TEL - Threshold Effect Level
TET - Toxic Effect Threshold

Table 6-130 Hazard Quotients for 90th Percentile COPC Concentrations in Walleye

Area / Analyte	Number of Samples	Number of Detects	Detection Frequency (%)	90th Percentile	Criteria		Hazard Quotients	
					NOAEC	LOAEC	NOAEC 90th Percentile	LOAEC 90th Percentile
Little Lake Butte des Morts								
PCBs (µg/kg)								
Total PCBs	13	11	85	3,800	760	7,600	5.0	0.5
Green Bay zones 1 and 2								
Metals (mg/kg)								
Mercury	11	10	91	0.4	0.25	2.37	1.5	0.2
PCBs (µg/kg)								
Total PCBs	91	91	100	10,923	760	7,600	14	1.4
Pesticides (µg/kg)								
Dieldrin	70	58	83	80.7	370	3,700	0.2	< 0.1
p,p'-DDD	14	1	7	62.5	300	2,950	0.2	< 0.1
p,p'-DDE	14	14	100	705	300	2,950	2.4	0.2
Green Bay Zone 3A								
PCBs (µg/kg)								
Total PCBs	14	14	100	7,000	760	7,600	9.2	0.9
Pesticides (µg/kg)								
Dieldrin	10	10	100	84.4	370	3,700	0.2	< 0.1
Green Bay Zone 3B								
PCBs (µg/kg)								
Total PCBs	26	26	100	12,421	760	7,600	16	1.6
Pesticides (µg/kg)								
Dieldrin	15	12	80	95.0	370	3,700	0.3	< 0.1
Green Bay Zone 4								
Metals (mg/kg)								
Mercury	20	20	100	0.3	0.25	2.37	1.2	0.1
PCBs (µg/kg)								
Total PCBs	36	36	100	5,867	760	7,600	7.7	0.8
Pesticides (µg/kg)								
Dieldrin	33	33	100	92.4	370	3,700	0.2	< 0.1
p,p'-DDD	20	20	100	43.4	300	2,950	0.1	< 0.1
p,p'-DDE	20	20	100	995	300	2,950	3.3	0.3
p,p'-DDT	20	20	100	59.6	300	2,950	0.2	< 0.1

Table 6-131 Hazard Quotients for 90th Percentile PCB Congener Concentrations in Whole Walleye

Area	Analyte	Number of Samples	Number of Detects	Detection Frequency (%)	90th Percentile	TEF	90th Percentile TEC	Hazard Quotients NOAEC 90th Percentile	Hazard Quotients LOAEC 90th Percentile
Green Bay zones 1 and 2									
	PCB Congeners (µg/kg)								
	PCB Congener 77	16	16	100	9.6	0.0001	0.00095823		
	PCB Congener 81	69	65	94	28.0	0.0005	0.014		
	PCB Congener 105	27	25	93	163.4	0.000005	0.000817		
	PCB Congener 118	83	83	100	306.8	0.000005	0.001534		
	PCB Congener 126	16	15	94	3.8	0.005	0.01915		
	PCB Congener 169	25	16	64	1.2	0.00005	6.013E-05		
	Total TEQ						**0.037**	**0.9**	**0.4**
Green Bay Zone 3A									
	PCB Congener 81	11	10	91	23.2	0.0005	0.0116		
	PCB Congener 118	14	14	100	195.8	0.000005	0.00097905		
	Total TEQ						**0.013**	**0.3**	**0.1**
Green Bay Zone 3B									
	PCB Congener 81	16	13	81	23.3	0.0005	0.01165		
	PCB Congener 105	13	13	100	222.0	0.000005	0.00111012		
	PCB Congener 118	25	25	100	509.8	0.000005	0.00254883		
	PCB Congener 169	12	9	75	6.8	0.00005	0.00034183		
	Total TEQ						**0.016**	**0.4**	**0.2**
Green Bay Zone 4									
	PCB Congener 81	14	13	93	19.5	0.0005	0.00975		
	PCB Congener 118	16	16	100	229.1	0.000005	0.00114565		
	Total TEQ						**0.011**	**0.3**	**0.1**

Note:

TEQ Criteria
NOAEC = 0.041 µg/kg
LOAEC = 0.084 µg/kg

Table 6-132 Hazard Quotients for 90th Percentile COPC Concentrations in Double-crested Cormorants

Area	Analyte	Tissue	Number of Samples	Number of Detects	Detection Frequency (%)	90th Percentile	Reproduction Criteria		Hazard Quotients		Deformity Criteria		Hazard Quotients	
							NOAEC	LOAEC	NOAEC 90th Percentile	LOAEC 90th Percentile	NOAEC	LOAEC	NOAEC 90th Percentile	LOAEC 90th Percentile
Green Bay Zone 2														
	PCBs (µg/kg)													
	Total PCBs	egg	34	34	100	25,000	4,700	7,600	5.3	3.3	800	8,000	31	3.1
	Total PCBs	whole body	74	74	100	21,500	4,700	7,600	4.6	2.8	800	8,000	27	2.7
	Pesticides (µg/kg)													
	Dieldrin	egg	34	32	94	545	100	1,000	5.5	0.5				
	p,p'-DDD	egg	34	22	65	29.0	3,000	5,100	< 0.1	< 0.1				
	p,p'-DDE	egg	34	34	100	8,800	3,000	5,100	2.9	1.7				
	p,p'-DDT	egg	34	3	9	13.0	3,000	5,100	< 0.1	< 0.1				
	Dieldrin	whole body	73	73	100	412	100	1,000	4.1	0.4				
	p,p'-DDD	whole body	73	14	19	13.6	3,000	5,100	< 0.1	< 0.1				
	p,p'-DDE	whole body	73	73	100	5,060	3,000	5,100	1.7	1.0				
	p,p'-DDT	whole body	73	19	26	18.0	3,000	5,100	< 0.1	< 0.1				
Green Bay Zone 3B														
	PCBs (µg/kg)													
	Total PCBs	whole body	21	20	95	13,400	4,700	7,600	2.9	1.8	800	8,000	17	1.7
	Pesticides (µg/kg)													
	Dieldrin	whole body	20	19	95	269	100	1,000	2.7	0.3				
	p,p'-DDD	whole body	20	3	15	10.0	3,000	5,100	< 0.1	< 0.1				
	p,p'-DDE	whole body	20	20	100	5,850	3,000	5,100	2.0	1.1				
	p,p'-DDT	whole body	20	11	55	20.0	3,000	5,100	< 0.1	< 0.1				

Table 6-133 Hazard Quotients for 90th Percentile PCB Congener Concentrations in Double-crested Cormorants

Area	Analyte	Tissue	Number of Samples	Number of Detects	Detection Frequency (%)	90th Percentile	Tillitt *et al.* (1991b) TEF	90th Percentile TEC	Van den Berg *et al.* (1998) TEF	90th Percentile TEC	Hazard Quotients Based on Tillitt TEFs			Hazard Quotients Based on Van den Berg TEFs		
											NOAEC 90th Percentile	LD_{20} 90th Percentile	LD_{30} 90th Percentile	NOAEC 90th Percentile	LD_{20} 90th Percentile	LD_{30} 90th Percentile
Green Bay Zone 2																
	PCB Congeners (µg/kg)															
	PCB Congener 77	whole body	26	9	35	1.5	0.000018	0.0000	0.05	0.0755	< 0.1	< 0.1	< 0.1	11	0.4	0.2
	PCB Congener 105	whole body	26	26	100	429	0.0000076	0.0033	0.0001	0.0429	0.5	< 0.1	< 0.1	6.1	0.2	0.1
	PCB Congener 118/106	whole body	26	26	100	1,046	0.00000037	0.0004	0.00001	0.0105	0.1	< 0.1	< 0.1	1.5	0.1	< 0.1
	PCB Congener 126	whole body	26	19	73	1.4	0.022	0.0315	0.1	0.1430	4.5	0.2	0.1	20	0.7	0.5
	PCB Congener 169	whole body	26	7	27	0.2	0.00047	0.0001	0.001	0.0002	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1
	Total TEQ							**0.04**		**0.27**	**5.0**	**0.2**	**0.1**	**39**	**1.4**	**0.9**
Green Bay Zone 2																
	PCB Congener 77	egg	12	9	75	2.2	0.000018	0.0000	0.05	0.1105	< 0.1	< 0.1	< 0.1	16	0.6	0.4
	PCB Congener 105	egg	12	12	100	558	0.0000076	0.0042	0.0001	0.0558	0.6	< 0.1	< 0.1	8.0	0.3	0.2
	PCB Congener 118/106	egg	12	12	100	1,414	0.00000037	0.0005	0.00001	0.0141	0.1	< 0.1	< 0.1	2.0	0.1	< 0.1
	PCB Congener 126	egg	12	11	92	2.3	0.022	0.0506	0.1	0.2300	7.2	0.3	0.2	33	1.2	0.7
	PCB Congener 169	egg	12	5	42	0.4	0.00047	0.0002	0.001	0.0004	< 0.1	< 0.1	< 0.1	0.1	< 0.1	< 0.1
	Total TEQ							**0.06**		**0.41**	**7.9**	**0.3**	**0.2**	**59**	**2.2**	**1.3**
Green Bay Zone 3B																
	PCB Congener 77	whole body	16	6	38	0.7	0.000018	0.0000	0.05	0.0330	< 0.1	< 0.1	< 0.1	4.7	0.2	0.1
	PCB Congener 105	whole body	16	16	100	209	0.0000076	0.0016	0.0001	0.0209	0.2	< 0.1	< 0.1	3.0	0.1	0.1
	PCB Congener 118/106	whole body	16	16	100	594	0.00000037	0.0002	0.00001	0.0059	< 0.1	< 0.1	< 0.1	0.8	< 0.1	< 0.1
	PCB Congener 126	whole body	16	13	81	1.4	0.022	0.0297	0.1	0.1350	4.2	0.2	0.1	19	0.7	0.4
	PCB Congener 169	whole body	16	5	31	0.1	0.00047	0.0001	0.001	0.0001	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1
	Total TEQ							**0.03**		**0.19**	**4.5**	**0.2**	**0.1**	**28**	**1.0**	**0.6**

Table 6-134 Ecological Risk Summary Table

Location	Water Column Invertebrates	Benthic Invertebrates	Benthic Fish	Pelagial Fish	Insectivorous Bird	Piscivorous Bird	Carnivorous Bird	Piscivorous Mammal
Little Lake Butte des Morts	● *Mercury* ○ *PCBs*	● *Lead; Mercury; 2,3,7,8-TCDD; PCBs; DDD; DDT*	○ *PCBs*	○ *PCBs*	○ *PCBs*	○ *Mercury; PCBs*	○ *PCBs*	● *PCBs*
Appleton to Little Rapids	○ *PCBs*	● *Lead; Mercury; PCBs*	○ *PCBs*	○ *PCBs*	NA	○ *Mercury; PCBs*	● *PCBs* ○ *Mercury*	● *PCBs*
Little Rapids to De Pere	● *Mercury*	● *Lead; Mercury; 2,3,7,8-TCDD; PCBs; DDE; DDT*	○ *Mercury; PCBs*	○ *Mercury; PCBs*	NA	○ *Mercury; PCBs*	○ *Mercury; PCBs*	● *PCBs*
Green Bay Zone 1	○ *PCBs*	● *Arsenic; Lead; Mercury; PCBs; DDD; DDE*	○ *PCBs; DDE*	○ *Mercury; PCBs; DDE*	○ *PCBs*	○ *Mercury; PCBs; Dieldrin; DDE*	○ *Mercury; PCBs; DDE*	● *PCBs*
Green Bay Zone 2	● *Mercury*	● *Mercury; PCBs*			○ *PCBs; DDE*			
Green Bay Zone 3A		● *PCBs*	○ *PCBs*	○ *PCBs*	NA	○ *Mercury; PCBs*	● *PCBs* ○ *Dieldrin*	● *PCBs* ○ *Dieldrin*
Green Bay Zone 3B		● *Arsenic; Lead; Mercury; PCBs*		● *PCBs* ○ *Mercury; DDE*	NA	● *PCBs* ○ *Mercury; Dieldrin; DDE*	○ *Mercury; PCBs; DDE*	● *PCBs* ○ *Dieldrin*
Green Bay Zone 4		● *PCBs*	NA	○ *PCBs; DDE*	NA	○ *Mercury; PCBs*	○ *Mercury; PCBs; DDE*	● *PCBs*

Note:
NA - No data available.

Risk Conclusions Based on Hazard Quotients:

- No risk.
● - Risk.
○ - Potential Risk.

Risk Conclusions Based on Weight of Evidence:

- Site-specific receptor data suggest that there is no risk.
- Because of the federal listing of the bald eagle as threatened, it is concluded that potential risk is actual risk.

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7 Sediment Quality Thresholds

The overall objective of the Fox River RI/FS is to evaluate corrective actions that may be applied to contaminated sediment within the Lower Fox River and Green Bay. Those corrective actions will be based on the projected reductions of risk to human health and the environment. To that end, the BLRA in Sections 5 and 6 defined the current (or baseline) human health and ecological risks associated with the chemicals of concern; PCBs, mercury, and DDE. Of those, PCBs were identified as the principal component of risk to human health and the environment. To facilitate the selection of a remedy that will result in a decrease in those risks, it is necessary to establish a link between levels of PCBs toxic to human and ecological receptors, and the principal source of those PCBs, the Lower Fox River and Green Bay sediment.

The final chapter role of the risk assessment for the Lower Fox River and Green Bay provides that link between risk in human, birds, mammals and fish by estimating safe thresholds of PCBs in sediment. This section details the methods by which safe thresholds in sediment can be determined. Mathematical bioaccumulation models are used to estimate threshold concentrations of PCBs in sediments that, below which, risks should not occur for the intended receptors. Called sediment quality thresholds (SQTs), these numeric and site-specific values are developed for each pathway and receptor identified as important by the response agencies of the Lower Fox River and Green Bay (e.g., sport fishing consumption, bald eagles). The SQTs themselves are not cleanup criteria, but are a good approximation of protective sediment values and can be considered to be "working values" from which to select a remedial action level. SQTs are used to evaluate harmful levels of contaminants that must be addressed, what levels of those chemicals can be safely left behind, and which remedial option offers the best risk reduction. From the array of PCB-SQTs for specific human health and ecological receptors, the response agencies can evaluate risk reduction and select cleanup standards, or remedial action levels, as a part of a feasibility study. The final selection of the remedial action levels carried forward in the FS is a policy decision left to the response agencies, and the array of PCB-SQTs are principal components of justifying these action levels.

Bioaccumulation modeling is an established part of cleanup programs in the Great Lakes (Pelka, 1998). The *Work Plan for Data Management, Remedial Investigation/ Feasibility Study, and Baseline Human Health and Ecological Risk Assessment for the Lower Fox River and Green Bay* identified the use of dynamic food web modeling (the FRFood Model) to establish risk relationships between sediment

concentrations of nonionic organic chemicals and concentrations of those compounds in fish tissue.

The objective of this section then is to develop that array of PCB-SQTs by:

- Estimating PCB-SQTs that would not result in accumulations to fish tissues at levels that exceed acceptable human health risk levels (cancer risks of 10^{-4}, 10^{-5}, and 10^{-6}, and noncancer risk of a hazard index equal to 1.0; and

- Estimating PCB-SQTs that would not result in unacceptable risks to ecological receptors (e.g., NOAEC, LOAEC).

7.1 Food Web Models for the Lower Fox River and Green Bay

For the overall Remedial Investigation and Feasibility Study, computer models have been developed to assist in the selection of cleanup action levels for PCBs, and for the evaluation of PCB fate and transport into the future. These include:

- The *Whole Lower Fox River Model* (wLFRM) is used to simulate the fate and transport of PCBs in the water and sediments in the Fox River.

- The *Green Bay Toxics* model (GBTOXe) simulates the fate and transport of PCBs in water and sediment in Green Bay.

- The *Fox River Food* (FRFood) is used to estimate PCB concentrations in the food webs leading to forage fish (e.g., shiners, gizzard shad, alewife), benthic fish (e.g., carp), and game fish (perch, walleye) in the river and lower Green Bay.

- The *Green Bay Food* (GBFood) bioaccumulation model receives input data from both wLFRM and GBTOXe and is used to estimate PCB concentrations in the food webs leading to brown trout and walleye Green Bay.

A complete description of all the models used in the RI/FS is given in the companion document *Model Documentation Technical Report for the Lower Fox River and Green Bay* (RETEC, 2002c). The rest of this section focuses on the bioaccumulation models used to develop the Sediment Quality Thresholds.

Numerous aquatic food chain bioaccumulation models have been developed to estimate transfer of hydrophobic contaminants from sediment and water to aquatic biota. The simplest of these models are the ratios of observed concentrations of contaminants in target organisms to observed concentrations in sediment or water: bioaccumulation factors (BAFs), or biota/sediment accumulation factors (BSAFs). While simple in their approach, BSAFs have been shown to provide reasonable accuracy in the prediction of fish tissue concentrations in areas where sufficient data are available (Boese and Lee, 1992). BSAFs have been used to establish cleanup goals for Saginaw River, Michigan and Manistique Harbor, Michigan (Pelka, 1998). However, BSAFs are limited because they are area-specific to the system and organisms, they cannot be used to predict contaminant uptake and distribution through the food chain, and they have limited ability to predict fish concentrations under future conditions.

Uptake models that predict the movement of contaminants from sediments into and through a given food web are often termed bioenergetic models (Boese and Lee, 1992). As compared to BSAFs, bioenergetic models are more mathematically sophisticated, require a greater understanding of the system ecology, and when constructed properly, these models can accurately predict contaminant distribution (Pelka, 1998). Examples of these models include the bioconcentration models of Veith *et al.* (1979) and Gobas (1993), the bioaccumulation models by Thomann (1989) and Thomann and Connolly (1984), the biomagnification models by Bierman (1990) and Clarke and McFarland (1991), and the fugacity-based model by Campfens and Mackay (1997).

For the Lower Fox River and Green Bay, two models have been developed for use in the RI/FS: the Fox River Food (FRFood) and Green Bay Food (GB Food) web. These are discussed in more detail below.

7.1.1 FRFood Model

The FRFood model was developed based on the algorithms of the Gobas model (1993). FRFood is used in the RI/FS to model PCB concentrations in fish within the Lower Fox River and lower Green Bay (zones 1 and 2), and to develop the PCB-SQTs. The Gobas model was selected for several reasons including:

- The model was developed for Great Lakes food chains and has been previously validated using both Lake Ontario and Green Bay PCB and food web data.

- EPA made extensive use of the Gobas model to derive bioaccumulation factors, bioconcentration factors, and food chain multipliers in the

development of the Great Lakes Water Quality Initiative (GLWQI) criteria (EPA, 1993b, 1994a).

- The Gobas model was used in the 1996 RI/FS for the Lower Fox River and found to yield reasonably good results between predicted and measured fish tissue PCB concentrations (GAS/SAIC, 1996).

- A modified version of the Gobas model was used for the Ecological Risk Assessment for the Sheboygan River, Wisconsin, and also found reasonable similarity between predicted and measured PCB levels in fish (EVS, 1998)

- The Gobas algorithms were used to project future PCB concentrations in fish for the Hudson River (EPA, 2000a).

The Gobas model has seen the most widespread use in the Great Lakes area. In 1993, Gobas introduced his methods by modeling a food web in Lake Ontario. He compared predicted levels of PCBs in a Lake Ontario food web to published observed data (Oliver and Niimi, 1988), and found that predicted versus observed PCB concentrations were within a factor of five for all organisms. The model was particularly accurate in determining PCB levels in higher trophic levels (all fish), where predicted levels of PCBs versus observed differed by less than a factor of two.

Both the Gobas model (1993), and a similar model constructed by Thomann (1989) and Thomann *et al.* (1992) have gained general scientific acceptance and are now being used in scientific and regulatory applications to predict concentrations of hydrophobic organic contaminants in aquatic food webs (Burkhard, 1998). Burkhard (1998) recently reviewed the predictive capabilities of these two models compared to field-collected fish data from Lake Ontario and concluded that the Gobas model provided slightly better predictions.

While the Gobas model was developed specifically for application in lake systems, the mathematical relationships have been successfully applied to predicting fish tissue concentrations in some river systems. As noted above, the 1996 RI/FS for the Fox River found good correlation between predicted and observed fish tissue concentrations. Likewise, a good fit between predicted and observed fish tissue concentration was observed when the model was used to describe the bioaccumulation of PCBs in Hudson River ecosystems (EPA, 2000a), and the Sheboygan River (EVS, 1998). In part, this may be because the lock and dam system on the Fox and Hudson rivers creates a series of large "pools" that behave more like reservoir or lake-like systems (e.g., Little Lake Butte des Morts).

The Gobas model assumes that equilibrium steady states exist between water and plankton, and between sediment and benthic invertebrates. Lipid-normalized phytoplankton and zooplankton concentrations are assumed to equal organic carbon-normalized water concentrations. Lipid-normalized benthic invertebrate concentrations are estimated to equal organic carbon-normalized sediment concentrations. Non-equilibrium steady-state concentrations in fish are calculated assuming mass balance where contaminant uptake from diet and gill ventilation is equal to loss through gill ventilation, egestion, metabolic breakdown, and dilution by growth.

Since 1993, several improvements/additions to the Gobas model have been suggested, including a time-dependent response to changes in PCB levels which incorporated age classes to organisms (Gobas *et al.*, 1995) and a more sophisticated model to describe bioaccumulation of PCBs in zooplankton and benthic invertebrates (Morrison *et al.*, 1996). Morrison *et al.* (1996) improved modeled zooplankton and benthic invertebrate bioaccumulation by considering PCB intake from diet (by filter feeding and consumption of detritus) and gill ventilation, and loss through gill ventilation, egestion, metabolic breakdown, and dilution by growth. A verification of an entire aquatic food web using the 1993 Gobas model and improved zooplankton and benthic invertebrate model was published in 1997 (Morrison *et al.*, 1997). All verification attempts found that estimated concentrations of PCBs typically fell well within an order of magnitude of observed results. However, these modifications were not incorporated into FRFood due to: 1) the lack of site-specific input parameters necessary to implement those modifications, and 2) the generally good agreement between predicted and observed PCB fish tissue concentrations for FRFood.

A discussion of the selection, development, calibration, validation, and application of the FRFood Model is provided in the Model Description Memorandum.

7.1.2 GBFood Model

The GBFood bioaccumulation model is a mathematical description of contaminant transfer within the Green Bay food web. The food web is comprised of the primary energy transfer pathways from the exposure sources (sediment and water) to the fish species of interest. These pathways include: chemical uptake across the gill surface, chemical uptake from food and chemical losses due to excretion and growth dilution. The mathematical descriptions are generic (common to all aquatic food webs) and were updated as part of this RI/FS.

GBFood is based on the work of Connolly *et al.* (1992) which incorporated algorithms from Thomann (1989) and Thomann *et al.* (1992). GBFood will be used in the FS to estimate fish tissue concentrations based on 100-year projected

sediment concentrations for different remedial alternatives. GBFood is not designed to estimate sediment PCB concentrations from fish tissue concentrations, and thus is not being used to develop PCB-SQTs. A description of the GBFood Model dietary assignments and model validation are contained in the Model Documentation Technical Report (RETEC, 2002c)

7.2 FRFood Model Food Web Review and Dietary Assignments

FRFood is constructed from the mathematical relationships between sediment, water, phytoplankton, zooplankton, and contaminant transfer factors to prey and predatory fish that were originally defined by Gobas (1993). The construct of the model, the input parameters, and the application of the model are documented in the *FRFood Users Guide* (RETEC, 2002d).

As note above, the Gobas algorithms were selected to develop the FRFood model in part because of the accuracy observed in predicting fish tissue concentrations in the 1996 RI/FS for the river above the De Pere dam (GAS/SAIC, 1996). While the 1996 food web model provided a reasonable degree of accuracy in predicting fish tissue concentrations, it was necessary to re-examine the food web relationships for use in the FRFood Model because the 1996 food web does not accurately reflect predator/prey relationships in the river and Green Bay.

A key assumption of the previous RA for the Lower Fox River was that the food web was principally based on sediment-dwelling insects (GAS/SAIC, 1996). In 1996, the benthic invertebrates selected for modeling included oligochaetes and chironomids, based upon their predominance in previous benthic analyses done within the Fox River system (WDNR, 1993), and on the work by Call *et al*. (1991), and the mayfly *Hexagenia*, based upon mayfly presence in both the reference sites for the WDNR (1993) study and the Call *et al.* (1991) study.

In the 1996 Lower Fox River bioaccumulation model, carp was selected as the benthic fish species for the model based upon the fact that it is the dominant benthic feeding fish found within the Lower Fox River system. In addition, carp PCB body burden data were measured as part of the mass balance study, and available information concerning size, lipid content, and diet were reviewed. Carp were assigned oligochaetes and chironomids as principle forage, but also assigned a smaller fraction of mayflies and zooplankton. Walleye were selected as the top piscivorous species for the model, based upon relative abundance, their importance to Lower Fox River anglers, and availability of data for modeling. A second key assumption of the 1996 model was that yellow perch are the preferred prey species for walleye (Ney, 1978; Ryder and Kerr, 1978). In the 1996 model,

yellow perch fed predominantly on benthic invertebrates (Ney, 1978), while walleye fed principally on yellow perch, small carp, and a smaller fraction of emergent *Hexagenia* larvae.

The FRFood Model was designed to accurately reflect food web interactions using information on receptors in the river and Green Bay. Two food web models were used to describe the food web in the Lower Fox River and southern Green Bay: one for above the De Pere dam and one for below the dam (Green Bay zones 1 and 2). The revised food webs were discussed and presented in Section 6 (see Figures 6-1 through 6-3). Selection and documentation of the important food webs for all of the Fox River and Green Bay are given in WDNR Technical Memorandum 7c (WDNR, 2001). The principal changes from the 1996 food web model is the shift from a primarily benthic-based food chain to a food web that equally includes both benthic and pelagic uptake routes. In addition, other fish species (e.g., alewife, gizzard shad) and year classes for yellow perch, alewife, and carp were added. An additional change to the Lower Fox River food web was the exclusion of *Hexagenia*, as it is generally not found in the Lower Fox River and Green Bay (WDNR, 1995).

Once the food webs were identified, a literature search was conducted to develop a range of values for diet composition (species and percent prey based on weight or volume of prey), weight, and lipid content. The range of values are presented in Table 7-1.

7.3 FRFood Model Calibration

The calibration for the FRFood Model was run using site-specific total PCB data for sediment and water as well as site-specific dietary relationships and lipids. Total PCB-SQTs were estimated for the following reasons: 1) total PCBs are used in the risk assessment to encompass all observed toxicity, including that from the dioxin-like coplanar congers as well non-coplanar PCB molecules; 2) transfer factors for specific PCB co-planar congeners between the various media (sediment, pore water, surface water, phytoplankton, zooplankton, prey fish, predator fish, birds, humans) are not well supported in the FRDB or scientific literature; and 3) remedial actions have been based to date on total PCBs, and not congener-specific cleanup levels (e.g., Deposit N, SMU 56/57 demonstration projects).

Calibration of FRFood was based upon comparing predicted versus actual fish tissue PCB concentrations, and is discussed in detail in the FRFood Model Documentation Memorandum (RETEC, 2002e), and in the *FRFood User's Guide* (RETEC, 2002d). Generally, sediment and water concentrations derived from the FRDB (discussed in Section 6.4) were used as inputs to the model for each reach. Dietary inputs for the food web species were generally based on average

consumption, but modified as necessary for calibration purposes within the range of parameters specified in Table 7-1. Lipid concentrations for fish were also treated as a calibration variable. In general, the arithmetic average concentration on a reach-specific basis for each species selected. FRFood Model output was then compared to actual measured fish concentrations from Little Lake Butte des Morts, Little Rapids to De Pere, De Pere to Green Bay (Green Bay Zone 1), and Green Bay Zone 2. There were only sufficient data for these four areas to check the model.

The model evaluation metrics that were used to determine if the FRFood Model was an effective tool for estimating PCB-SQTs for the FS were those used in the Green Bay Mass Balance Study and agreed upon by the WDNR in cooperation with the Fox River Group of companies (Limno-Tech, 1998). The goals are to achieve agreement of ±30 percent between model predictions and observations for water and sediment, and plus or minus one-half order of magnitude for fish. Input parameters, both physical and dietary, for each species and each of the areas are presented in Tables 7-2 through 7-5.

Sediment-weighted average concentrations (SWAC) were used as input to the FRFood. The surface sediment interpolated total PCB concentrations (I_d) from the bed maps (see Section 2.3) were selected over non-interpolated total PCB sediment concentrations (average or 95th UCL), because between river reaches, the spatial degree of PCB analysis conducted on sediment in each area varied. Using the surface SWAC normalized total PCB concentrations between river reaches.

PCB concentrations inputs for water were based upon the filtered fraction of water samples collected, and reported in the FRDB. The filtered fraction represents the PCB fraction that is available for uptake; i.e., not bound up with the particulate or organic (i.e., phytoplankton or zooplankton) fractions in the water column. Using the filtered water as an input ensured that the phytoplankton/zooplankton component was not counted twice in the model calibration. Details of this analysis are covered in the FRFood Model Documentation Memorandum (RETEC, 2002e).

The comparison of FRFood Model output to the mean and 95% UCL whole fish tissue concentrations collected by reach are shown in Table 7-6. The starting sediment and water concentrations are boxed and bolded. Calibration of the FRFood Model indicated that all predicted fish tissue concentrations were within one-half order of magnitude of observed concentrations of total PCBs, except for yellow perch in the Little Lake Butte des Morts Reach. However, within this reach data were only available for one fish. All other predicted fish concentrations were within a factor of two compared to the observed tissue concentrations of

total PCBs, except for common and emerald shiners in Green Bay Zone 1. Importantly, the predicted shiner concentrations in this zone were only 14 percent more than the measured concentrations in golden shiner. Based upon these observed/predicted results compared to the model evaluation metrics, the Lower Fox River bioaccumulation model is judged suitable for use in estimating PCB-SQT concentrations within the Lower Fox River. These results indicate that the FRFood model meets the metrics goal of achieving agreement in predicted and observed fish tissue concentrations to within plus or minus one-half order of magnitude for fish.

7.4 Determination of Sediment Quality Thresholds

7.4.1 Estimating Sediment-to-water Ratios

To calculate a PCB-SQT from a fish tissue concentration, it was necessary to identify a generalized term relating the concentration of total PCBs in filtered water relative to that found in the sediments. The same water and sediment data used to calibrate the mass balance for the Fox River were used to estimate this term. These data are shown in Table 7-7, and represent the minimum, maximum, and average values computed for 1989 through 1990 calibration period. For the Lower Fox River, the data suggest that the non-particulate water PCB concentration is between 10^{-6} and 10^{-7} of the bedded sediment concentration. For the De Pere to Green Bay Reach (Zone 1), the value lies between 10^{-4} and 10^{-6}. As a general term for developing the river SQTs, a value of 10^{-6} was used to estimate SQTs.

The estimated sediment-to-water ratios for Zone 2 is complicated by the fact that approximately 70 percent of the water in Zone 2 (Long Tail Point to Point Sable) is comprised of water from the Lower Fox River (Brazner and Beals, 1997). To estimate the sediment/water resuspension rates for PCBs, the GBTOX mass balance model was run using zero PCB loading from the Lower Fox River. Given no loads from the Fox River, the average water column concentrations ranged between 10^{-7} to 10^{-5} of the interpolated sediment concentrations. Given these estimates, a 10^{-6} term is also applicable to Zone 2 sediments.

Because of the uncertainty associated with the sediment-to-water ratio, SQTs may differ by an order of magnitude. For example, walleye NOAEC SQTs based on a sediment-to-water ratio of 10^{-5} are eight times less than an SQTs based on a sediment-to-water ratio of 10^{-6} and 25 times less than an SQT based on a sediment-to-water ration of 10^{-7}.

7.4.2 Human Health Sediment Quality Thresholds

Human health PCB-SQTs were developed for recreational anglers and high-intake fish consumers at both the 10^{-5} risk level and at a hazard index of 1.0 for walleye, perch, and carp. SQTs were estimated for reasonable maximum exposure and the central tendency exposure scenarios. SQTs associated with cancer risk levels of 10^{-4} and 10^{-6} are one order of magnitude below, and one order of magnitude higher than the SQTs for the 10^{-5} risk level.

To estimate the human health PCB-SQT, risk-based fish concentrations (RBFCs) were developed for PCBs in fish fillets (see Section 5.9.9). Since these RBFCs are expressed as concentrations of PCBs in fillets, it was necessary to convert RBFCs for the fish fillet to RBFCs for whole body fish. Based on data obtained from the literature, the ratio of PCB concentrations in fillet to whole body can be estimated:

$$C_{fish-f} = a_{f-wb} \cdot C_{fish-wb}$$

where:

C_{fish-f} = concentration of PCBs in fish fillet (μg/kg-fillet),
a_{f-wb} = ratio of concentrations in fish fillet to concentrations in whole body of fish (kg-fish/kg-fillets), and
$C_{fish-wb}$ = concentration of PCBs in whole body of fish (μg/kg-whole body).

Once whole body RBFCs for total PCBs were obtained, these concentrations were used as inputs to the FRFood Model, which then output PCB concentrations in sediment that represent PCB-SQTs.

To calculate fillet-to-whole body ratios, both site-specific data and literature-derived ratios were examined. Table 7-8 summarizes ratios of PCB concentrations for fillet and whole body for a number of different fish species. For the Lower Fox River, data were available in the FRDB to estimate fillet-to-whole body ratios for walleye (0.17), carp (0.53), white bass (0.44), and white sucker (0.48). For perch, there were insufficient data to estimate a ratio specific to perch, but the walleye ratio was deemed applicable. Perch are from the same family as walleye (*Percidae*) and have similar lipid values. Table 7-8 also presents the ratios from other studies. The ratios range from 0.04 for perch to 1.0 for brown trout. The perch value of 0.04 from Parkerton (1993) for fish collected at Lake Erie and the data used to develop this ratio were not available for review. Thus, the perch value of 0.04 was not used. There is variability within the same species, with ratios ranging from 0.57 to 1.0 for brown trout; 0.59 to 0.89 for coho salmon; 0.34 to 0.68 for rainbow trout; and 0.09 to 0.17 for walleye.

Table 7-9 presents the PCB-SQTs associated with a risk level of 10^{-5} and a hazard index of 1.0 for carp, walleye, and perch for the Lower Fox River. These values ranged between 11 μg/kg-sediment PCBs for the high-intake fish consumer eating carp under an RME scenario, to 1,128 μg/kg for a recreational angler eating perch under a CTE scenario. It is important to note that Table 7-9 presents the SQTs associated with a target rate of 10^{-5}; the SQTs associated with cancer ratios of 10^{-6} and 10^{-4} are an order of magnitude lower, or higher, respectively. All three ranges of cancer risks are carried forward into the Feasibility Study to be evaluated as part of the action level selection process, and for the evaluation of remedial alternatives.

7.4.3 Ecological Sediment Quality Thresholds

Total PCB-SQTs protective of ecological receptors were derived from the toxicity reference values listed in Table 6-5 of the ecological risk assessment. The total PCB fish Toxicity Reference Value (TRV) for the various receptors were used as inputs to the FRFood Model, and then back-calculated to yield the PCB-SQT. Total PCB-SQTs were directly derived from the TRVs for fish survival and reproduction and for mink reproduction and kit survival based upon total PCB concentrations in fish as part of their diet. The fish species selected for PCB-SQT determinations were walleye and carp, because they are the highest trophic level pelagic and benthic fish present in the river. Sediment quality thresholds that are protective of walleye and carp should also be protective of other fish species present.

For piscivorous and carnivorous birds, TRVs were based on egg or whole body concentrations. Therefore, it was necessary to derive site-specific biomagnification factors (BMFs) to determine what were safe concentrations in fish, their sole or primary prey. For bald eagles, carp were assumed to be the primary prey, and for both tern species and double-crested cormorants, alewife were assumed to be the primary prey. Total PCB concentrations in these bird species (egg or whole body) were compared to primary prey concentrations within the same reach to derive species-specific BMFs. The BMF was calculated by dividing the bird receptor egg or whole body concentration by the fish concentration. To facilitate the calculation of the BMF, it was conservatively assumed that the diet of these bird species was 100 percent alewife, and that all of the PCBs are transferred from fish to eggs. These BMFs were then applied to the total PCB TRVs for birds in order to convert these bird tissue TRVs into fish tissue TRVs. While limitations of the BMF model were discussed previously, there are no kinetic bioaccumulation models that have been validated for fish-to-bird contaminant transfers. The BMF model, used with site-specific data and within this context, is the best approximation of bird contaminant exposure. BMFs and estimated threshold fish

tissue concentrations for effects to reproduction and embryo physiology are given in Table 7-10.

Total PCB sediment quality thresholds for fish, birds, and mink are given in Table 7-11. The PCB-SQTs range from a low of 24 μg/kg that is protective of mink reproduction and kit survival, to a high of 5,231 μg/kg that corresponds to a LOAEC for common tern deformity.

7.5 Section 7 Tables

Section 7 tables follow this page and include:

Table 7-1 References Reviewed for Potential Input Parameter to the Lower Fox River Bioaccumulation Model

Organisms	Dietary Composition (based on weight or volume)	Whole Fish Lipid Content (%)	Weight (kg)
Plankton			
Zooplankton		5 (Gobas, 1993)	0 (Campfens and Mackay, 1997)
Benthic Organisms			
Oligochaetes		1 (Campfens and Mackay, 1997)	0.0001 (Campfens and Mackay, 1997)
Chironomids		2 (Zaranko *et al.* , 1997)	
Fish			
Rainbow Smelt	25%–100% zooplankton, 0%–25% alewife (Mills *et al.* , 1995; Price, 1963)	1.7–9.8 (site-specific data)	0.085 (Seagrant web page)
Gizzard Shad	10%–70% zooplankton, 10%–90% algae, 10% benthic invertebrates (Muth and Busch, 1989; Kolok *et al.* , 1996; Exponent, 1999)	2.5–19.0 (site-specific data)	0.025 (Levine *et al.* , 1995)
Emerald Shiner	90% zooplankton, 5% algae, 5% chironomids (Muth and Busch, 1989)	5.1–6.2 (site-specific data)	
Carp			
YOY[1]	14%–100% benthic invertebrates, 10%–60% plankton (Weber and Otis, 1984; Exponent, 1999)		0.00629 (Weber and Otis, 1984)
adults	14%–100% benthic invertebrates, 25%–45% plankton (Scott and Crossman, 1973)	0.8–25.4 (site-specific data)	1.4–6.8 (Scott and Crossman, 1973)
Alewife			
YOY	20%–90% copepods, 10%–80% cladocerans (Hewett and Stewart, 1989; Urban and Brandt, 1993)		avg. = 0.00071 (Flath and Diana, 1985)
adults	25%–93% plankton, 7%–20% benthic invertebrates (Gobas *et al.* , 1995; Hewett and Stewart, 1989; Exponent, 1999)	2.5–17.0 (site-specific data)	0.056 $\pm$ 0.007 (Hewett and Stewart, 1989)
Perch			
YOY and adults	40%–100% benthic invertebrates, 60% plankton (Scott and Crossman, 1973; Weber and Otis, 1984; Exponent, 1999; Carlander, 1997a)	2.2–6.1 (site-specific data)	0.01–0.588 (Wells and Jorgenson, 1983)
Walleye			
YOY	0%–96% rainbow smelt, 0%–78% gizzard shad, 0%–20% emerald shiner, 0%–80% white perch, 0%–29% yellow perch, 0%–28% white sucker, 0%–24% benthic invertebrates (Wolfert and Bur, 1992; Exponent, 1999; Carlander, 1997b)		0.04 (Magnuson and Smith, 1987)
adults	10% plankton, 14%–24% benthic invertebrates, 12%–100% alewife, 0%–76% rainbow smelt, 0%–74% gizzard shad, 0%–1% sculpin, 0%–38% white sucker, 0%–44% yellow perch, 0%–23% small mouth bass (Magnuson and Smith, 1987; Wolfert and Bur, 1992)	0.4–23.2 (site-specific data)	2.3 (site-specific data)

Note:

[1] YOY - Young-of-the-year.

Table 7-2 Inputs to the FRFood Model for Model Calibration in Little Lake Butte des Morts Reach

A. Diet

Prey	Receptors							
	Shiner Species Muth & Busch, 1989	Gizzard Shad Muth & Busch, 1989; Kolok *et al.* ,	Yellow Perch YOY Carlander, 1997a; Scott & Crossman,	Yellow Perch Adult Carlander, 1997a	Carp YOY Weber & Otis, 1984	Carp Adult Scott & Crossman, 1973	Walleye YOY Carlander, 1997b; Wolfert & Bur,	Walleye Adult Wolfert & Bur, 1992; Magnuson & Smith, 1987
Phytoplankton	0.7	1		0.3	0.3			0.1
Zooplankton	0.2		0.9	0.4	0.4	0.45	0.05	
Chironomids	0.1		0.1	0.3	0.3	0.35	0.1	0.2
Oligochaetes						0.2		
Emerald Shiner							0.4	0.25
Gizzard Shad							0.45	0.45

B. Lipid Concentrations

Lipids (%)	Receptor							
	Shiner Species	Gizzard Shad	Yellow Perch YOY	Yellow Perch Adult	Carp YOY	Carp Adult	Walleye YOY	Walleye Adult
Lipid Used in Model	5.4	12.0	4.4	4.4	7.6	7.6	7.3	7.3
Mean Lipids for this	5.4	12.0		4.4		7.6		7.3
Mean Lipids over All	5.6	7.3		3.4		10.1		9.7

C. Sediment and Water Total PCB Concentrations

Media	Mean (ppb)	95% UCL (ppb)	Average TOC (%)
Water (filtered)	0.011	0.015	
Sediment (I_d)	3,699	3,749	14

Table 7-3 Inputs to the FRFood Model for Model Calibration in Little Rapids to De Pere Reach

A. Diet

Prey	Receptors							
	Shiner Species Muth & Busch, 1989	Gizzard Shad Muth & Busch, 1989; Kolok *et al.* ,	Yellow Perch YOY Carlander, 1997a; Scott & Crossman,	Yellow Perch Adult Carlander, 1997a	Carp YOY Weber & Otis, 1984	Carp Adult Scott & Crossman, 1973	Walleye YOY Carlander, 1997b; Wolfert & Bur,	Walleye Adult Wolfert & Bur, 1992; Magnuson & Smith,
Phytoplankton	0.7	0.7		0.3	0.3			0.1
Zooplankton	0.2	0.3	0.9	0.4	0.4	0.45	0.05	
Chironomids	0.1		0.1	0.3	0.3	0.35	0.1	0.2
Oligochaetes						0.2		
Emerald Shiner							0.4	0.25
Gizzard Shad							0.45	0.45

B. Lipid Concentrations

Lipids (%)	Receptor							
	Shiner Species	Gizzard Shad	Yellow Perch YOY	Yellow Perch Adult	Carp YOY	Carp Adult	Walleye YOY	Walleye Adult
Lipid Used in Model	7.0	2.8	2.2	2.2	6.9	6.9	8.1	8.1
Mean Lipids for this	7.0	2.8		2.2		6.9		8.1
Mean Lipids over All	5.6	7.3		3.4		10.1		9.7

C. Sediment and Water Total PCB Concentrations

Media	Mean (ppb)	95% UCL (ppb)	Average TOC (%)
Water (filtered)	0.011	0.012	
Sediment (I_d)	2,078	2,112	5.3

Table 7-4 Inputs to the FRFood Model for Model Calibration in Green Bay Zone 1

A. Diet

Prey	Receptors										
	Rainbow Smelt Mills *et al.*, 1995	Gizzard Shad * Muth & Busch, 1989;	Shiner Species Muth & Busch, 1989	Alewife YOY Hewett & Stewart, 1989;	Alewife Adult Hewett & Stewart, 1989	Yellow Perch YOY Carlander, 1997a; Scott & Crossman,	Yellow Perch Adult Carlander, 1997a	Carp YOY Weber & Otis, 1984	Carp Adult Scott & Crossman, 1973	Walleye YOY Carlander, 1997b; Wolfert & Bur,	Walleye Adult Wolfert & Bur, 1992; Magnuson & Smith,
Phytoplankton		0.3	0.6				0.3	0.3			
Zooplankton	0.9	0.6	0.3	1	0.95	0.9	0.4	0.4	0.45	0.05	
Chironomids		0.1	0.1		0.05	0.1	0.3	0.3	0.35	0.3	0.1
Oligochaetes									0.2		
Yellow Perch YOY											
Alewife YOY	0.1									0.15	
Alewife adult											0.1
Rainbow Smelt										0.1	0.1
Emerald Shiner											
Gizzard Shad										0.4	0.7

B. Lipid Concentrations

Prey	Receptor										
	Rainbow Smelt	Gizzard Shad	Shiner Species	Alewife YOY	Alewife Adult	Yellow Perch YOY	Yellow Perch Adult	Carp YOY	Carp Adult	Walleye YOY	Walleye Adult
Lipid Used in Model	4.6 *	7.1	6	5.7	5.7	4.5	4.5	9.2	9.2	10.7	10.7
Mean Lipids for this	4.6 *	7.1	5.6/6.1		5.7		4.5		9.2		10.7
Mean Lipids over All	4.6	7.3	5.6		8.6		3.4		10.1		9.7

Note:

* Zone 2 average; rainbow smelt were not caught in Zone 1.

C. Sediment and Water Total PCB Concentrations

Media	Mean (ppb)	95% UCL (ppb)	Average TOC (%)
Water (filtered)	0.017	0.018	
Sediment (I_d)	2,959	2,984	5

Table 7-5 Inputs to the FRFood Model for Model Calibration in Green Bay Zone 2

A. Diet

Prey	Receptors										
	Rainbow Smelt Mills *et al.*, 1995	Gizzard Shad * Muth & Busch, 1989;	Shiner Species Muth & Busch, 1989	Alewife YOY Hewett & Stewart, 1989;	Alewife Adult Hewett & Stewart, 1989	Yellow Perch YOY Carlander, 1997a; Scott & Crossman,	Yellow Perch Adult Carlander, 1997a	Carp YOY Weber & Otis, 1984	Carp Adult Scott & Crossman, 1973	Walleye YOY Carlander, 1997b; Wolfert & Bur,	Walleye Adult Wolfert & Bur, 1992; Magnuson & Smith,
Phytoplankton		0.3	0.6				0.3	0.3			
Zooplankton	0.9	0.6	0.3	1	0.95	0.9	0.4	0.4	0.45	0.05	
Chironomids		0.1	0.1		0.05	0.1	0.3	0.3	0.35	0.3	0.1
Oligochaetes									0.2		
Yellow Perch YOY											
Alewife YOY	0.1									0.15	
Alewife adult											0.1
Rainbow Smelt										0.1	0.1
Emerald Shiner											
Gizzard Shad										0.4	0.7

B. Lipid Concentrations

Prey	Receptor										
	Rainbow Smelt	Gizzard Shad	Shiner Species	Alewife YOY	Alewife Adult	Yellow Perch YOY	Yellow Perch Adult	Carp YOY	Carp Adult	Walleye YOY	Walleye Adult
Lipid Used in Model	4.6	6.9	6	9.8	9.8	3.2	3.2	11.3	11.3	10.4	10.4
Mean Lipids for this Reach	4.6	6.9	—	—	9.8	—	3.2	—	11.3	—	10.4
Mean Lipids over All Areas	4.6	7.3	5.6		8.6		3.4		10.1		9.7

C. Sediment and Water Total PCB Concentrations

Media	Mean (ppb)	95% UCL (ppb)	Average TOC (%)
Water (filtered)	0.0048	0.0054	
Sediment (I_d)	1,132	1,154	1.5

Table 7-6 Lower Fox River Bioaccumulation Model Calibration

Location	Species	Number of Samples	Number of Detects	Detection Frequency	Observed Total PCB Mean	Observed Total PCB 95% UCL		Predicted Total PCB Mean	Predicted Total PCB 95% UCL	Units
Little Lake Butte des Morts										
	Water (filtered)	46	40	87	**0.011**	**0.015**				µg/L
	Surface Sediments (N)	302	294	97	10,724	22,848				µg/kg
	Surface Sediments (I_0)	57,724	57,724	100	3,284	3,330				µg/kg
	Surface Sediments (I_d)	51,261	51,261	100	**3,699**	**3,749**				µg/kg
	Gizzard Shad	4	4	100	296	530	*	263	358	µg/kg
	Golden Shiner	2	2	100	993	1,140	*	723	868	µg/kg
	Yellow Perch	1	1	100	363	363	*	1,266	1,443	µg/kg
	Carp	30	30	100	1,992	2,957		2,374	2,639	µg/kg
	Walleye	13	11	85	1,159	3,800	*	1,756	2,109	µg/kg
Little Rapids to DePere										
	Water (filtered)	98	97	99	**0.011**	**0.012**				µg/L
	Surface Sediments (N)	209	203	97	4,782	10,543				µg/kg
	Surface Sediments (I_0)	37,490	37,490	100	2,054	2,088				µg/kg
	Surface Sediments (I_d)	37,060	37,060	100	**2,078**	**2,112**				µg/kg
	Gizzard Shad	3	3	100	347	370	*	318	347	ug/kg
	Golden Shiner	2	2	100	1,020	1,036	*	997	1,046	ug/kg
	Yellow Perch	1	1	100	627	627	*	1,017	1,055	µg/kg
	Carp	20	20	100	3,919	5,800		3,038	3,135	µg/kg
	Walleye	4	4	100	3,179	4,587	*	3,881	4,079	µg/kg

Table 7-6 Lower Fox River Bioaccumulation Model Calibration (Continued)

Location	Species	Number of Samples	Number of Detects	Detection Frequency	Observed Total PCB Mean	Observed Total PCB 95% UCL		Predicted Total PCB Mean	Predicted Total PCB 95% UCL	Units
Green Bay Zone 1										
	Water (filtered)	143	142	99	**0.017**	**0.018**				µg/L
	Surface Sediments (N)	290	285	98	4,184	5,510				µg/kg
	Surface Sediments (I_0)	52,115	52,115	100	2,950	2,976				µg/kg
	Surface Sediments (I_d)	51,963	51,963	100	**2,959**	**2,984**				µg/kg
	Alewife	13	13	100	2,596	3,018		1,491	1,566	µg/kg
	Gizzard Shad	18	18	100	2,017	2,369		1,560	1,613	µg/kg
	Common Shiner	5	5	100	3,520	3,846		1,572	1,636	µg/kg
	Emerald Shiner	5	5	100	3,520	3,846		1,572	1,636	µg/kg
	Golden Shiner	2	2	100	1,385	1,443	*	1,572	1,636	µg/kg
	Yellow Perch	5	5	100	1,435	2,005		2,552	2,610	µg/kg
	Carp	66	66	100	7,203	8,286		5,352	5,454	µg/kg
	Walleye	51	51	100	6,902	8,414		9,091	9,419	µg/kg
Green Bay Zone 2										
	Water (filtered)	63	63	100	**0.0048**	**0.0054**				µg/L
	Surface Sediments (N)	15	14	93	251	5,510				µg/kg
	Surface Sediments (I_0)	11,713	11,713	100	1,117	2,976				µg/kg
	Surface Sediments (I_d)	11,566	11,566	100	**1,132**	**2,984**				µg/kg
	Alewife	38	38	100	2,600	3,374		923	992	µg/kg
	Gizzard Shad	32	32	100	1,759	1,906		1,184	1,230	µg/kg
	Rainbow Smelt	33	33	100	1,049	1,152		410	462	µg/kg
	Yellow Perch	4	4	100	920	1,637	*	2,028	2,084	µg/kg
	Carp	49	49	100	5,875	8,914		6,267	6,425	µg/kg
	Walleye	40	40	100	6,076	6,790		6,473	6,750	µg/kg

Notes:

Boxed and bolded values represent sediment inputs to the Lower Fox River bioaccumulation model.

* Maximum concentration and not the 95% UCL.

Table 7-7 Reach-specific and River-wide Total PCB Water-to-Sediment Ratios

Location	Media	Year	Minimum	Maximum	Average
Little Lake Butte des Morts	Sediment	1989	25	130,000	13,535
Little Lake Butte des Morts	Water	1989/90	0.0015	0.0592	0.0276
Water-to-sediment Ratio			6.00E-05	4.55E-07	2.04E-06
Appleton to Little Rapids	Sediment	1989	50	57000	3,651
Appleton to Little Rapids	Water	1989/90	0.00004	0.0710	0.0168
Water-to-sediment Ratio			8.00E-07	1.25E-06	4.60E-06
Little Rapids to De Pere	Sediment	1989	80	33,000	3,873
Little Rapids to De Pere	Water	1989/90	0.0004	0.1240	0.0411
Water-to-sediment Ratio			5.00E-06	3.76E-06	1.06E-05
Green Bay Zone 1	Sediment	1989	20	18,700	2,700
Green Bay Zone 1	Water	1989/90	0.0038	0.1940	0.0609
Water-to-sediment Ratio			1.91E-04	1.04E-05	2.26E-05
Green Bay Zone 2					
Water-to-sediment Ratio		GBTOXe*	5.26E-07	2.43E-05	8.47E-06

Notes:

Water represents the estimated total PCB concentration.

Zone 2 sediment:water ratios estimated from GBTOXe output.

Concentrations in units of ppb.

Table 7-8 Ratio of PCB Concentrations in Fillet to Whole Body for Different Species

Study and Species	Fillet-to-whole Fish Ratio
Lower Fox River	
Walleye	0.17
Carp	0.53*
Perch	0.17
White Bass	0.44
White Sucker	0.48
Parkerton (1993)	
Perch	0.04 *
Walleye	0.1 *
Bevelhimer et al. (1997)	
Black Bass	0.43
Amhreim et al. (1999)	
Coho Salmon	0.59
Rainbow Trout	0.68
Niimi and Oliver (1983)	
Rainbow Trout	0.34
Connolly (1991)	
Flounder	0.18
Connolly et al. (1992)	
Brown Trout	1
Brown Trout	0.88
Brown Trout	0.57
Coho Salmon	0.89
Walleye adult	0.09
Channel Catfish	0.59
Drum	0.32
Perch	0.04

Notes:

CPCB-f - Concentration of PCB in fish fillet.

CPCB-wb - Concentration of PCB in whole body of fish.

* Fillet-to-whole body ratios selected.

Table 7-9 Sediment Quality Thresholds Estimated for Human Health Effects at a 10^{-5} Cancer Risk and Noncancer Hazard Index of 1.0

	Fish Parameters	Sediment Quality Thresholds			
	Fillet-to-whole Fish Ratio	Recreational Anglers: (West *et al.* , 1989; West, 1993)		High-intake Fish Consumers: (West, 1993; Hutchison and Kraft, 1994)	
		RME µg/kg	CTE µg/kg	RME µg/kg	CTE µg/kg
Sediment Quality Thresholds for Risk of 10^{-5} *					
Carp	0.53	***16***	***180***	***11***	***57***
Walleye	0.17	***21***	***143***	***14***	***75***
Yellow Perch	0.17	***105***	***677***	***68***	***356***
Sediment Quality Thresholds for HI of 1.0					
Carp	0.53	***44***	***180***	***28***	***90***
Walleye	0.17	***58***	***238***	***37***	***119***
Yellow Perch	0.17	***276***	***1,128***	***175***	***564***

Notes:

* ***SQTs for cancer risks of 10^{-4} and 10^{-6} are an order of magnitude higher, and lower, respectively.***

RME indicates reasonable maximum exposure and CTE indicates central tendency exposure.

Sediment Quality Thresholds are **bolded** and in *italics* .

Table 7-10 Derivation of Bird Biomagnification Factors (BMFs) for Total PCBs

Location	Bird		Total PCB (µg/kg)	Fish		Total PCB (µg/kg)	BMF
	Species	Tissue	RME	Species	Tissue	RME	RME
Appleton to Little Rapids	Bald Eagle	egg	36,000	carp	whole	3,606	9.98
Zone 2	Double-crested Cormorant	egg	21,127	alewife	whole	3,182	6.64
Zone 2	Double-crested Cormorant	whole	13,870	alewife	whole	3,182	4.36
Zone 2	Common Tern	egg	5,963	alewife	whole	3,182	1.87
Zone 2	Forster's Tern	egg	6,234	alewife	whole	3,182	1.96
Zone 3B	Double-crested Cormorant	whole	15,000	alewife	whole	2,375	6.32
Zone 3A	Bald Eagle	egg	13,000	carp	whole	3,974	3.27

Species	RME BMF	TRVs				RME Whole Fish Concentrations (µg/kg)			
		Reproduction		Deformity		Reproduction		Deformity	
		NOAEC (µg/kg)	LOAEC (µg/kg)	NOAEC (µg/kg)	LOAEC (µg/kg)	NOAEC (µg/kg)	LOAEC (µg/kg)	NOAEC (µg/kg)	LOAEC (µg/kg)
Common Tern	1.87	4,700	7,600	800	8,000	2,508	4,055	427	4,269
Forster's Tern	1.96	4,700	7,600	800	8,000	2,399	3,879	408	4,083
Double-crested Cormorant	5.77	4,700	7,600	800	8,000	814	1,317	139	1,386
Bald Eagle	6.63	4,700	7,600	800	8,000	709	1,147	121	1,207

Table 7-11 Sediment Quality Thresholds Estimated for Ecological Effects

Species	Effect	Whole Fish Concentration (µg/kg ww)	Estimated SQT (µg/kg)
benthic invertebrates	Threshold Effect Concentration (TEL)	—	31.6
walleye	NOAEC - fry growth and mortality LOAEC - fry growth and mortality	760 7,600	176 1,759
carp	NOAEC - fry growth and mortality LOAEC - fry growth and mortality	760 7,600	363 3,633
common tern	NOAEC - hatching success LOAEC - hatching success NOAEC - deformity LOAEC - deformity	2,508 4,055 427 4,269	3,073 4,969 523 5,231
Forster's tern	NOAEC - hatching success LOAEC - hatching success NOAEC - deformity LOAEC - deformity	2,399 3,879 408 4,083	2,940 4,753 500 5,003
double-crested cormorant	NOAEC - hatching success LOAEC - hatching success NOAEC - deformity LOAEC - deformity	814 1,317 139 1,386	997 1,614 170 1,698
bald eagle	NOAEC - hatching success LOAEC - hatching success NOAEC - deformity LOAEC - deformity	709 1,147 121 1,207	339 548 58 577
mink	NOAEC - reproduction and kit survival LOAEC - reproduction and kit survival	50 500	24 239

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Appendix A

Letter from David Webb Wisconsin Department of Natural Resources, August 3, 1998

CORRESPONDENCE/MEMORANDUM — **State of Wisconsin**

DATE: August 3, 1998 FILE REF: 3200

TO: Bruce Baker - AD/5

FROM: Ed Lynch - RR/3 EXL
David Webb - WT/2 DW

SUBJECT: Screening for Chemicals of Concern for the Fox River Risk Assessment

SUMMARY

The Screening Level Risk Assessment (SLRA) for the Fox River was completed by RETEC and delivered to the Department on June 16, 1998. One of its primary purposes was to use as a starting point the "universe" of chemicals and identify (screen) chemicals from which existing risk to the ecosystem and human health would be quantified in the Baseline Risk Assessment. Not all chemicals are important to consider for the Baseline Risk Assessment. Ideally, the Baseline Risk Assessment should focus on chemicals which are most likely to contribute significantly to risks calculated for exposure scenarios involving that medium, so that the risk assessment is focused on the "most significant" chemicals (U.S. EPA; Risk Assessment Guidance for Superfund Volume 1; Publ. No. EPA/540/1-89/002). The SLRA provides a scientific process for eliminating chemicals from consideration when quantifying risk of injury or toxicity to the ecosystem and humans.

PCBs are a known source of risk to humans who consume Fox River fish and are the primary contributor to overall risk in the Fox River system. It was important for the SLRA to provide a means by which other chemicals could be screened to quantify overall risk to the system (to determine whether any other chemicals needed to be advanced to the Baseline Risk Assessment in addition to PCBs). Chemicals to be Advanced to the Baseline Risk Assessment include the following:

PCBs (Total and/or Aroclor 1242)
2,3,7,8-TCDD (Dioxin)
2,3,7,8-TCDF (Furan)
DDT/DDE/DDD
Dieldrin
Arsenic
Lead
Mercury

DISCUSSION

The Screening Level Risk Assessment was designed to screen risk and chemicals based upon both ecosystem and human-health effects. Screening for chemicals to be considered in the Baseline Risk Assessment, and further in the Remedial Investigation (RI) and Feasibility Study (FS) was to involve a 4-tiered approach. The following is a summary of the layers of screening - narrowing the list of chemicals as the tiers are executed:

Tier 1: Include all chemicals which have been detected in the system.
Tier 2: Compare detected chemicals against background levels - include those which exceed background levels.
Tier 3: Compare chemicals from tier 2 against applicable"toxicology based" values (water quality standards, sediment benchmarks, etc.)
Tier 4: Ascertain whether chemicals resulting from tier 3 co-occur with PCBs. If a particular chemical significantly co-occurs with PCBs, it would be excluded.

There is a massive amount of data from which to complete the screening tiers. Because of difficulties in distilling all of the data to a usable format, and also a need to adhere to deadlines for completing tasks, tiers 2 and 4 were not fully completed. It is unlikely that executing tier 2 (comparing against background) would have resulted in screening out many chemicals, since the relationship between chemical levels and "background" is often unclear. It is unlikely that formal comparisons will be made to background levels for the purposes of removing chemicals from consideration.

Executing tier 4 (co-occurrence with PCBs) likely would have resulted in many chemicals being eliminated from consideration since many of the chemicals in the Fox River system are located in similar areas (depositional areas of soft-sediment). As available information allows, efforts to ascertain whether particular chemicals co-occur with PCBs will occur and there is a high probability that many chemicals would co-occur with PCBs to a significant extent. At that time, such a chemical would be removed from the list of chemicals considered in the Baseline Risk Assessment.

The tally of all chemicals which were identified in the SLRA (those that did not get screened-out) is approximately 75 chemicals (treating Polycyclic Aromatic Hydrocarbons - PAHs as one chemical). For the purposes of the SLRA, the Fox River was segmented into five areas of interest (Green Bay, between DePere and Green Bay, between Wrightstown and DePere, between Appleton and Wrightstown, and between Little Lake Butte des Morts and Wrightstown). For each area of interest, chemicals were screened in various environmental media (water, sediment, tissue, etc.). An important consideration for whether a chemical will significantly contribute to toxicity and/or risk to the ecosystem and human health is the spatial extent of a chemical's impact. It is important to account for whether a particular chemical is eliciting system-wide impacts, or whether it was retained through the screening process due to its detection in one media in one area of interest. The final list of chemicals (PCBs, 2,3,7,8-TCDD, 2,3,7,8-TCDF, DDT/DDE/DDD, Dieldrin, Arsenic, Lead, and Mercury) is inclusive of chemicals identified in both screens (ecosystem and human health).

In addition to the system-wide nature of a chemical's impact, it is helpful to make a qualitative determination of how a chemical was retained through the screening process relative to other chemicals (PCBs). The approaches to quantify potential risk in the ecological and human-health screens use metrics referred to as the Hazard Quotient (HQ) and the Relative Risk Ratio (RRR), respectively. The HQ is derived by taking the ratio of the maximum detected concentration of the chemical in a particular media to its applicable corresponding media-specific toxicologically based value (e.g., a water quality standard, a sediment benchmark, etc.). Generating RRRs for human health is more complex than ecological due to cancer and non-cancer effects, and the complexity of working with human health toxocological data. HQs and RRRs are conceptually similar, but not identical in their derivation or interpretation. In either case, as the magnitude of the HQ or RRR increases, the potential for a particular chemical to exert an adverse effect to the ecosystem or to human health increases. In order to put some of the chemicals into perspective, it is useful to compare the HQ or RRR of a particular chemical to the HQ and RRR for PCBs. The HQs and RRRs for the other chemicals (based upon either the human health screen or the ecosystem screen) are the following:

	Highest HQ or RRR
PCBs (RRR)	1,300,000
PCBs (HQ)	5,900
2,3,7,8-TCDD (RRR)	9,800,000
2,3,7,8-TCDF (RRR)	1,800,000
Dieldrin (RRR)	12,903
DDT (RRR)	4,500
Arsenic (RRR)	2,200
Mercury (RRR)	220

Of the metals analyzed in sediment and tissue samples from the Lower Fox River and Green Bay system, only arsenic, lead, and mercury were retained for the Baseline Risk Assessment. The most critical exposure pathway from the perspective of human health effects is ingestion of fish. Arsenic, cadmium, chromium, copper, lead, mercury, selenium, and zinc were analyzed in fish tissue and all but arsenic and mercury were at levels that did not warrant advancement to the Baseline Risk Assessment. Thus, arsenic and mercury were retained for the Baseline Risk Assessment. Lead was found in fish tissue but was not evaluated in the Screening Level Risk Assessment because appropriate toxicological thresholds were not readily available. Thus, lead was retained for a more detailed evaluation in the Baseline Risk Assessment.

The attachment to this memo identifies the rationale for the selection of each of the chemicals of concern being advanced to the Baseline Risk Assessment. In addition, it provides comments on a few additional chemicals.

Please contact David Webb if you have questions.

Approved:

Bruce Baker, Deputy Administrator
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Dave Morgan - RETEC
Milt Clark - U.S. EPA Region 5; SR16J
Jim Hahnenberg - U.S. EPA Region 5; SR16J
Keith Eastin - Delliotte and Touche

coc.daw

ATTACHMENT

For each chemical considered, there were different factors which caused a chemical to be retained or discarded from consideration. The following is a synopsis of the factors which caused a particular chemical (or class of chemical) to be retained for consideration in the Baseline Risk Assessment; additionally, a brief discussion follows for chemicals (or classes of chemicals) which are notably absent:

PCBs (Total and/or Aroclor 1242):
Fish consumption advisories are wide-spread.
The range of RRRs (based upon human health) is 172,000 to 1,261,314.
The prevelance and impact of PCBs in the Fox River system is widespread.

2,3,7,8-TCDD (Dioxin):
The range of RRRs (based upon human health) is large and as high as 9,766,082 (lack of complete data preclude comparison using all river areas and exposure pathways).
The prevelance and potential impact to the ecosystem and human health of 2,3,7,8-TCDD is widespread in the Fox River system.

2,3,7,8-TCDF (Furan):
The range of RRRs (based upon human health) is large and as high as 1,752,895 (lack of complete data preclude comparison using all river areas and exposure pathways).
The prevelance and potential impact to the ecosystem and human health of 2,3,7,8-TCDD is widespread in the Fox River system.

DDT/DDE/DDD:
The range of RRRs (based upon human health) is 43 to 4,528. In addition, the potential for ecological impacts, especially to avian populations, is significant.

Dieldrin:
The range of RRRs (based upon human health) is 323 to 12,903 (lack of complete data preclude comparison using all river areas and exposure pathways).

Arsenic:
Human health RRRs for arsenic were as high as 2,200. Also, arsenic was found in fish at levels which exceed appropriate cancer risk (10^{-4}) used in a screening context.

Lead:
Ecological HQ's were as high as 41. Human health risk were not evaluated thoroughly because screening levels were not readily available.

Mercury:
The range of HQs (based upon human health) is 48 to 220. Mercury impacts on human health and ecosystem integrity are relatively well known and in its methylated form, mercury is highly bioaccumulative.

Other chemicals which may be considered in the other risk assessment efforts include the following:

Copper:
Copper was detected at significant concentrations in game fish - up to 3,400 ug/kg in the area downstream of DePere. HQ's for copper are not elevated enough relative to other compounds, it is not widespread, and not bioaccumulative (relatively).

Zinc:
Zinc was detected at significant concentrations in game fish - up to 13,000 ug/kg in the area downstream of DePere. However, the RRR associated with ingestion of this fish was below one. Ecologically based HQs for zinc are not elevated enough relative to other compounds, it is not widespread, and not bioaccumulative (relatively).

Polycyclic Aromatic Hydrocarbons (PAHs):
Risks due to PAHs are difficult to quantify since many PAHs are efficiently metabolized. System-wide impacts are unlikely to occur, and while bioaccumulation can be an exposure pathway, reliable data is not available and in some cases not accurate due to metabolism.

Note: The concepts of HQs and RRRs are used here for presentation purposes and have been simplified for brevity and comparative purposes. For a more complete and thorough discussion/presentation of the details, the Screening Level Human Health and Ecological Risk Assessment - Lower Fox River Site, Wisconsin, June 15, 1998 should be consulted.

Appendix B

Human Health Fate and Transport Models, Transport Factors, and Reduction Factors

Appendix B1

Additional Evaluation of Exposure to PCBs in Fish from the Lower Fox River and Green Bay

Appendix B1

Additional Evaluation of Exposure to PCBs in Fish from the Lower Fox River and Green Bay

JUNE 2, 2000

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Table of Contents

Table of Contents

List of Tables

List of Tables

List of Figures

List of Figures

List of Figures

List of Figures

List of Figures

1 Introduction

This appendix expands upon the focused evaluation of exposure to PCBs in fish provided in Section 5.9 of the main report. The focused evaluation of exposure to PCBs in fish examined exposures for two categories of anglers (recreational anglers and high intake fish consumers), for five different categories of fish (all fish data, carp, perch, walleye and white bass) and for the four reaches of the Lower Fox River (Little Lake Butte des Morts, Appleton to Little Rapids, Little Rapids to De Pere, and De Pere to Green Bay) and three zones of Green Bay (zones 3A, 3B and 4). In the main report, for each category of angler, intake assumptions were developed for two exposure scenarios: reasonable maximum exposures (RMEs) and central tendency exposures (CTEs). For each intake parameter, a distribution of values was developed and point values were selected from the distribution for the RME and CTE scenarios. For a number of parameters, the 90^{th} or 95^{th} percentile was selected as the point estimates for the RME scenario while for other parameters, the mean or median was selected as the point estimate for the RME scenario. For the CTE scenario, mean or median values were selected as the point estimate for all parameters. These point estimates for individual parameters were used in Section 5.9 to generate point estimates of risk and hazard index for the RME and CTE scenarios. Point estimates of risk and hazard index were generated for each category of angler, for the different categories of fish and for the various reaches and zones of the Lower Fox River and Green Bay.

Section 2 of this appendix provides the equations used to calculate exposures, risks and hazard indices for the two categories of anglers. This section also discusses the distributions utilized for each intake parameter and the RME and CTE point estimates selected for each parameter from the distribution for each parameter.

On behalf of the Fox River Group (FRG), Exponent, Inc. prepared a human health risk assessment for the Lower Fox River. In their risk assessment, Exponent (2000) developed distributions for a variety of intake parameters and used those distributions to develop distributions of risks and hazard indices. Exponent (2000) did not use the distributions to select RME and CTE values and then calculate point estimates of the risks and hazard indices for these two scenarios. In Section 3 of this appendix, RME and CTE values are selected for each intake parameter using the distributions provided by Exponent (2000). Using the selected values, point estimates of risks and hazard indices are calculated for the RME and CTE scenarios. These point estimates of risks and hazard indices are

then compared to the point estimates of risks and hazard indices calculated in the focused evaluation in Section 5.9 of the main report.

Section 4 of this appendix presents a probabilistic evaluation for the recreational angler and high intake fish consumer based on the distributions of intake parameters presented in Section 2 of this appendix. Since the probabilistic risk assessment develops distributions of risks and hazard indices, the location on the distributions of the point estimates of risk and hazard index using RME and CTE values can be ascertained. This allows the RME and CTE point estimates of risks and hazard indices to be placed in the range of risks calculated using probabilistic methods and provides a context for interpreting the point estimates of risks and hazard indices.

It is important to emphasize that the probabilistic risk assessment is not intended to be the principal basis for decisions regarding the need for remedial action at a site. EPA guidance specifies that point estimates of risks and hazard indices calculated using point estimates of intake parameters for RME and CTE scenarios are the principal basis for such decisions. Therefore, the probabilistic risk assessment does not supercede the point estimate evaluation, but is intended to supplement and complement the point estimates of risks and hazard indices. The probabilistic risk assessment has considered draft EPA guidance on probabilistic risk assessment (EPA, 1999).

Section 5 of this appendix provides the references cited in the appendix.

2 Basic Equations and Intake Parameters

This section presents the basic equations for calculating risks and hazard indices for receptors potentially exposed to PCBs present in fish in the Lower Fox River and Green Bay. The notation used is consistent with that used in the main report. In addition, this section discusses the concepts of variability and uncertainty, as well as the choice of the distribution for each intake parameter used in the probabilistic assessment presented in Section 4 of this appendix.

2.1 Equations for Calculating Cancer Risks and Hazard Indices

2.1.1 Cancer Risk Evaluation

The equation used to assess cancer risks from ingestion of fish is:

$$R = Ic \cdot CSFo$$

where:

R = cancer risk
Ic = intake from ingestion of fish averaged over a lifetime (mg/kg-day)
CSFo = oral cancer slope factor [$(mg/kg\text{-}day)^{-1}$]

The intake from fish ingestion averaged over a lifetime is given by:

$$Ic = \frac{Cfish \cdot IR \cdot RF \cdot ABS \cdot CF \cdot EF \cdot ED}{BW \cdot ATc}$$

where:

Cfish = concentration in fish (mg/kg)
IR = fish ingestion rate (g/day or g/meal)
RF = reduction factor due to trimming and cooking fish (mg/mg)
ABS = absorption factor for ingestion of fish (mg/mg)
CF = 10^{-3} kg/g
EF = exposure frequency (days/year or meals/year)
ED= exposure duration (years)
BW = body weight (kg)

ATc = averaging time for cancer risks (days)

The intake equation can be rewritten as:

$$Ic = Cfish \cdot IntFacC$$

$$IntFacC = \frac{IR \cdot RF \cdot ABS \cdot CF \cdot EF \cdot ED}{BW \cdot ATc}$$

where:

IntFacC = intake factor for cancer risk [$(mg/kg)^{-1}$]

The equation for assessing cancer risks from ingestion of fish can be rewritten as:

$$R = Cfish \cdot IntFacC \cdot CSFo$$

2.1.2 Noncancer Effects Evaluation

The equation for calculating the chronic hazard index from ingestion of fish is:

$$HI = \frac{Inc}{RfDo}$$

where:

HI = chronic, noncancer hazard index
Inc = intake from ingestion of fish averaged over the exposure period (mg/kg-day)
RfDo = oral reference dose for chronic, noncancer effects (mg/kg-day)

The intake from fish ingestion averaged over the exposure period is given by:

$$Inc = \frac{Cfish \cdot IR \cdot RF \cdot ABS \cdot CF \cdot EF \cdot ED}{BW \cdot ATnc}$$

These variables are the same as before except:

ATnc = averaging time for chronic, noncancer effects (days)

The intake equation can be rewritten:

$$Inc = Cfish \cdot IntFacNC$$

$$IntFacNC = \frac{IR \cdot RF \cdot ABS \cdot CF \cdot EF \cdot ED}{BW \cdot ATnc}$$

where:

IntFacNC = intake factor for chronic, noncancer effects [$(mg/kg)^{-1}$]

The equation for calculating the chronic hazard index from ingestion of fish can be rewritten as:

$$HI = \frac{Cfish \cdot IntFacNC}{RfDo}$$

2.2 Variability and Uncertainty

In Section 2.1, the equations used to calculate intakes and estimate cancer risks and noncancer hazard indices were presented. A number of parameters appear in the intake equations. For almost all of these parameters, a single fixed value does not definitively characterize the parameters. Instead, a distribution of values is a more appropriate choice for representing the parameter. The value selected for the parameter in a point estimate analysis depends on the objectives of a particular analysis. A distribution of values exists for most parameters due to variability, uncertainty or both. The concepts of variability and uncertainty, as applied to probabilistic risk assessment, are defined in EPA (1999) as follows (pages 1-3 and 1-4):

- Variability: True heterogeneity or diversity that characterizes an exposure variable or response in a population. Further study (e.g., increasing sample size, n) will not reduce variability, but it can provide greater confidence in quantitative characterization of variability.

- Uncertainty: Lack of knowledge about specific variables, parameters, models, or other factors (e.g., uncertainty regarding the concentration of a contaminant in an environmental medium, local fish consumption practices). Uncertainty may be reduced through further study.

To illustrate the difference in variability and uncertainty, consider the parameters fish ingestion rate (IR) and exposure frequency (EF). When multiplied together, these parameters yield the quantity of self-caught fish that an angler consumes in a year. It is known that in the population of recreational anglers, there is a

considerable range in the amount of fish consumed annually by recreational anglers. Some anglers eat none of the fish they catch, while other anglers eat many meals of self-caught fish each year. This range in annual fish consumption rates is inherent to the population of recreational anglers and reflects the variability in annual fish consumption rates in the recreational angler population. A number of surveys of recreational anglers have been conducted to define the distribution of annual fish consumption rates for recreational anglers. Each survey generates a somewhat different distribution of annual fish consumption rates and the differences in the various distributions reflect the uncertainty in the characterization of variability.

EPA guidance (1999) indicates that probabilistic risk assessments should attempt to isolate the influences of variability and uncertainty on the calculation of risks and hazard indices. This guidance (EPA, 1999) recommends performing one dimensional or two dimensional probabilistic assessment. In a one dimensional probabilistic assessment, distributions are assigned to parameters to characterize the variability in each parameter. If there is uncertainty in the distribution that should be assigned to a specific parameter, this can be evaluated by performing multiple one dimensional analyses with different distributions assigned to a parameter in each analysis. In theory, the evaluation of uncertainty can be taken one step further. Distributions can be assigned to variables that are uncertain and the influence of uncertainty can be evaluated in a two dimensional probabilistic assessment. For example, if the variability in a parameter is characterized by a normal distribution, but there is uncertainty associated with the mean and standard deviation that define this normal distribution, then the uncertainty can be expressed by assigning distributions to the mean and standard deviation. In a two dimensional probabilistic assessment, values for the mean and standard deviation are randomly selected from the distributions for these variables and these values are used to perform a probabilistic risk assessment to characterize the variability associated with these values. Then new values for the mean and standard deviation are chosen and the process is repeated. The outcome of a one dimensional probabilistic risk assessment is typically a single distribution characterizing the range in risk (or hazard index). The outcome of an uncertainty analysis (either multiple one dimensional probabilistic risk assessments or a two dimensional probabilistic risk assessment)is a series of distributions characterizing both the range in risk (or hazard index) and the uncertainty associated with this range.

In the probabilistic analysis presented in this appendix, an attempt has been made to characterize the variability inherent in a number of parameters by using probability distributions for the variable parameters and performing one dimensional probabilistic risk assessments. By doing so, the likelihood of different

risks in a potentially exposed population was quantified. The probability distributions of risks and hazard indices presented in Section 4 can be used to answer the question "what is the probability that the risks (or hazard indices) will exceed a regulatory level of concern (e.g., 10^{-5} or 10^{-6})?"

It should be noted that in the analysis presented in this appendix, for all parameters but one, no attempt has been made to quantify uncertainty. The only quantity for which uncertainty has been incorporated in the analysis is the average yearly quantity of fish ingested by the receptor population (g/year) (this is the product of two parameters, the daily fish ingestion rate and the exposure frequency). For this quantity, probability distributions of risks and hazard indices were generated based on fish ingestion rates calculated in various studies (see Section 2.4). The probability distributions obtained based on each study were then combined and used to estimate the uncertainty in the risk and hazard index estimates using a procedure recommended in EPA (1999) (see Section 4).

Section 2-3 reviews each intake parameter. The assumptions used for fish intake rate and exposure frequency are examined in more detail in Section 2.4. As discussed above, the results of the various studies consulted were used to evaluate the uncertainty in the risks and hazard indices. Attachment 1 provides summary statistics tables and histogram plots for all input distributions used in the probabilistic risk evaluation.

2.3 Intake Parameter Evaluation

2.3.1 Fish Concentration (Cfish)

The parameter Cfish represents the mean concentration of PCBs in fish consumed by anglers over the exposure period. Tables 2-1 through 2-3 present the distributions of Cfish used in the analysis presented in this appendix. Table 2-1 describes the distribution of fish concentration used by Exponent (2000), which represents fish fillet (no skin) data collected from the entire Lower Fox River. ThermoRetec calculated exposures to anglers for different reaches of the Lower Fox River and zones of Green Bay. For this evaluation, distributions were developed for the Little Lake Butte des Morts (Table 2-2) and De Pere to Green Bay (Table 2-3) reaches using all fish fillet data (most of these samples were fillet with skin). These reaches are the most populated and likely to have the most anglers.

It is recognized that there is wide variability in the PCB concentrations in fish caught in the Lower Fox River. There is also variability associated with the mean concentration in fish consumed by anglers over the exposure period (which

represents the exposure point concentration in the probabilistic risk assessment). This can be understood with the following considerations.

If a large number of anglers (say, a thousand) were engaged in a study and the concentration of total PCBs in each self-caught fish the angler consumed was determined over a long period of time (such as a 10 year period), an average concentration of total PCBs in fish could be determined for each angler. These data could then be used to determine a distribution of the mean concentration of total PCBs in fish for the angler population. There are at least three sources of this variability.

- Variability of concentrations in a species: The concentrations of total PCBs in fish of the same species vary considerably based on analysis of samples from different fish of the same species. This variability is due to a number of factors including the age of the fish, the length of time the fish has spent in the Lower Fox River or Green Bay, the intrinsic biochemical process such as metabolism and depuration in the individual fish, and the mix of food (zooplankton, benthic invertebrates, fish) the fish consumes.

- Variability between species: The concentrations of total PCBs in fish vary between species. For example, the concentration of total PCBs in carp is greater on average than the concentration of total PCBs in bass, perch or walleye. This variability can be characterized by analyzing samples from different fish species.

- Mix of fish the angler consumes: Based on surveys, the mix of fish species that angler populations consume has been characterized. In general, this survey data is presented for the angler population as a whole. For example, Hutchison and Kraft (1994) report that only 2% of the fish caught by Hmong anglers is carp. What is not known is whether a small number of Hmong anglers eat a substantial amount of carp and all other Hmong anglers eat virtually no carp (scenario 1) or a large number of Hmong anglers eat a small amount of carp (scenario 2). The first scenario leads to a small number of anglers eating fish with significantly higher concentrations of total PCBs than more commonly consumed fish species such as bass, perch and walleye. This scenario leads to a larger range in the mean concentration in fish that anglers are exposed to than does the second scenario. The mix of fish consumed by individual anglers is therefore both variable and uncertain.

If all other factors are held constant, the variability in the mean PCB concentration in fish consumed by anglers over the study period will be greater for anglers who eat a small number of meals over the study period than for anglers who eat a large number of meals over the same period. As more fish are consumed, the standard deviation on the distribution of the mean PCB concentration (Cfish) will become smaller.

ThermoRetec and Exponent (2000) have taken different approaches to estimate Cfish. These approaches are discussed below.

Exponent (2000)

Exponent (2000) did not distinguish between reaches in the Lower Fox River, and used a distribution for the fish PCB concentration that is representative of the time averaged concentration in fish that are caught and eaten after each fishing trip. The sampling distribution for the arithmetic mean was used to describe the PCB concentration in fish tissue. For each species of fish, this distribution was taken as normal with a mean equal to the sample mean, and a standard deviation equal to the sample standard deviation divided by the square root of the number of values for which the sample mean was calculated. To constrain the distribution to physically relevant values, the distribution was truncated at a minimum of zero and a maximum of three standard deviations above the mean. Distributions were calculated for a number of species and added up; distributions were then weighted by the fraction of times the fish species was determined to be consumed. The weighting factors used by Exponent were:

- Walleye: 0.26
- Smallmouth Bass: 0.03
- Yellow perch: 0.70
- Brown trout: 0.01

In addition, Exponent assumed that the average PCB concentration in fish is decreasing exponentially over time. The average concentration for each fish species over the exposure period was taken as the exposure point concentration. This concentration is lower than the concentration measured in fish today, as fish concentration is assumed to decrease over the exposure period. The rates at which fish concentrations were assumed to decrease are as follows:

- Walleye: 0.058/year (half life of 12 years)
- Smallmouth Bass: 0.116/year (half life of 6 years)
- Yellow perch: 1.16/year (half life of 6 years)
- Brown trout: 0.12/year (half life of 5.8 years)

It should be noted that Exponent (2000) was essentially characterizing the uncertainty in the mean and not the variability. In exponents approach, if more fish samples were collected, the standard deviation would decrease which is a characteristic of uncertainty. Also, in their approach, the uncertainty in the PCB concentration in fish consumed by anglers is assumed to be independent of the number of meals consumed over the exposure period. This is not consistent with the results that would be obtained by performing the thought experiment described above.

ThermoRetec

In determining the average PCB concentration in fish consumed by Lower Fox River and Green Bay anglers, ThermoRetec simulated numerically the thought experiment described above. The procedure used by ThermoRetec is described below.

- The anglers were assumed to catch all their fish from either the Little Lake Butte des Morts or the De Pere to Green Bay reach. All fillet data for the 1990s from each reach were used as the data set for each reach on the assumption that more commonly consumed fish species were caught and tested during this time period. For each dataset, the mean and standard deviation were calculated.

- For each reach, it was assumed that the mean and standard deviation of the fish data represents the variability inherent in fish caught by anglers over the exposure period and consumed in a single meal. This distribution was assumed to be constant over the exposure period, i.e., no decreases over time were assumed.

- The mean PCB concentration in fish to which an individual angler is exposed over the exposure period was determined by calculating the distribution of the mean of the single-meal PCB fish concentration over the number of meals consumed during the exposure period. Since for the majority of anglers the assumed number of meals over the exposure period is large (greater then 100), the distribution of the mean fish concentration over the exposure period was calculated (using the central limit theorem, confirmed by numerical experimentation) as a normal distribution with mean equal to the mean of the fish concentration within each reach, and standard deviation equal to the standard deviation of the fish concentration within each reach divided by the square root of the number of meals consumed. The distributions were truncated at minimum and maximum values equal to the minimum and maximum values measured within each reach (see Tables 2-2 and 2-3).

- It should be noted that by assuming that all anglers catch fish from the same pool of fish, it is assumed that all anglers have the same preference for individual fish species. If some anglers prefer carp to all other fish, their average PCB concentration would be higher than the average for other anglers. Thus, this procedure underestimates the variability in the mean PCB concentration associated with the fact that some anglers may eat more fish of one species than other anglers.

The procedure used by ThermoRetec is consistent with the results that would be obtained by performing the thought experiment described earlier. It should be noted that as the number of meals increases, the variability in the mean fish concentration decreases (as measured by the standard deviation of the distribution of the mean), as is expected based on the thought experiment previously described.

It should also be noted that ThermoRetec's procedure does not include an evaluation of uncertainty in the mean fish concentration, and that the distribution used for the mean PCB concentration to which anglers are exposed only represents the variability of this parameter.

2.3.2 Fish Ingestion Rate (IR) and Exposure Frequency (EF)

These parameters are the amount of fish consumed per meal (IR) and the exposure frequency (EF) or meals per year. Both parameters are known to vary within the angling population, and both parameters can be characterized by surveys of anglers. Many surveys, however, only characterize the number of fish meals per year (or in a shorter period), so the meal size must be estimated from other sources. In many studies, the estimates of IR and EF are multiplied together to give the mass of fish consumed per year, and this result is then divided by 365 days/years to give an annualized IR. The distribution of this annualized IR (in g/day) is the final published result.

The data summarized in this appendix utilizes information gleaned from studies that used both approaches. As such, depending on the study cited, IR is given in either g/day or g/meal, and EF is given in either days/year or meals/year. Regardless, the product IR*EF is always in g/year.

It should also be noted that EF or a normalized IR should reflect the number of meals of fish caught from the Lower Fox River or Green Bay. It is known that the Lower Fox River and Green Bay are not the only water bodies used for fishing by anglers living in the region. Surveys can quantify existing behavior and the data used by Exponent (2000) uses survey results for the Lower Fox River. However, it is known from other surveys (e.g., Hutchison, 1999) that the behavior of

anglers has been modified by fish consumption advisories on the Lower Fox River and Green Bay. These advisories affect 1) the frequency of fishing on these water bodies; 2) whether fish are kept for consumption or returned to the water body; and 3) the type of fish kept for consumption. A baseline evaluation should estimate what potential exposures would be in the absence of such advisories. Thus, survey data from the Lower Fox River should not be used unless these data are adjusted in some manner to account for the influence of fish advisories on sport-fish consumption patterns. If such data are used without adjustment, the risks and hazard indices calculated with these data will underestimate the risks and hazard indices that would result if the advisories were lifted. Therefore, these results must be used with caution.

The distributions of IR and EF used in the probabilistic risk evaluation are discussed in Section 2.4 based on studies of different angler populations. As discussed in Section 2.4, different studies report different results for the average fish intake. Within each study, IR and EF are characterized by variability. That is, different receptors have different fish intakes, according to the various published distributions. However, the fact that different studies report different results indicates that some uncertainty is present in the estimation of intake rates. Thus, the product IR*EF is characterized by both uncertainty and variability. Both are accounted for in ThermoRetec's analysis, as discussed in Section 4.

2.3.3 Reduction Factor (RF)

The reduction factor represents the fraction of the initial mass of PCBs in fish that remains after trimming and cooking. In this appendix, a distinction is made between the reduction factor for fish consumed in a single meal (referred to as the single-meal reduction factor) and the mean reduction factor over a number of meals. The latter is the parameter relevant to the risk assessment calculation and is referred to as the mean reduction factor, designated by the variable RF.

It is recognized that variability is associated with the single-meal reduction factor for fish caught in the Lower Fox River or Green Bay, cooked, trimmed and then consumed by local anglers. As a consequence, variability is also associated with the mean reduction factor in fish consumed by anglers (RF) which is used in the risk assessment.

Losses due to trimming and cooking are a source of variability. For an individual angler, the losses can vary between meals depending on how the angler prepares and cooks the fish. In addition, different anglers may use preferentially different cooking and trimming techniques. As such, the single-meal reduction factor is characterized by inherent variability among the angling population. If a large number of anglers (say, a thousand) were engaged in a study and the reduction

in mass of total PCBs was measured from the raw fish to the final trimmed and cooked product in each meal the angler ate, an average single-meal reduction factor could be determined for each angler. These data could then be used to determine a distribution of the reduction factor for all anglers. This experiment has not been performed; however, data are available on the reduction in PCB concentrations in fish due to trimming and cooking techniques.

If all other factors are held constant, the variability in the mean reduction factor in fish consumed by anglers (which is used in the risk assessment) will be greater for anglers who eat a small number of meals over the study period than for anglers who eat a large number of meals over the same period. As more fish is consumed, the standard deviation on the distribution of RF will become smaller. As such, the distribution of RF is a function of the number of fish meals consumed by anglers over the exposure period.

Table 2-4 presents the distributions used for the RF used by Exponent (2000) and ThermoRetec. These distributions are discussed below.

Exponent (2000)

Exponent (2000) used only fillet with no skin data in estimating their fish concentration. Consequently, Exponent (2000) used a distribution for RF that reflects losses from cooking only. This reduction factor was developed by Wilson et al. (1998) based on reductions observed from cooking fish. Exponent used a cumulative distribution with a mean of 0.635, maximum of 1 (corresponding to no reduction in the PCB concentration in fish), and minimum of 0 (corresponding to 100% reduction in the PCB concentration in fish).

It should be noted that in the risk assessment, Exponent used the distribution for the single-meal reduction factor, rather than the distribution for the mean reduction factor over the meals consumed during the exposure period.

ThermoRetec

When ThermoRetec fish concentration data are used, a reduction factor reflecting losses due to trimming as well as cooking is needed, because the ThermoRetec fish concentration distribution was developed from fish concentration data that are primarily fillet with skin data.

As previously discussed, the reduction factor is a function of how fish is trimmed and how it is cooked (e.g., broiled vs. fried). It is reasonable to expect that each individual angler will not trim and cook fish always in the same manner. As such, the reduction factor will vary according to a certain probability distribution.

To estimate this probability distribution, ThermoRetec made the following assumptions.

- Trimming is generally performed by anglers prior to cooking the caught fish.

- The reduction factor estimated by Wilson et al. (1998) for fillet with no skin can be used for estimating the reduction factor in fish already trimmed.

Based on the first assumption, the reduction factor due to cooking and trimming can be expressed as:

$$\text{Single-Meal Reduction Factor} = RF_{trim} * RF_{cook}$$

Where RF_{trim} represents the fraction of PCB mass remaining in fish after trimming (single-meal), and RF_{cook} represents the fraction remaining after cooking (single-meal).

Based on the second assumption, RF_{cook} was taken to be distributed according to the data presented in Wilson et al. (1998), consistent with Exponent (2000) assumptions. Limited data are available specifically on RF_{trim} and these data have not been reviewed and compiled by investigators with the same level of scrutiny as for RF_{cook}. However, based on information published in Anderson et al. (1993), the average of the single-meal reduction factor due to the combined effect of trimming and cooking is likely to be approximately 50%. Thus, RF_{trim} was chosen such that the average of $RF_{trim} * RF_{cook}$ is 50%. Since, based on the distribution presented in Wilson et al. (1998), the average of RF_{cook} is 63.5%, a distribution was assumed for RF_{trim} whose average is 78.7%. This distribution was assumed to be uniform with a variation of plus or minus 19.7% (which represents 25% of the average value) around the average value of 78.7%. The single-meal reduction factor was therefore taken as the product of the cumulative distribution described in Table 2-4a, and a uniform probability distribution with maximum and minimum values of 59% and 98.4%.

As previously discussed, the distribution discussed above represents the variability in the overall reduction factor associated with generally cooking and trimming fish in a single meal. The distribution of the mean single-meal reduction factor in fish consumed by anglers (RF) depends on the number of meals consumed by anglers over the exposure period.

Similar to what was assumed for Cfish, the mean reduction factor for an individual angler over the exposure period was calculated by estimating the distribution of the mean of the single-meal reduction factor over the number of meals consumed during the exposure period. Since for the great majority of anglers, the assumed number of meals over the exposure period is large (greater then 100), the distribution of the mean reduction factor over the exposure period was calculated (using the central limit theorem) as a normal distribution, with mean equal to the mean of the single-meal reduction factor (0.5) and standard deviation equal to the standard deviation of the single-meal reduction factor (0.2) divided by the square root of the number of meals consumed (see Table 2-4b).

It should be noted that this procedure assumes that all anglers trim and cook fish in a similar way. If some anglers trim less and cook fish in a stew on a regular basis, their average reduction factor would be higher (i.e., less PCBs would be lost) than estimated here. Therefore, this procedure tends to underestimate variability.

It should also be noted that ThermoRetec's procedure does not include an evaluation of uncertainty in the mean reduction factor, and that the distribution used for the mean reduction factor to which anglers are exposed only represents the variability of this parameter.

2.3.4 Absorption Efficiency (ABS)

The absorption efficiency is based on the studies used to generate the cancer slope factors and reference doses for PCBs. In general, PCBs in fish are considered to be fairly readily assimilated when ingested, and the vehicle for delivering PCBs to the animals used to develop the cancer slope factor and reference dose for total PCBs also resulted in significant absorption as discussed in Section 5 of the main report. Therefore, it was assumed that all PCBs in ingested fish were assimilated by the body in a manner similar to the animals used to develop the cancer slope factors and reference doses, so ABS was set to 1. This same assumption was used in Exponent (2000).

2.3.5 Exposure Duration (ED)

The exposure duration represents the number of years that the angler pursues angling. More specifically, the exposure duration is the number of years an angler catches fish at the rates specified by IR and EF. Variability is associated with ED. For the population of anglers, ED will vary since some anglers will start fishing later in life and continue fishing for a short period of time and others will begin fishing when they are young and continue fishing for their whole lives. The parameter also depends on how long the angler lives in the study area. Thus, the parameter ED depends on: when anglers begin fishing during their lifetime; the

number of years they engage in fishing; and the number of years they remain in the Lower Fox River and Green Bay area and therefore, have the opportunity to fish from these water bodies on a regular basis.

Exponent (2000)

Table 2-5 presents the distribution of exposure duration used by Exponent (2000). Exponent (2000) developed a distribution for ED based on the survey data they had for the Fox River using the methodology of Price et al. (1998). A limitation of this method is that it depends on the survey data collected from the Lower Fox River, since it is known that angler behavior has been affected by the existence of fish advisories for the Lower Fox River and Green Bay, as discussed previously.

ThermoRetec

Table 2-6 presents the distribution of exposure duration used by ThermoRetec. ThermoRetec developed a distribution for ED based on data for residence time and information on where people move. EPA (1997) provides data on the time people spend in one residence (Table 2-7). EPA (1997) also provides data on where people move when they change residences. In general, 62 percent of the time people move within the same county, 18.5 percent of the time they move to a different county within the same state, and the remaining moves are to a different state or out of the country. These data were used to simulate the moves of an individual from one residence to another. The following process was simulated:

1) The process begins (i.e., time zero is established) when the individual enters the region (either through birth or a move into the region).

2) If i is a number representing the i^{th} residence since entering into the region, set i to 1 at time 0.

3) For the i^{th} residence determine the time spent at this residence (T_i) by picking a value randomly from the distribution of time spent in a residence (see Table 2-7).

4) Determine if the move from the i to the $i+1$ residence is within the region or out of the region. This is accomplished by selecting a value randomly from a discrete distribution described below that is either a 1 (move is within region) or a 0 (move is out of region). If the move is out of the region, it is assumed that the individual never returns to the region, so the time in a residence within the region for the $i+1$ residence and all subsequent residences is set to 0.

5) Steps 3 and 4 are repeated until the individual moves out of the region or the individual dies (i.e., the age when entering the region plus the total time spent in the region exceeds the years in a lifetime).

There are two critical assumptions needed to execute this simulation. First, the age of the individual when they enter the region must be specified. Second, the distribution specifying whether a move is within the region or out of the region must be established. As noted previously, data from EPA (1997) indicates that 62 percent of moves are within the same county, 18.5 percent of moves are to a different county within the same state, and the remaining moves are out of the state.

For this evaluation, six different starting ages were examined: age 0 years (i.e., born into region), 10 years, 20 years, 30 years, 40 years and 50 years. Also, two different distributions for moves were utilized. The first distribution assumed that all moves within the same county and 20% of the moves to a different county within the same state were within the region. All other moves were outside the region. In other words, 65.7% of moves are within the region and 34.3% of moves are out of the region. The second distribution assumed that all moves within the same county and 50% of the moves to a different county within the same state were within the region and all other moves were outside the region. In other words, 71.3% of moves are within the region and 28.7% of moves are out of the region.

Table 2-8 shows the result of simulating the time spent within the region for twenty individuals assuming the starting age is 0 years, 71.3% of moves are within the region and the lifetime is 75 years (the years in the region cannot exceed 75 years). In Table 2-8, the first column is the number identifying the individual. The next 20 columns represents the time in each residence. If the value is zero, it is assumed the individual moved out of the region in a previous move. The last column is the total number of years in the region. This is calculated by summing the years in a residence and capping this number by the years in a lifetime (75 years). This process was simulated for 5000 individuals and the results were used to develop a distribution of time spent in the region.

This distribution depends on two inputs, the age of the individual when he or she enter the region and the probability that a move will be within the region. Table 2-9 presents the mean and 95th percentile of time spent within the region depending on the start age and the percentage of moves that are within the region.

For the evaluation of exposure to an angler, the cumulative distribution presented in Table 2-6 was used. This cumulative distribution is for a person born into the region (start age is 0 years) and 65.7% of moves are within the region.

It should be noted that some uncertainty exists in ED for a variety of reasons. All sport fish consumption surveys are short term, reflecting behavior over a few weeks to a year. There are no long term angler surveys that attempt to quantify sport fish consumption patterns over a long period of time. For the Lower Fox River and Green Bay, the answers to two questions are critical in determining exposure duration.

- To what extent does short term sport fish consumption behavior reflect long term behavior by an angler? Do anglers maintain the same level of fishing and sport-fish consumption over their entire lifetime or does this behavior change? The behavior is certain to change dramatically for some anglers (either increasing or decreasing), but this change is not characterized in any long-term angler survey. Thus, this is a significant source of uncertainty.

- How many years does an angler catch fish from the Lower Fox River and Green Bay? This question can be restated in a way that relates to the previous question: How many years can short term behavior be used to predict long term behavior? As discussed previously, the answer to this question is subject to significant uncertainty. Exponent (2000) used angler survey data from the Lower Fox River to develop a distribution for ED using a methodology developed by Price et al. (1998). ThermoRetec (2000) took a different approach to estimating ED, assuming that the number of years an angler fishes from the Lower Fox River and Green Bay depends on the number of years an individual lives in the Lower Fox River and Green Bay region.

The probability distribution for ED used by ThermoRetec (Table 2-6) is representative only of variability in exposure duration among different anglers. As discussed above, there is considerable uncertainty in this estimate of variability and the results presented in Table 2-9 are reflective of this uncertainty.

2.3.6 Body Weight (BW)

The parameter BW represents the body weight of potential receptors. This parameter varies within the angling population. The distribution of body weight of the general population of the United States has been fairly well characterized, and, assuming the distribution of body weight for the angling population is similar to the general population of the United States, the distribution of body weight for

the angling population is well characterized. Because this parameter has been extensively studied, there is no significant uncertainty associated with the distribution of body weight. According to EPA (1997), the mean body weights for males of all races between the ages of 18 and 74 years is 78.1 kg, with a standard deviation of 13.5 kg. The mean body weights for females of all races between the ages of 18 and 74 years is 65.4 kg with a standard deviation of 14.6 kg.

ThermoRetec assumed that the distributions of body weights for males and females between the ages of 18 and 74 are truncated normal, with the above referenced means and standard deviations. For males, the body weight was truncated between a minimum of 40 kg and a maximum of 200 kg. For females, the body weight was truncated between a minimum of 35 kg and a maximum of 150 kg. The distribution of the body weight of the angling population was determined by adding up the probability distributions for males and females between 18 and 74 years of age with equal weight (i.e., 50% each).

Selected statistical measures of the distribution thus obtained are presented in Table 2-10. This table shows that the mean body weight for the potentially exposed population is 72.1 kg, and the 5% and 95% percentiles are 56.8 kg and 88.2 kg, respectively. In their evaluation Exponent (2000) used a fixed body weight of 70 kg.

2.3.7 Averaging Time (ATc and ATnc)

The averaging time for estimating the daily intake averaged over a lifetime, (ATc) is used in the calculation of cancer risks. Exponent (2000) used 70 years, while ThermoRetec (2000) used 75 years (EPA, 1997).

The averaging time for estimating the daily intake averaged over the exposure period (ATnc) is used in the calculation of noncancer hazard indices. The exposure period is equal to the exposure duration (converted from years to days) in this evaluation.

2.4 Distributions for Fish Intake Rate and Exposure Frequency (IR and EF)

This subsection discusses the distributions for fish ingestion rate (grams of fish consumed per meal or per day) and exposure frequency (meals per year or days per year) used in this analysis. Estimates of these distributions were obtained from the following studies:

- Recreational Angler
  - West et al. (1989);
  - West et al (1993);
  - Fiore et al. (1989); and
  - Exponent (2000).

- High Intake Fish Consumers
  - low income minorities from West et al. (1993);
  - Hmong for all fishing sources from Hutchison and Kraft (1994) and Hutchinson(1994); and
  - Hmong for the Lower Fox River only from Hutchison (1999).

The following subsections discuss the data presented in each of these studies and the assumptions used by ThermoRetec and Exponent (2000).

2.4.1 Recreational Anglers

West (1989)

EPA (1997) presents distributional data derived from the West et al. (1989) study. IR is presented as a probability distribution of the average daily ingestion rate (in g/day) over the course of a year. As such, EF is taken as 365 days. West et al. (1989) provide data on the quantity of fish consumed by only those anglers who eat sport caught fish and indicate that 16% of all the anglers surveyed did not eat any fish. The probability distribution of fish intake rate is calculated by multiplying the distribution of all anglers that eat sport-caught fish by the distribution of fish intake rate for the anglers who eat such fish. The data included in the distributions used for these calculations and the statistics of the resulting ingestion rate distribution are presented in Table 2-11.

West et al. (1993)

SAIC (1995) developed a probability distribution for the annualized intake rate (g/day) for all anglers in the West et al. (1993) study. This distribution is presented in Table 2-12. EF is taken as 365 days.

Fiore et al. (1989)

EPA (1997) presents distributional data on the number of meals per year for Wisconsin anglers who eat fish based on the study by Fiore et al. (1989). It should be noted that, based on a conversation with Jackie Moya of the EPA, the percentile data presented in Table 10-70 of EPA (1997) refers to the population of anglers who eat fish. In contrast, the mean annual number of sport caught meals presented in that table (18 meals) refers to the whole population of anglers. It is stated in EPA (1997) that 91% of the angler population eat sport caught fish.

As such, for 9% of the angler population, the intake rate is zero. The exposure frequency distribution for all anglers is obtained by multiplying the exposure frequency distribution for sport anglers who eat fish by the distribution of recreational anglers who eat such fish. It should also be noted that, in order for the mean number of meals for all recreational anglers to match the reported value of 18 meals/year, it was necessary to set the maximum number of meals to 140 per year, rather than the 365 meals/year presented in Table 10-70 of EPA (1997). The 365 meals per year is interpreted as the maximum theoretical yearly number of meals. These distributions are presented in Table 2-13 along with the statistics of the resulting distribution for EF. The fish ingestion rate (IR) is taken as 227 g/meal.

Exponent (2000)

Exponent (2000) estimated the fish intake rate (IR) and exposure frequency (EF) based on an angler survey of the Lower Fox River conducted by Triangle Economic Research, Inc. Tables 2-14 and 2-15 present the parameters for these two distributions.

It should be noted that this survey data was not adjusted to account for the influence of fish advisories on angler behavior. The survey by Hutchison (1999) indicated that anglers who fish from the Lower Fox River have altered their behavior based on fish advisories. Thus, the Exponent (2000) distribution for EF represents a lower bound estimate of fish ingestion rates for the scenario where there are no fish advisories.

2.4.2 High Intake Fish Consumers

West et al. (1993)

SAIC (1995) developed a cumulative distribution for the annualized intake rate for low income minority anglers in the West et al. (1993) study. This distribution is presented in Table 2-16. EF is taken as 365 days/year.

Hutchison (1994) and Hutchison and Kraft (1994)

Hutchison and Kraft (1994) provide distributional data on the number of meals of sport-caught fish consumed by Hmong anglers from all fishing locations. This information is presented in Table 2-17. Hutchison and Kraft (1994) did not quantify the meal size, but Hutchison (1994), in a study of Hmong anglers in the Sheboygan, Wisconsin area, developed distributional data on meal size. This distributional data is presented in Table 2-18.

Hutchison (1999)

Hutchison (1999) provides distributional data on the number of meals of sport-caught fish consumed by Hmong/Laotian anglers from the Lower Fox River in the city of Green Bay. This distributional data is presented in Table 2-19. No information is presented in Hutchison (1999) on meal size. The meal size was taken as 227 g/meal in the analysis presented in this appendix.

Hutchison (1999) surveyed anglers who fish from the Lower Fox River and determined the amount of fish they consume from the Lower Fox River. Hutchison (1999) also asked anglers if they were aware of the fish advisories on the river and if their fishing behavior had been modified by these advisories. Many anglers indicated that they were aware of the fish advisories and that their behavior had been modified. The results of the Hutchison (1999) survey presented in Table 2-19 have not been adjusted to account for the influence of the fish advisories. Thus, the distribution in Table 2-19 for EF represents a lower bound estimate of fish ingestion rates for the scenario where there are no fish advisories.

2.4.3 Evaluation of Uncertainty and Variability

As discussed above, different studies produced different results for the distributions of IR and EF, and, therefore, for the distribution of the product IR*EF (g/year), which represents the grams of fish ingested by anglers over the course of a year. Each distribution is representative of variability associated with IR and EF.

The fact that different distributions were obtained by different researchers, however, is representative of the fact that, in addition to variability, uncertainty is also associated with the estimation of the quantity IR*EF. Consistent with draft EPA guidance (EPA, 1999), separate risk calculations are performed in Section 4 of this appendix, based on each of the studies discussed in Sections 2.4.1 and 2.4.2. The results of these separate calculations are then used to provide a quantitative estimate of the confidence of the estimates of risks and hazard indices (Section 4).

3 Comparison of Exponent Assumptions and ThermoRetec Assumptions

This section presents a comparison of the assumptions used in the evaluation of risks and hazard indices in Exponent (2000) and the focused evaluation presented in the main report. To make the comparison more clear, and eliminate the influence of the assumptions used for fish concentrations, unit risks and unit hazard indices are calculated and compared. Unit risks and unit hazard indices are the risk and hazard index associated with a concentration of 1 mg/kg PCBs in fish.

Risks and hazard indices were calculated in the main report for a Reasonable Maximum Exposure (RME) scenario and Central Tendency Exposure (CTE) scenario for the four reaches of the Lower Fox River and three zones within Green Bay. Different values of risk and hazard index were calculated based on different assumptions regarding intake parameters and concentrations of PCBs in fish. Exponent (2000) used a probabilistic approach to calculate probability distributions of risks and hazard indices over the whole Lower Fox River, independent of the stretch.

High intake fish consumers represent subpopulations of the recreational angler population that are more highly exposed than the general population of recreational anglers. In the main text, ThermoRetec identified three such subpopulations: low-income minorities, Native Americans and Hmong. Exponent (2000) argued that these subpopulations did not eat significantly more fish from the Lower Fox River and Green Bay, so Exponent (2000) did not evaluate exposures and health effects for any subpopulations. Since Exponent (2000) did not explicitly evaluate exposures to high intake fish consumers, a comparison of ThermoRetec and Exponent (2000) results with respect to high intake fish consumers cannot be performed.

The two risk assessments provide different outputs [point value estimates of risks and HIs for RME and CTE scenarios in the main report, and probability distributions of risk and hazard index for Exponent (2000)]. As such, the results of the two risk assessments are not directly comparable. To better understand the fundamental similarities and differences between the two approaches, RME and CTE values were developed from the Exponent (2000) distributions for each intake parameter and unit risks and unit hazard indices were calculated for the RME and CTE scenarios.

Table 3-1 summarizes the intake assumptions and toxicological parameters used in this analysis. Intake assumptions are provided in Table 3-1 for the two studies of Michigan anglers by West et al. (1989) and West et al. (1993); the average of the two West et al. Studies; the study of Wisconsin anglers by Fiore et al. (1989); and the study by Exponent (2000). The values for each parameters in Table 3-1 are the same across the studies with the following exceptions: daily intake rate of fish (IR), exposure frequency (EF), exposure duration (EP) and body weight (BW). The basis for ThermoRetec's assumptions are provided in Sections 5.3 and 5.9 of the main text.

As discussed in Section 2.3.2 of this appendix, this appendix discusses studies that use different approaches to estimate the annual fish intake rate for recreational anglers (i.e., the product IR*EF in g/year). To facilitate the comparison of the fish intake assumptions in the various studies, the annual fish consumption rates were calculated using two common bases. For the first basis, the annual quantity of fish consumed is calculated and divided by 365 days to yield an annualized daily average for IR. This basis is termed Annualized IR in Table 3-1 and EF is constant at 365 days per year. For the second basis, the annual quantity of fish consumed is calculated and divided by an average meal size of 227 g/meal to yield the number of meals of fish per year for EF. This basis is termed Normalized Meals per Year in Table 3-1 and IR is constant at 227 g/meal. This comparison is presented at the bottom of Table 3-1.

The values of IR and EF provided in Table 3-1 for Exponent (2000) were determined as follows. Exponent provides distribution for both IR and EF. These distributions were numerically multiplied together (using Monte Carlo techniques) to yield the distribution of the annual rate of fish consumption and then divided by 365 to give the distribution of the annual rate of fish consumption on a daily basis. The mean of this distribution was selected for the CTE scenario and the 95% value was selected for the RME scenario.

The values for ED provided in Table 3-1 for Exponent (2000) were determined similarly. Exponent (2000) provides a distribution for ED. The mean for this distribution was selected for the CTE scenario and the 95% value was selected for the RME scenario.

The body weights used by ThermoRetec, 71.8 kg, and Exponent (2000), 70 kg, differ, but the differences are so slight that it was a negligible effect on the calculated unit risks and HIs.

In Table 3-1, the reduction factor (RF) is the same for all studies and scenarios, even though the RF developed by Exponent (2000) differs from the RF developed by ThermoRetec.

In their analysis, Exponent (2000) assigned a distribution to the reduction factor (RF). Their reduction factor is based on the overall reduction in mass of PCBs that would be consumed as a result of cooking fish fillets. Exponent used only fillet without skin data in estimating the fish concentration. In the main text, ThermoRetec used mostly skin on fillet data along with some fillet data to estimate their fish concentrations. There is greater reduction in PCB mass associated with the use of skin on fillet data-reduction from trimming as well as cooking. Therefore, to make a more accurate comparison of Exponent's (2000) assumptions to ThermoRetec's assumptions, the ThermoRetec reduction factor of 0.5 was used in the calculations with Exponent (2000) assumptions.

Table 3-1 provides the calculated unit risks and unit hazard indices using ThermoRetec's and Exponent's (2000) assumptions. Unit risks and unit hazard indices are the cancer risks and hazard indices for a total PCB concentration of 1 mg/kg in fish. The highest unit risk and unit hazard index are calculated using the RME and CTE assumptions from West et al. (1993). Table 3-1 also presents the ratio of each unit risk to the unit risk for West et al. (1993) and the ratio of each unit hazard index to the unit hazard index for West et al. 1993. Figure 3-1 plots the unit risks and Figure 3-2 plots the unit hazard indices.

The RME assumptions from West et al. (1989) produced the second highest unit risk and unit hazard index [at 50% of the values using West et al. (1993)]. Similarly, the RME assumptions from Fiore et al. (1989) resulted in unit risk and unit hazard index values of 47.8% of the value using West et al. (1993). The unit risk and unit hazard index calculated using the RME assumptions from the Exponent(2000) evaluation were 22% of the values using West et al. (1993). While these are the lowest values in the evaluation, they are comparable to the values used by West et al. (1989) and Fiore et al. (1989).

For the CTE scenario, the unit risk and unit hazard index values for West et al. (1989) and Fiore et al. (1989) are 71% and 66% of the values for West et al. (1993). The Exponent (2000) unit risk value is 15% of the CTE value from West et al. (1993). The unit hazard index is 28% of the value for West et al. (1993). These values are lower than the values from Fiore et al. (1989). These lower values are mostly due to a lower value for exposure duration (15 years) used by Exponent (2000) as compared to the value of 30 years used by ThermoRetec.

In conclusion, a comparison of intake assumptions used by Exponent (2000) and ThermoRetec (in the main report) indicates that Exponent (2000) intake assumptions result in a generally lower unit risk and unit hazard index than the assumptions used by ThermoRetec. The difference between the unit risks and unit hazard indices calculated by Exponent (2000) and ThermoRetec depends on the study used to estimate fish intake assumptions. This difference is generally greatest for the West et al. (1993) study and least for the Fiore et al. (1989) study.

4 Probabilistic Evaluation of Exposure to PCBs in Fish

This section presents the results of probabilistic calculations for cancer risks and hazard indices for three data sets characterizing PCB concentrations in fish. These data sets are the Exponent (2000) distribution of PCB concentrations in fish for the entire Lower Fox River, the distribution of PCB concentrations in the Little Lake Butte des Morts reach and the distribution of PCB concentrations in the De Pere to Green Bay reach.

4.1 Results Using Fish Concentration Distribution from Exponent (2000)

Table 4-1 summarizes the intake assumptions used for the recreational anglers utilizing the fish concentration data from Exponent (2000). Four separate calculations were performed, based on different intake assumptions. Table 4-2 presents the cancer risk and hazard index distributions resulting from the probabilistic calculations, as well as the CTE and RME values calculated using intake parameters for the CTE and RME scenarios and the mean concentration for the distribution of PCB concentrations in fish from Exponent (2000). Figures 4-1 through 4-4 provide the cumulative distributions for cancer risks and also show the mean cancer risk from the simulation as well as the cancer risks based on CTE and RME assumptions. Figures 4-5 through 4-8 provide analogous information on the distribution of hazard indices. In general, the mean of the risk and hazard indices probability distributions and CTE risks and hazard indices are very similar. The RME risks and hazard indices occur within the 85^{th} to 99^{th} percentiles, with most between the 90^{th} and 95^{th} percentiles.

Table 4-3 summarizes the intake assumptions used for the high intake fish consumers utilizing the fish concentration data from Exponent (2000). Three separate calculations were performed, based on the different intake assumptions. Table 4-4 presents the cancer risk and hazard index distributions resulting from the probabilistic calculations. Figures 4-9 through 4-11 provide the cumulative distributions for cancer risks, while Figures 4-12 through 4-14 provide analogous information on the distribution of hazard indices. In general, the mean of the risk and hazard indices probability distributions and CTE risks and hazard indices are very similar. The RME risks and hazard indices occur within the 90^{th} to 98^{th} percentiles, with most between the 90^{th} and 95^{th} percentiles.

4.2 Results Using Fish Concentration Distribution for the Little Lake Butte des Morts Reach

Table 4-5 summarizes the intake assumptions used for the recreational anglers utilizing the fish concentration data from Little Lake Butte des Morts. Four separate calculations were performed, based on different intake assumptions. Table 4-6 presents the cancer risk and hazard index distributions resulting from the probabilistic calculations. Figures 4-15 through 4-19 provide the cumulative distributions for cancer risks and Figures 4-19 through 4-22 provide analogous information on the distribution of hazard indices. In general, the mean and CTE risks and hazard indices are very similar. The RME risks and hazard indices occur within the 90th to 99th percentiles, with most between the 94th and 98th percentiles.

Table 4-7 summarizes the intake assumptions used for the high intake fish consumers utilizing the fish concentration data from Little Lake Butte des Morts. Three separate calculations were performed, based on different intake assumptions. Table 4-8 presents the cancer risk and hazard index distributions resulting from the probabilistic calculations. Figures 4-23 through 4-25 provide the cumulative distributions for cancer risks and Figures 4-26 through 4-28 provide analogous information on the distribution of hazard indices. In general, the mean of the risk and hazard index probability distributions and CTE risks and hazard indices are very similar. The RME risks and hazard indices occur within the 90th to 98th percentiles, with most between the 90th and 95th percentiles.

4.3 Results Using Fish Concentration Distribution for the De Pere to Green Bay Reach

Table 4-9 summarizes the intake assumptions used for the recreational anglers utilizing the fish concentration data from the De Pere to Green Bay. Four separate calculations were performed, based on different intake assumptions. Table 4-10 presents the cancer risk and hazard index distributions resulting from the probabilistic calculations. Figures 4-29 through 4-32 provide the cumulative distributions for cancer risks and Figures 4-33 through 4-36 provide analogous information on the distribution of hazard indices. In general, the mean and CTE risks and hazard indices are very similar. The RME risks and hazard indices occur within the 90th to 99th percentiles, with most between the 94th and 98th percentiles.

Table 4-11 summarizes the intake assumptions used for the high intake fish consumers utilizing the fish concentration data from the De Pere to Green Bay. Three separate calculations were performed, based on different intake

assumptions. Table 4-12 presents the cancer risk and hazard index distributions resulting from the probabilistic calculations. Figures 4-37 through 4-39 provide the cumulative distributions for cancer risks and Figures 4-40 through 4-42 provide similar information on the distribution of hazard indices. In general, the mean of the risk and hazard index probability distributions and CTE risks and hazard indices are very similar. The RME risks and hazard indices occur within the 94th to 98th percentiles, with most between the 95th and 98th percentiles.

4.4 Comparison of Probabilistic Results with CTE and RME Values

As pointed out in Sections 4.1 through 4.3, the CTE and RME values of risk and hazard indes calculated in the main report are generally close to the mean and 95% values of the respective probability distributions. This is consistent with the interpretation provided in EPA (1999) of the RME value as corresponding to the 90th to 99th percentile of the risk and hazard indices distributions, and being representative of the high-end range of risk and hazard index. Figures 4-43 through 4-48 present an explicit comparison of CTE and RME values with the probability distribution data. These figures provide a visual means for evaluating the position of the CTE and RME values with respect to the probability distribution generated in the probabilistic analysis, and confirm the observations provided in Sections 4.1 through 4-3 that the CTE values generally correspond to the means of the distributions, and the RME values are generally at the high end (90th to 99th percentiles).

4.5 Interpretation of Results

Probabilistic calculations of risk and hazard index were performed in this appendix for the following cases:

- Entire Fox River
  - Recreational Anglers
    - West et al., 1989
    - West et al., 1993
    - Fiore et al., 1989
    - Exponent, 2000
  - High Intake Fish Consumers
    - Low Income Minorities, West et al., 1993
    - Hmong, Hutchison, 1994 and Hutchison & Kraft, 1994
    - Hmong/Laotians, Hutchison, 1999

- Little Lake Butte des Morts Reach
  - Recreational Anglers
    - West et al., 1989
    - West et al., 1993
    - Fiore et al., 1989
    - Exponent, 2000

  - High Intake Fish Consumers
    - Low Income Minorities, West, et al., 1993
    - Hmong, Hutchison, 1994 and Hutchison & Kraft, 1994
    - Hmong/Laotian, Hutchison, 1999

- De Pere to Green Bay Reach
  - Recreational Anglers
    - West et al., 1989
    - West et al., 1993
    - Fiore et al., 1989
    - Exponent, 2000

  - High Intake Fish Consumers
    - Low Income Minorities, West, et al., 1993
    - Hmong, Hutchison, 1994 and Hutchison & Kraft, 1994
    - Hmong/Laotian, Hutchison, 1999

As discussed in Section 2, for each of the above cases, some of the parameters relevant to the calculation of risk and hazard index are characterized by variability. As such, the calculated risks and hazard indices reflect variability in exposure and are specified as probability distributions rather than single values. These probability distributions of risk and hazard indices are presented in Tables 4-2, 4-4, 4-6, 4-8 and 4-10. These distributions do not reflect uncertainty in the input parameters. In the terminology used in the draft EPA guidance on probabilistic risk assessment (EPA, 1999), these distributions are the result of a one-dimensional probabilistic risk analysis.

The above referenced tables (and associated figures showing cumulative risks and hazard index distributions) explicitly provide the probability of a specific risk or hazard index for an individual from the exposed population based on a set of assumptions. For example, from Table 4-6, it can be seen that based on the assumptions presented in West et al. (1989) for a recreational angler, there is a 50% probability that an angler has a cancer risk less than or equal to 2.7×10^{-5}, and has an associated noncancer hazard index of less than or equal to 2.8. Similarly, using the same probability distribution, there is a 95% probability that the same

recreational angler has a cancer risk less than or equal to $3.1x10^{-4}$, has an associated noncancer hazard index of 13. All columns in Tables 4-2, 4-4, 4-6, 4-8 and 4-10 can be read in the same way.

Tables 4-2, 4-4, 4-6, 4-8 and 4-10 (and associated figures showing cumulative risk and hazard index distributions) can also be used to answer the question *what is the probability that the risk or hazard index for an exposed individual will exceed a specified level?* Using again the data in Table 4-6, based on the West et al. (1989) intake assumptions there is a probability between 20% and 25% that the risk to an exposed angler is less than or equal to $1x10^{-6}$. Also, there is a probability of just over 35% that the risk exceeds $1x10^{-5}$. Conversely, there is a greater than 75% probability that the risk is less than or equal to $1x10^{-4}$. This means that there is a less then 25% probability that the risk exceeds $1x10^{-4}$. All columns in Tables 4-2, 4-4, 4-6, 4-8 and 4-10 can be read in the same way.

4.6 Evaluation of Uncertainty

As previously indicated, the probability distributions discussed above do not reflect the fact that uncertainty is associated with some of the input exposure parameters. For example, there is uncertainty in the assumptions used to estimate fish intake rates for recreational anglers and high intake fish consumers. This is reflected in the fact that, as discussed in Section 2.4, different studies provide different probability distributions for the ingestion rate (IR) and the exposure frequency (EF) for the same populations (recreational anglers and high intake fish consumers). In this subsection, a procedure consistent with draft EPA guidance for probabilistic risk assessment (EPA, 1999) is used to estimate the uncertainty associated with the risk and hazard index calculations for recreational anglers and high intake fish consumers in the three portions of the Fox River considered [whole river (Exponent, 2000), Little Lake Butte des Morts reach, and De Pere to Green Bay reach].

Figure 4-49 and 4-50 show the cumulative probability distributions of risk and hazard index to recreational anglers, based on the Exponent (2000) fish concentration distribution for the whole river. The results for the four set of studies used (reflecting four different set of intake assumptions) are shown on the same graph. It should be noted that the assumptions for IR and EF are the only differences among the four curves shown. Figures 4-51 and 4-52 show the probability distributions for high intake fish consumers, based on the three studies (and intake assumptions) used. Figures 4-53 through 4-60 present analogous information for the Little Lake Butte des Morts reach and De Pere to Green Bay reach.

It should be noted that the three studies used of anglers to evaluate high intake fish consumers do not evaluate the same populations, although they are still

representative of the same category of anglers. The low income minority anglers surveyed by West et al. (1993) probably include very few Hmong or Laotians. The fishing behavior of Hmongs characterized by Hutchison and Kraft (1994) is for all fishing, while the fishing behavior of Hmongs and Laotions characterized by Hutchison (1999) is for fishing only from the Lower Fox River in the city of Green Bay. Thus, the results presented in Figures 4-51 and 4-52, 4-55 and 4-56, and 4-59 and 4-60 should be interpreted with these distinctions in mind.

Inspection of these figures reveals that different values of risk (and hazard index) are calculated, based on each study, for a given percentile. For example, based on Figure 4-49, the 90% risk value ranges between less than 10^{-5} based on Exponent (2000), and about than 10^{-4} based on West et al. (1993). Similarly the 50% risk value ranges between less than $2x10^{-6}$ based on Exponent (2000) and $1x10^{-5}$ based on West et al. (1989).

Thus, for each percentile value of risk and hazard index, a range (rather than a single value) was estimated, reflecting the fact that there is uncertainty in the exposure assumptions. Figures 4-61 through 4-72 present a graphical evaluation of the uncertainty in the variability statistics in a format consistent with the format recommended in EPA (1999). In these figures, the calculated range for the mean and selected percentiles is plotted on the vertical axis for the three portions of the river evaluated, and for the two receptor categories (recreational anglers and high intake fish consumers). The data presented in Figures 4-61 through 4-72 is summarized in Tables 4-13 through 4-15.

The following should be noted.

- In Figures 4-61 through 4-72, some of the lower percentile values have a risk and HI of zero (this is due to the fact that under some of the assumptions, some percent of the potentially exposed population does not eat fish, and therefore is not exposed to PCBs through the fish ingestion pathway). Since the risk data is plotted on a logarithmic vertical scale, a value of zero cannot be plotted. In these cases, a value of 1E-08, corresponding to the lowest value included on the vertical axis is plotted. This problem does not arise for the plots of hazard index, as the vertical scale is linear in these plots.

- For each percentile value, average risks and hazard indices are calculated, representing the arithmetic average of the values for each study utilized (four values for the recreational angler, and three values for the high intake fish consumer). This means, essentially, that each study is assigned the same weight in the uncertainty evaluation. A

more detailed statistical evaluation of the data used to generate the probability distribution excerpted for each study might indicate that non-uniform weights could be assigned to the data generated in the studies. However, such statistical evaluation is beyond the scope of the analysis presented in this appendix. As such, the equal weight assumption is used in this evaluation.

The information presented in Figures 4-61 through 4-72 and Tables 4-13 through 4-15 can be used to provide a quantitative estimate of each percentile value for risk and HI, and of the confidence in the estimate. For example, Based on the data presented in Table 4-14, the best estimate of the mean value of risk to recreational anglers in the Little Lake Butte des Morts reach is 6.5×10^{-5}. However, this value could be as low as 1.4×10^{-5}, and as high as 10^{-4}. The additional data in Figures 4-61 through 4-72 and Tables 4-13 through 4-15 can be interpreted in the same manner. The data presented in these tables and figiures show that the uncertainty in the estimate of the probability distributions of risk and hazard indices is moderate, as reflected by the fact that the minimum and maximum values for the selected statistical parameters are generally within a factor of 10 of each other.

4.7 Sensitivity Analysis

A qualitative sensitivity analysis was performed to understand how the variability in the various input parameters specified as probability distributions affects the calculated variability of risk and hazard index The starting point for this analysis are the equations used for risk and hazard index, which were discussed in Section 2.1, and are reproduced below (all variables have been previously defined).

Risk is calculated according to the following equation:

$$R = Cfish \cdot IntFacC \cdot CSFo$$

where *IntfacC* represents the intake factor for cancer risk [$(mg/kg)^{-1}$], given by:

$$IntFacC = \frac{IR \cdot RF \cdot ABS \cdot CF \cdot EF \cdot ED}{BW \cdot ATc}$$

Combining the two equations yields the following expression for risk:

$$R = Cfish \cdot \frac{IR \cdot RF \cdot ABS \cdot CF \cdot EF \cdot ED}{BW \cdot ATc} \cdot CSFo$$

Similarly, the expression for hazard index is:

$$HI = Cfish \cdot \frac{IR \cdot RF \cdot ABS \cdot CF \cdot EF \cdot ED}{BW \cdot ATnc \cdot RfDo}$$

Using basic concepts from calculus, small variations in risk can be written (Young, 1968):

$$\frac{\Delta R}{R} = \frac{\Delta Cfish}{Cfish} + \frac{\Delta IR}{IR} + \frac{\Delta RF}{RF} + \frac{\Delta ABS}{ABS} + \frac{\Delta CF}{CF} + \frac{\Delta EF}{EF} + \frac{\Delta ED}{ED} + \frac{\Delta BW}{BW} + \frac{\Delta AT_c}{AT_c} + \frac{\Delta C}{C}$$

Where ΔR represents a small variation in risk, ΔIR a represents small variation in daily fish ingestion rate, and similarly for all other variables. An analogous expression can be written for ΔHI, but this is not done here to simplify the discussion, and the following discussion is restricted to risk. The results for risk can be easily extended to hazard index the calculation of hazard index.

To understand qualitatively how variations in each parameter on the right hand side of the equation above affect relative variations in the magnitude of risk, the following approximations is made: $\Delta IR \sim \sigma_{IR}$, where σ_{IR} represents the standard deviation of the probability distribution of IR (the fish ingestion rate). In addition, the mean value is taken as representative of each variable. Analogous approximations are made for all other variables entering the calculation of risk. Using these approximations, the relative variation in the magnitude of risk can be written as:

$$\frac{\Delta R}{R} \approx \frac{\sigma_{Cfish}}{Cfish} + \frac{\sigma_{IR}}{IR} + \frac{\sigma_{RF}}{RF} + \frac{\sigma_{ED}}{ED} + \frac{\sigma_{BW}}{BW}$$

Similarly, it can be shown that the relative variation of HI can be written as:

$$\frac{\Delta HI}{HI} \approx \frac{\sigma_{Cfish}}{Cfish} + \frac{\sigma_{IR}}{IR} + \frac{\sigma_{RF}}{RF} + \frac{\sigma_{BW}}{BW}$$

It should be noted that in the risk and HI equations *CF, ABS, ATc, EF* and *CSFo* are taken as point values (i.e., their standard deviation is zero); as such their standard deviation is zero, and their respective terms disappear from the equations. In addition, the terms associated with ED and AT_{NC} disappear from the hazard index equation above because they are taken to be equal, and therefore cancel out.

The above equations provides qualitative tools to evaluate the effect of variability of each input variable on the resulting calculated risk and HI. Inspection of these equations reveals that the variables with the greatest effect on the variability of risk and hazard index are the ones with the greatest relative variability, i.e., those whose relative standard deviation (i.e., the ratio of standard deviation to mean value) is greatest.

Tables 4-16 and 4-17 present an explicit evaluation of the relative effect of each variable in the calculations of risk and HI for recreational anglers and high intake fish consumers for two selected studies for the Little Lake Butte des Morts reach. Analysis of the other studies would yield qualitatively similar results.

The results presented in Table 4-16 indicate that the variability in the risk calculations is mostly due to the variability of two parameters, namely IR (g/day), the fish ingestion rate, and ED (years), the exposure duration. Variability in all other parameters is essentially negligible. Similarly, Table 4-17 indicates that the variability in hazard index is due essentially in its entirety to the variability in IR. In addition, a comparison of the relative standard deviations for the calculated risks and hazard indices with the sum of the relative standard deviations of all variable parameters, indicates that the two quantities are relatively close (based on the analysis discussed above, these quantities should be essentially equal). This indicates that the assumptions used to derive the equations used for the sensitivity analysis are reasonable ones.

4.8 Conclusions

A probabilistic risk assessment of exposure to PCBs in fish was performed, and is documented in Section 4. Consistent with EPA guidance (EPA, 1999), the probabilistic risk assessment included an evaluation of both variability and uncertainty. The most significant findings of the focused probabilistic risk assessment are as follows.

- The deterministic CTE estimates of risk and hazard index provided in the main report are generally close to the means of the respective probability distributions of risk and hazard index. This is consistent

with the interpretation of the CTE as the average risk or hazard index for the exposed population.

- The deterministic RME estimates of risk and hazard index provided in the main report are generally in the range of the 90th to 95th percentiles of the respective and HI probability distributions of risk and hazard index. This is consistent with the interpretation provided in EPA (1999) of the RME as a plausible high end risk or hazard index for the exposed population.

- The uncertainty in the estimate of the probability distributions of risk and hazard index due to uncertainty in the fish ingestion rate is moderate, and the minimum and maximum values of selected statistical parameters are generally within a factor of 10 of each other.

5 References

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Appendix B2

General Statistics

1.0 Introduction

This appendix provides statistical summaries of concentration data for fish, waterfowl, surface water and sediment. Section 2 provides statistical summaries of data collected from these media. Section 3 presents a statistical summary of total PCB concentrations in sediments based on interpolation of the sediment analytical data onto a grid. Section 4 provides references.

2.0 Statistical Summary of Analytical Data

Tables 1 through 9 present statistical summaries for constituents in Fish, Waterfowl, Unfiltered Surface Water (total surface water), Filtered Surface Water (dissolved surface water), and Sediment respectively. Each table provides statistical information in a particular category for the following five reaches of Fox River, Little Lake Butte des Morts, Appleton to Little Rapids, Little Rapids to De Pere, De Pere to Green Bay, and Green Bay.

A. Sample Counts. In this category, the following statistical information is provided:

- number of samples
- number of detects
- number of nondetects
- percent nondetects

The percent nondetects is calculated as follows (assuming the number of valid samples is greater than zero):

$$PercNonDet = 100 \cdot \frac{NonDet}{NumSamp}$$

where

PercNonDet = percent of nondetects
NonDet = number of nondetects
NumSamp = number of samples

B. Basic Statistics. In this category, the following statistical information is provided:

- minimum detection limit
- maximum detection limit
- minimum detected concentration
- maximum detected concentration

C. General Summary Statistics. This category provides a variety of summary statistics, including:

- average or mean
- standard deviation
- coefficient of variation
- geometric mean
- geometric standard deviation

In calculating all these summary statistics, nondetects are replaced with half the detection limit.

The median is the concentration at the middle of a sorted list of samples. If the number of samples is odd, the median is the concentration of the middle sample. If the number of samples is even, the median is the average of the concentrations of the two samples in the middle of the list.

The average, x_{avg}, is given by:

$$x_{avg} = \frac{\Sigma\, x_i}{n}$$

where

x_i = the value of sample number *i*; and
n = number of samples.

The standard deviation is the sample standard deviation, s, given by:

$$s = \sqrt{\Sigma \frac{(x_{avg} - x_i)^2}{(n - 1)}}$$

The coefficient of variation, CoefVar, is given by:

$$CoefVar = \frac{s}{x_{avg}}$$

The geometric mean and geometric standard deviation are calculated as follows:

- The data is logarithmically transformed using the natural logarithm (ln).

- The average and standard deviation are calculated for the transformed data, $x_{t\text{-}avg}$ and s_t, respectively, using the equations above.
- The geometric mean, x_{gmean}, and geometric standard deviation, s_g, are calculated by transforming back $x_{t\text{-}avg}$ and s_t, as follows:

$$x_{gmean} = e^{x_{t-avg}}$$
$$s_g = e^{s_t}$$

D. Testing of Normality of Data. In this category, the data is tested to determine if it is represented by a normal distribution. One of two tests is employed. If there are 50 samples or less, the Shapiro-Wilk test of normality is utilized (Shapiro and Wilk, 1965). Using the procedures outlined in Gilbert (1987), the data is sorted and manipulated to calculate a W test statistic. This W-statistic was compared to a W value at a 0.05 quantile. The W value at the 0.05 quantile is found by referring to a lookup table (see Table A7 in Appendix A of Gilbert (1987)). If the W-statistic is greater than or equal to the W value, the data is considered to be normally distributed.

If there are more than 50 samples, the D'Agostino test of normality is utilized (D'Agostino, 1971), which is a two tailed statistical test. Using the procedures outlined in Gilbert (1987), the data is sorted and manipulated to calculate the Y test statistic. For a test of normality at the 0.05 level of significance, the Y values at the 0.025 quantile, $Y_{0.025}$, and 0.975 quantile, $Y_{0.975}$, are determined by interpolating from a lookup table (e.g., Table A8 in Appendix A of Gilbert (1987)). The data is considered to be normally distributed if the Y statistic satisfies the following condition:

$$Y_{0.25} \leq Y\text{-}statistic \leq Y_{0.975}$$

E. Testing of Log-Normality of Data. In this category, the data is tested to determine if it is represented by a log-normal distribution. The data is transformed by taking the natural logarithm of each sample value. The procedures described previously are then applied to the transformed data. If there are 50 samples or less, the Shapiro-Wilk test of normality is used. If there are more than 50 samples, the D'Agostino test of normality is utilized.

F. Source Concentrations. In this category, the source concentration is calculated following USEPA (1992) guidance. First, the 95% upper confidence limit (UCL) on the mean, which depends on the distribution type, is calculated.

For normally distributed data, the 95% UCL on the mean, UCL_{norm}, is calculated with the following equation (USEPA, 1992):

$$UCL_{norm} = x_{avg} + \frac{t \cdot s}{\sqrt{n}}$$

The one tail t-statistic at a 95% level, t, depends on the number of samples, n, and the standard deviation of the log-transformed data, s_t, and comes from Table A2 in Appendix A of Gilbert (1987).

For log-normally distributed data, the 95% UCL on the arithmetic mean, UCL_{ln}, is calculated with the following equation (USEPA, 1992):

$$UCL_{ln} = e^{(x_{t-avg} + 0.5 \cdot s_t^2 + s_t \cdot H/\sqrt{n-1})}$$

The one tail H-statistic at a 95% level, H, depends on the number of samples, n, and the standard deviation of the log-transformed data, s_t, and comes from Table A12 in Appendix A of Gilbert (1987).

The source concentration is established as the 95% UCL on the mean or the maximum detected concentration, whichever is lower (USEPA, 1992). For data which is nonparametric (i.e., neither normally nor log-normally distributed), the source concentration is established as the greater of the two 95% UCLs (one assuming the data is normally distributed, the other assuming the data is lognormally distributed). If the higher of the two 95% UCLs exceeds the maximum detected concentration, the maximum detected concentration is the source concentration. In this evaluation, if there were more than 15% nondetects, the data was assumed to be nonparametric.

The last two columns in this section provide the adjusted average concentration and the upperbound concentration. The adjusted average concentration was determined as follows. If there were no detects, the adjusted average concentration is ND. If there were detects, the average concentration is the minimum of the average concentration or the maximum detected concentration. All values have units of either mg/kg (for fish, waterfowl and sediment) or mg/L (for surface water). The upperbound concentration was determined with a similar procedure. If there were no detects, the upperbound concentration is ND. If there were detects, the upperbound concentration is the source concentration. All values have units of either mg/kg (for fish, waterfowl and sediment) or mg/L (for surface water).

3.0 Statistical Summary of Interpolated Sediment Data

The analytical results for total PCBs in sediment were compiled, a grid was imposed over each reach of the Lower Fox River and each zone of Green Bay and the analytical data was interpolated to provide a concentration at each point on the grid. The mean and 95% UCL on the mean were determined for each reach and each zone using the data at each grid point. Since there are a large number of grid points for each reach and zone

(at least 9,000) the 95% UCL on the mean was calculated assuming the mean is normally distributed. This is consistent with the Central Limit Theorem of statistics (De Groot, 1975).

In calculating values for each point on the grid, interpolations were made only for grid points where there were analytical data nearby. Grid points outside the area with analytical data were assigned a value of -1 to indicate no data was available at these grid points. Three approaches were utilized for handling these grid points with no data when calculating statistics.

In the first approach, all grid points without data were deleted when calculating statistics. In the second approach, all grid points without data were assigned a concentration of 0.1 ug/kg, which is a nominal detection limit for total PCBs. In the third approach, all grid points without data were assigned a concentration of 0 ug/kg.

Table 10 presents summary statistics for the interpolated total PCB data in surface sediment. This table provides the number of samples, the average, 95% UCL on the average assuming the mean is normally distributed and the maximum. Also presented is the adjusted average concentration which is the average converted from units of ug/kg to units of mg/kg and the upperbound concentration which is the 95% UCL on the mean converted from units of ug/kg to mg/kg.

4.0 References

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Appendix B3

Fate and Transport Models and Transfer Factors

1 Introduction

This appendix presents the mathematical models used in the estimation of exposure point concentrations. The following models were used:

- Shower Water to Air Volatilization Model: This model utilizes concentrations in shower water to estimate air concentrations in the bathroom during showering.

- Bath Water to Air Volatilization Model: This model utilizes concentrations in bath water to estimate air concentrations in the bathroom during a bath.

- Surface Water to Air Volatilization Model: This model uses concentrations in surface water to estimate concentrations in outdoor air.

- Sediment to Pore Water Partitioning Model: This model uses concentrations in sediment to estimate concentrations in sediment pore water.

In this analysis, transfer factors were estimated which are the ratio of exposure point concentrations to the source concentrations. These transfer factors can be multiplied by actual source concentrations to produce exposure point concentrations. For the Shower Water to Air Volatilization Model and Bath Water to Air Volatilization Model, the only site-specific data are the water concentrations, so the transfer factors from these models are the same for all areas of interest. For the Surface Water to Air Volatilization Model and Sediment to Pore Water Partitioning Model, the transfer factors use site-specific data, so a separate transfer factor is calculated for each area reach in the Lower Fox River and for Green Bay as a whole.

2 Shower Water to Air Volatilization Model

2.1 Equations

During a shower, chemicals present in shower water are assumed to have the opportunity to volatilize into the air within the shower room. Initially, the concentration of a chemical in the shower room in the air is assumed to be zero. As time goes on, the concentration increases until the shower is finished. At this time, the concentration in the air begins to decrease as air turns over in the room. Figure 2-1 depicts the evolution of the shower room air concentrations over time. The individual taking a shower is assumed to be in the shower room during the shower and for a period after the shower. The relevant exposure point concentration for this individual is the average concentration of the chemical in the air during this exposure event, since the individual is assumed to maintain a constant inhalation rate both during and after the shower.

To estimate this average concentration in the shower room air, the model of Foster and Chrostowski (1987) was used. The model is based on a simple box model of air exchange in the shower room with constant emission of chemicals during the shower. The average concentration in the shower room air is the time weighted sum of the average concentration in the shower room air during the shower and the average concentration in the shower room air after the shower is completed.

$$C_{asav} = \frac{C_{asav1} \cdot T_1 + C_{asav2} \cdot T_2}{T_1 + T_2}$$

where:

C_{asav} = average concentration of chemical in the shower room air (mg/m^3);
C_{asav1} = average concentration of chemical in the shower room air during showering (mg/m^3);
C_{asav2} = average concentration of chemical in the shower room air after showering (mg/m^3);
T_1 = duration of shower (hours); and
T_2 = period individual is in shower room after shower (hours).

Both C_{asav1} and C_{asav2} depend upon the concentration of the chemical in the shower water and this relationship can be expressed through a transfer factor such that the following relationships and transfer factors can be defined:

$$C_{asav1} = TF_{sh1} \cdot C_{ws}$$

$$C_{asav2} = TF_{sh2} \cdot C_{ws}$$

$$C_{asav} = TF_{sh} \cdot C_{ws}$$

$$TF_{sh} = \frac{TF_{sh1} \cdot T_1 + TF_{sh2} \cdot T_2}{T_1 + T_2}$$

where:

C_{ws} = concentration of chemical in the shower water (mg/L);
TF_{sh1} = transfer factor describing relationship between concentration in shower air and shower water during showering (L/m^3);
TF_{sh2} = transfer factor describing relationship between concentration in shower air and shower water after showering (L/m^3); and
TF_{sh} = overall transfer factor describing overall relationship between concentration in shower air and shower water (L/m^3).

The variable TF_{sh1} is determined by first developing an expression for C_{asav1}. This variable is determined by solving the following equation:

$$C_{asav1} = \frac{\int_0^{T_1} C_{as}(t)\ dt}{T_1}$$

where:

$C_{as}(t)$ = concentration of chemical in the shower room air over time (mg/m^3).

The variable $C_{as}(t)$ can be determined by solving the following differential equation:

$$Vsh \cdot \frac{d}{dt} C_{as}(t) = rsh - Qash \cdot C_{as}(t)$$

where:

Vsh = volume of shower room (m^3);
rsh = rate of chemical emission into shower room (mg/hr); and
Qash = rate of air flow out of shower room (m^3/hr).

This equation states that the rate at which the mass of chemical in the shower room changes depends on the difference in the rate at which the chemical is volatilized from the shower water minus the rate at which the chemical leaves the shower room as air circulates through the shower room.

The rate at which the chemical is introduced into the shower room with shower water is the flow rate of the shower water times the concentration of the chemical in the water. Only a fraction of the chemical so introduced is volatilized, however, before the water drains out of the shower. Thus, the rate at which the chemical is introduced into the shower room is given by:

$$rsh = fv \cdot Qwsh \cdot C_{ws}$$

where:

fv = fraction of chemical volatilized; and
Qwsh = shower water flow rate (L/hr).

The differential equation describing the change in C_{as} over time becomes:

$$\frac{d}{dt} C_{as} = \frac{fv \cdot Qwsh}{Vsh} \cdot C_{ws} - \frac{Qash}{Vsh} \cdot C_{as}$$

By defining two rate constants, *ksw* and *ksa*, this equation can be restated:

$$\frac{d}{dt} C_{as} = ksw \cdot C_{ws} - ksa \cdot C_{as}$$

The constants *ksw* and *ksa* are defined as:

$$ksw = \frac{fv \cdot Qwsh}{Vsh}$$

where:

$$ksa = \frac{Qash}{Vsh}$$

ksw = first order rate constant describing release of chemical from shower water to air (L/m^3-hr); and

ksa = first order rate constant describing turnover of air in shower room (1/hr).

The solution to this differential equation is:

$$C_{as}(t) = \left(\frac{ksw}{ksa}\right) C_{ws} \left(1 - e^{-ksa \cdot t}\right)$$

The average concentration in the shower room during showering is given by:

$$C_{asav1} = \left(\frac{ksw}{ksa}\right) C_{ws} \left[1 - \left(\frac{1}{ksa \cdot T_1}\right) \left(1 - e^{-ksa \cdot T_1}\right)\right]$$

The transfer factor TF_{sh1} is given by:

$$TF_{sh1} = \left(\frac{ksw}{ksa}\right) \left[1 - \left(\frac{1}{ksa \cdot T_1}\right) \left(1 - e^{-ksa \cdot T_1}\right)\right]$$

In order to solve this equation, the parameter *fv* must be determined. Foster and Chrostowski (1987) estimated *fv* by assuming the shower water atomizes into droplets and considering the rate of volatilization from a droplet and the time of descent for the droplet. Their expression for *fv* is:

$$fv = 1 - \exp\left(\frac{-kao \cdot tdr}{60 \cdot d}\right)$$

where:

kao = overall mass transfer coefficient (cm/hr);
tdr = shower droplet drop time (sec); and
d = droplet diameter (mm).

The term *kao/(60d)* combines both the rate of transfer and the available interfacial area across which volatilization can occur. The value *1/(60d)* equals the specific interfacial area, *6/d*, for a spherical shower droplet of diameter *d* multiplied by conversion factors (hr/3600 sec and 10 mm/cm). The overall mass transfer coefficient, *kao*, is based on an ambient overall mass transfer coefficient, *ko*, that is adjusted for the higher shower water temperature.

$$kao = ko \cdot \left(\frac{Tms \cdot ma}{Tma \cdot ms}\right)^{0.5}$$

In this expression:

ko = ambient overall mass transfer coefficient (cm/hr);
Tms = shower water temperature (˚K);
ms = water viscosity at shower temperature (cp);
Tma = ambient temperature (˚K); and
ma = water viscosity at ambient temperature (cp).

The ambient overall mass transfer coefficient is given by:

$$ko = \left(\frac{1}{kw} + \frac{R \cdot Tma}{H \cdot kg}\right)^{-1}$$

where:

kw = mass transfer resistance through the water (cm/hr);
R = gas constant, 8.2 x 10^{-5} atm-m^3/mol-K;
H = Henry's law constant (atm-m^3/mol); and
kg = mass transfer resistance through the gas (cm/hr).

The following empirical relationships are used for the water and gas mass transfer coefficients.

$$kw = 20 \cdot \left(\frac{44}{MW} \right)^{0.5}$$

$$kg = 3000 \cdot \left(\frac{18}{MW} \right)^{0.5}$$

where:

MW = molecular weight of the chemical (g/mol).

The transfer factor TF_{sh2} is determined by generating an expression for C_{asav2} which depends on C_{ws}. The quantity C_{asav2} is the average concentration of chemical in the shower room air after showering and is found by solving:

$$C_{asav2} = \frac{\int_0^{T_2} C_{as}(t)\ dt}{T_2}$$

The concentration of the chemical in the shower room, $C_{as}\ (t)$, is found by solving the following differential equation:

$$Vsh \cdot \frac{d}{dt} C_{as} = - Qash \cdot C_{as}$$

This equation is similar to the previous differential equation for C_{as} except the source term, *rsh*, is now zero since the shower is off. The solution to this equation is:

$$C_{as}(t) = C_{as2z}\ e^{-ksa \cdot t}$$

The parameter *ksa* was defined previously, and the variable C_{as2z} is the concentration in the shower room when the shower is turned off and is given by:

$$C_{as2z} = \left(\frac{ksw}{ksa} \right) C_{ws} \left(1 - e^{-ksa \cdot T_1}\right)$$

The average concentration of the chemical in the air following showering is given by:

$$C_{asav2} = \frac{C_{as2z}}{ksa \cdot T_2} (1 - e^{-ksa \cdot T_2})$$

Substituting the expression for C_{as2z} into the equation yields:

$$C_{asav2} = \left(\frac{ksw}{ksa^2 \cdot T_2} \right) (1 - e^{-ksa \cdot T_1}) (1 - e^{-ksa \cdot T_2}) C_{ws}$$

The transfer factor TF_{sh2} is therefore given by:

$$TF_{sh2} = \left(\frac{ksw}{ksa^2 \cdot T_2} \right) (1 - e^{-ksa \cdot T_1}) (1 - e^{-ksa \cdot T_2})$$

2.2 Results

The results of running the model are presented in Table 2-1. The values for the volume of the shower room, *Vsh*, the rate of air flow through the shower room, *Qash*, the rate of water flow from the shower, *Qwsh*, fall time for a water droplet, *tdr*, diameter of water droplet, *d*, ambient temperature, *Tma*, shower water temperature, *Tms*, and viscosities of water at different temperatures come from Foster and Chrostowski (1987). The time spent in the shower, T_1, is also from Foster and Chrostowski (1987), while T_2 was selected to sum with T_1 to be 0.25 hr or 15 minutes, the typical time spent showering. The molecular weight, *MW*, and Henry's law constant, *H*, were taken from EPA (1996), Mackay et al. (1992a) or Mackay et al. (1992b).

3 Bath Water to Air Volatilization Model

3.1 Equations

During a bath, chemicals present in the bath water are assumed to have the opportunity to volatilize into the air within the bathroom. Initially, the concentration of a chemical in the bathroom in the air is assumed to be zero. As time goes on, the concentration increases until the bath is finished. At this time, the concentration in the air begins to decrease as air turns over in the room. Figure 3-1 depicts the evolution of the bathroom air concentrations over time. The individual taking a bath is assumed to be in the bathroom during the bath and for a period after the bath. The relevant exposure point concentration for this individual is the average concentration of the chemical in the air during this exposure event, since the individual is assumed to maintain a constant inhalation rate both during and after the bath.

To estimate this average concentration in the bathroom air, the shower water to air volatilization model of Foster and Chrostowski (1987) was modified. The model is based on a simple box model of air exchange in the bathroom with constant emission of chemicals during the bath. The average concentration in the bathroom air is the time weighted sum of the average concentration in the bathroom air during the bath and the average concentration in the bathroom air after the bath is completed.

$$C_{abav} = \frac{C_{abav1} \cdot T_1 + C_{abav2} \cdot T_2}{T_1 + T_2}$$

where:

C_{abav} = average concentration of chemical in the bathroom air (mg/m^3);
C_{abav1} = average concentration of chemical in the bathroom air during the bath (mg/m^3);
C_{abav2} = average concentration of chemical in the bathroom air after the bath (mg/m^3);
T_1 = duration of the bath (hours); and
T_2 = period individual is in bathroom after the bath (hours).

Both C_{abav1} and C_{abav2} depend upon the concentration of the chemical in the bath water and this relationship can be expressed through a transfer factor such that the following relationships and transfer factors can be defined:

$$C_{abav1} = TF_{bwa1} \cdot C_{wb}$$

$$C_{abav2} = TF_{bwa2} \cdot C_{wb}$$

$$C_{abav} = TF_{bwa} \cdot C_{wb}$$

$$TF_{bwa} = \frac{TF_{bwa1} \cdot T_1 + TF_{bwa2} \cdot T_2}{T_1 + T_2}$$

where:

C_{wb} = concentration of chemical in the bath water (mg/L);
TF_{bwa1} = transfer factor describing relationship between concentration in bathroom air and bath water during the bath (L/m^3);
TF_{bwa2} = transfer factor describing relationship between concentration in bathroom air and bath water after the bath (L/m^3); and
TF_{bwa} = overall transfer factor describing overall relationship between concentration in bathroom air and bath water (L/m^3).

The variable TF_{bwa1} is determined by first developing an expression for C_{abav1}. This variable is determined by solving the following equation:

$$C_{abav1} = \frac{\int_0^{T_1} C_{ab}(t)\ dt}{T_1}$$

where:

$C_{ab}(t)$ = concentration of chemical in the bathroom air over time (mg/m^3).

The variable $C_{ab}(t)$ can be determined by solving the following differential equation:

$$Vbrm \cdot \frac{d}{dt} C_{ab}(t) = rbrm - Qabrm \cdot C_{ab}(t)$$

where:

Vbrm	=	volume of bathroom (m^3);
rbrm	=	rate of chemical emission into bathroom (mg/hr); and
Qabrm	=	rate of air flow out of bathroom (m^3/hr).

This equation states that the rate at which the mass of chemical in the bathroom changes depends on the difference in the rate at which the chemical is volatilized from the bath water minus the rate at which the chemical leaves the bathroom as air circulates through the bathroom.

The rate at which the chemical is introduced into the bathroom from bath water is the mass of the chemical in the bath water times the fraction volatilized during the bath. Thus, the rate at which the chemical is introduced into the bathroom is given by:

$$rbrm = \frac{fv \cdot Vbw \cdot C_{wb}}{T_1}$$

where:

fv	=	fraction of chemical volatilized; and
Vbw	=	volume of bath water (m^3).

The differential equation describing the change in C_{as} over time becomes:

$$\frac{d}{dt} C_{ab} = \frac{fv \cdot Vbw}{T_1 \cdot Vbrm} \cdot C_{wb} - \frac{Qabrm}{Vbrm} \cdot C_{ab}$$

By defining two rate constants, *kbw* and *kba*, this equation can be restated:

$$\frac{d}{dt} C_{ab} = kbw \cdot C_{wb} - kba \cdot C_{ab}$$

The constants *kbw* and *kba* are defined as:

$$kbw = \frac{fv \cdot Vbw}{T_1 \cdot Vbrm}$$

where:

$$kba = \frac{Qabrm}{Vbrm}$$

kbw = first order rate constant describing release of chemical from bath water to air (L/m^3-hr); and

kba = first order rate constant describing turnover of air in bathroom (1/hr).

The solution to this differential equation is:

$$C_{ab}(t) = \left(\frac{kbw}{kba}\right) C_{wb} \left(1 - e^{-kba \cdot t}\right)$$

The average concentration in the bathroom during the bath is given by:

$$C_{abav1} = \left(\frac{kbw}{kba}\right) C_{wb} \left[1 - \left(\frac{1}{kba \cdot T_1}\right) \left(1 - e^{-kba \cdot T_1}\right)\right]$$

The transfer factor TF_{bwa1} is given by:

$$TF_{bwa1} = \left(\frac{kbw}{kba}\right) \left[1 - \left(\frac{1}{kba \cdot T_1}\right) \left(1 - e^{-kba \cdot T_1}\right)\right]$$

In order to solve this equation, the parameter *fv* must be determined. This parameter is estimated by determining the change in concentration of the chemical in the bath water which depends on the rate of volatilization. The rate at which the chemical is emitted from the bath water is given by:

$$Vbw \cdot \frac{d}{dt} Cwb = - kao \cdot Abw \cdot CF1 \cdot C_{wb}$$

where:

kao = overall mass transfer coefficient (cm/hr);
Abw = area of the bath water (m^2); and
CF1 = conversion factor (10^{-2} m/cm).

The solution to this differential equation is:

$$C_{wb}(t) = C_{wbz}\, e^{-\left(\frac{kao \cdot Abw \cdot CF1}{Vbw}\right)}$$

where:

Cwbz = initial concentration in bath water (mg/L).

The fraction volatilized at time T_1 is:

$$fv = 1 - \left(\frac{C_{wb}(t)}{C_{wbz}}\right)$$

or

$$fv = 1\, e^{-\left(\frac{kao \cdot Abw \cdot CF1}{Vbw}\right) T_1}$$

The overall mass transfer coefficient, *kao*, is based on an ambient overall mass transfer coefficient that is adjusted for the higher bath water temperature.

$$kao = ko \cdot \left(\frac{Tmb \cdot ma}{Tma \cdot mb}\right)^{0.5}$$

In this expression:

ko = ambient overall mass transfer coefficient (cm/hr);
Tmb = bath water temperature (˚K);
ms = water viscosity at bath water temperature (cp);

Tma = ambient temperature (˚K); and
ma = water viscosity at ambient temperature (cp).

The ambient overall mass transfer coefficient is given by:

$$ko = \left(\frac{1}{kw} + \frac{R \cdot Tma}{H \cdot kg} \right)^{-1}$$

where:

kw = mass transfer resistance through the water (cm/hr);
R = gas constant, 8.2 x 10^{-5} atm-m^3/mol-K;
H = Henry's law constant (atm-m^3/mol); and
kg = mass transfer resistance through the gas (cm/hr).

The following empirical relationships are used for the water and gas mass transfer coefficients.

$$kw = 20 \cdot \left(\frac{44}{MW} \right)^{0.5}$$

$$kg = 3000 \cdot \left(\frac{18}{MW} \right)^{0.5}$$

where:

MW = molecular weight of the chemical (g/mol).

The transfer factor TF_{bwa2} is determined by generating an expression for C_{abav2} which depends on C_{wb}. The quantity C_{abav2} is the average concentration of chemical in the bathroom air after the bath and is found by solving:

$$C_{abav2} = \frac{\int_0^{T_2} C_{ab}(t)\ dt}{T_2}$$

The concentration of the chemical in the bathroom, $C_{ab}(t)$, is found by solving the following differential equation:

$$Vbrm \cdot \frac{d}{dt} C_{ab} = - Qabrm \cdot C_{ab}$$

This equation is similar to the previous differential equation for C_{ab} except the source term, *rbrm*, is now zero since the bath water has been drained. The solution to this equation is:

$$C_{ab}(t) = C_{ab2z} \, e^{- kba \cdot t}$$

The parameter *kba* was defined previously, and the variable C_{ab2z} is the concentration in the bathroom when the bath water drains and is given by:

$$C_{ab2z} = \left(\frac{kbw}{kba} \right) C_{wb} \left(1 - e^{- kba \cdot T_1} \right)$$

The average concentration of the chemical in the air following the bath is given by:

$$C_{abav2} = \frac{C_{ab2z}}{kba \cdot T_2} \left(1 - e^{- kba \cdot T_2} \right)$$

Substituting the expression for C_{as2z} into the equation yields:

$$C_{abav2} = \left(\frac{kbw}{kba^2 \cdot T_2} \right) (1 - e^{- kba \cdot T_1}) (1 - e^{- kba \cdot T_2}) \, C_{wb}$$

The transfer factor TF_{bwa2} is given by:

$$TF_{bwa2} = \left(\frac{kbw}{kba^2 \cdot T_2} \right) (1 - e^{- kba \cdot T_1}) (1 - e^{- kba \cdot T_2})$$

3.2 Results

The results of running the model are presented in Table 3-1. The values for the volume of the bathroom, *Vbrm*, and rate of air flow through the bathroom, *Qabrm*,

were taken from Foster and Chrotowski (1987). The area of bath water, *Abw*, and the depth of the bath water, *dbw*, were estimated from a typical bath (approximately 4 feet by 2 feet for the area and 8 inches for the depth). The quantity ***Abw · dbw*** gives *Vbw*. The ambient temperature of water, *Tma*, and the viscosity at this temperature is taken from Foster and Chrostowski (1987). The temperature of the bath water was estimated while the viscosity of water at this temperature was estimated from Linsley and Franzini (1979). The time spent in a bath, T_1, was estimated to be 0.25 hr or 15 minutes, while T_2 was selected to sum with T_1 to be 0.33 hr or 20 minutes, the typical time in the bathroom during and just after a bath. The molecular weight, *MW* and Henry's law constant, *H*, were taken from EPA (1996), Mackay et al. (1992a) or Mackay et al. (1992b).

4 Surface Water to Air Volatilization Model

4.1 Equations

Ambient concentrations of chemicals in air resulting from volatilization from surface water may be estimated as follows:

$$C_{oa} = TF_{swoa} \cdot C_{sw}$$

where:

C_{oa} = concentration of chemical in outdoor air (mg/m^3);
TF_{swoa} = transfer factor from surface water to outdoor air (L/m^3); and
C_{sw} = concentration of chemical in surface water (mg/L).

The transfer factor, TF_{swoa}, describes the relationship between the concentration in outdoor air and the concentration in surface water and is given by the following expression:

$$TF_{swoa} = DF_{swoa} \cdot FF_{swoa} \cdot CF1$$

where:

DF_{swoa} = dispersion factor [(m^2-s)/(m^3)];
FF_{swoa} = flux factor (m/s); and
$CF1$ = conversion factor (1000 L/m^3).

The dispersion factor, DF_{swoa}, translates a flux of a chemical from surface water to an air concentration. The flux factor, FF_{swoa}, is given by the following expression:

$$FF_{swoa} = K_{ol} \cdot CF2$$

where:

K_{ol} = overall mass-transfer coefficient (m/day); and
$CF2$ = conversion factor (day/86,400 sec).

The overall mass-transfer coefficient is dependent on the physical and chemical properties of the compound as well as environmental conditions (Achman et al., 1993). The reciprocal of K_{ol} is the total resistance to transfer expressed on a water and vapor phase basis and is given by the following expression (Achman et al., 1993])

$$\frac{1}{K_{ol}} = \frac{1}{k_w} + \frac{RT}{Hk_a}$$

where:

k_w = water phase mass-transfer coefficient (m/day);
k_a = vapor phase mass-transfer coefficient (m/day);
R = universal gas constant (atm-m^3/mol-K);
T = absolute temperature (K); and
H = Henry's law constant (atm-m^3/mol).

The water phase mass-transfer coefficient for a particular chemical, k_w, can be related to the water phase mass-transfer coefficient for carbon dioxide (CO_2) through an empirical relationship involving a dimensionless number known as the Schmidt number (Sc) (Achman et al., 1993):

$$k_w = k_{w(CO_2)} \cdot \left(\frac{Sc}{Sc_{(CO_2)}} \right)^{nw} \cdot CF3$$

where:

$k_{w(CO_2)}$ = water phase mass-transfer coefficient for CO_2 (cm/hr);
Sc = Schmidt number for the chemical;
$Sc_{(CO_2)}$ = Schmidt number for CO_2;
nw = an empirical coefficient; and
CF3 = conversion factor (0.24 m/day per cm/hr).

An expression for $k_{w(CO_2)}$ that is dependent on windspeed (Achman et al., 1993) is:

$$k_{w_{(CO_2)}} = 0.17 \cdot u_{10} \qquad \text{for } u_{10} < 3.6 \text{ m/s}$$

$$k_{w_{(CO_2)}} = 2.85 \cdot u_{10} - 9.65 \qquad \text{for } 3.6 < u_{10} < 13 \text{ m/s}$$

$$k_{w_{(CO_2)}} = 5.9 \cdot u_{10} - 49.3 \qquad \text{for } u_{10} > 13 \text{ m/s}$$

where u_{10} is the wind speed at a reference height of 10m (in units of m/s) and $k_{w(CO_2)}$ has units of cm/hr. The Schmidt number of a chemical is given by the following expression (Achman et al., 1993):

$$Sc = \frac{\nu_w}{D_w}$$

where:

ν_w = kinematic viscosity of water (cm^2/s); and
D_w = diffusivity of a chemical through water (cm^2/s).

Achman et al. (1993) give a Schmidt number for carbon dioxide through water of 600 and indicated that *nw* is equal to -2/3 for u_{10} less than 3.6 m/s or -1/2 for u_{10} greater than 3.6 m/s.

The vapor phase mass-transfer coefficient, k_a, can be related to the vapor phase mass-transfer coefficient for water vapor, $k_{a(H_2O)}$, through an empirical equation involving diffusivities in air (Achman et al.):

$$k_a = k_{a(H_2O)} \cdot \left(\frac{D_a}{D_{a(H_2O)}} \right)^{na} \cdot CF4$$

where:

$k_{a(H_2O)}$ = vapor phase mass-transfer coefficient for water vapor (cm/sec);
D_a = diffusivity of the chemical in air (cm^2/sec);
$D_{a(H_2O)}$ = diffusivity of water vapor in air (cm^2/sec);
na = an empirical coefficient; and
CF4 = conversion factor (864 m/day per cm/sec).

The vapor phase mass-transfer coefficient for water vapor, $k_{a(H_2O)}$, is given by the following empirical equation (Achman et al., 1993):

$$k_{a(H_2O)} = 0.2\ u_{10} + 0.3$$

where $k_{a(H2O)}$ has units of cm/sec. Achman et al. (1993) estimate *na* to be 0.61.

For this analysis, the dispersion factor, DF_{swoa}, was determined from Q/C data in the EPA Soil Screening Guidance (EPA, 1996). The parameter Q/C is the inverse of the concentration in the center of a square surface source. Values of Q/C are given for Chicago for six areas:

Area (acre)	Q/C (g/m^2-s) per (kg/m^3)
0.5	97.78
1	85.81
2	76.08
5	65.75
10	59.16
30	50.60

The values of Q/C were translated into values of DF_{swoa} through the following equation:

$$DF_{swoa} = \frac{10^3 g/kg}{(Q/C)}$$

This expression gives DF_{swoa} in the correct units of m^2-s/m^3. The resulting values for DF_{swoa} as a function of area were then fit to the following equation through regression analysis:

$$DF_{swoa} = C_1 \cdot A^{C_2}$$

where:

A = area of surface source (acres).

The regression analysis yielded the following values for C_1 and C_2:

$$C_1 = 11.62$$
$$C_2 = 0.1604$$

Figure 4-1 presents the values of DF_{swoa} as a function of area and the fitted line through the data.

4.2 Results

The model requires a number of system parameters and chemical properties entered as inputs. The system parameters include the temperature (T), the source area (A), the wind speed (u_{10}), the kinematic viscosity of water (ν_w), and the vapor phase diffusivity of water ($D_{a(H_2O)}$). The temperature T was estimated to be about 288°K (15°C or 59°F) and the vapor phase diffusivity of water $D_{a(H_2O)}$ was estimated to be 0.24 cm^2/s (Weast et al., 1984). The average wind speed for Green Bay was used in this analysis (GRI, 1987). The areas A for the different areas of interest (AOI) were estimated as indicated in Attachment 1. These areas were used in the regression equation to calculate a value of DF_{swoa} for each AOI. The values for the chemical properties water phase diffusivity, D_w, vapor phase diffusivity, D_a, and Henry's Law constant, H, were taken from EPA (1996), Mackay et al. (1992a) or Mackay et al. (199b). The results for the Little Lake Butte des Morts, Appleton to Little Rapids, Little Rapids to DePere and DePere to Green Bay reaches are provided in Tables 4-1 through 4-4, respectively. Table 4-5 presents results for Green Bay.

5 Sediment to Pore Water Partitioning Model

5.1 Equations

The concentration of chemicals in sediment pore water can be estimated from the following equation:

$$C_{pw} = TF_{sdpw} \cdot C_{sed}$$

where:

C_{pw} = concentration of chemical in sediment pore water (mg/L);
TF_{sdpw} = transfer factor from sediment to sediment pore water ((mg/L)/(mg/Kg)); and
C_{sed} = concentration of chemical in sediment (mg/kg).

The transfer factor, TF_{sdpw}, is the inverse of the sediment to pore water partitioning coefficient, Kp:

$$TF_{sdpw} = \frac{1}{K_p}$$

The sediment to pore water partitioning coefficient depends on the type of chemical. For organic chemicals, the partitioning coefficient is given by:

$$Kp = foc \cdot Koc$$

where:

Kp = sediment to pore water partitioning coefficient ((mg/kg)/(mg/L));
foc = fraction organic carbon in sediment (kg-oc/kg-sed); and
Koc = organic carbon to water partitioning coefficient.

For inorganic chemicals, a partitioning coefficient that is dependent on pH is given in EPA (1996).

5.2 Results

The sediment to pore water partitioning coefficients for chemicals of potential concern are provided for the Appleton to Little Rapids, Little Rapids to DePere and DePere to Green Bay reaches in Tables 5-1 through 5-4, respectively. Table 5-5 provides sediment to pore water partitioning coefficients for Green Bay. For each location, the fraction of organic carbon, *foc*, was taken as the arithmetic average of the fraction organic carbon in all sediment samples. The organic carbon to water partitioning coefficients for organic chemicals were obtained from EPA (1996), Mackay et al. (1992a) or Mackay et al. (1992b). These values are provided in the column labeled *Koc* with a *Koc* Type of 1. The sediment to water partitioning coefficient for inorganic chemicals is provided in the column labeled *Koc* with *Koc* Type equal to 3. These values were obtained from EPA (1996) for a pH of 6.8.

6 References

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Appendix B4

Exposure Point Concentrations, Unit Cancer Risks, Unit Hazard Indices, Cancer Risks, and Hazard Indices for Different Receptors

This appendix provides exposure point concentrations for each reach of the Lower Fox River and Green Bay, and unit cancer risks, unit hazard indices, cancer risks and hazard indices for the following receptors:

- recreational anglers;
- high intake fish consumers;
- hunters;
- drinking water users;
- local residents;
- recreational water users; and
- marine construction workers.

Exposure Point Concentrations for Reaches:

- Little Lake Butte des Morts Reach - Upperbound
- Little Lake Butte des Morts Reach - Average
- Appleton to Little Rapids Reach - Upperbound
- Appleton to Little Rapids Reach - Average
- Little Rapids to DePere Reach - Upperbound
- Little Rapids to DePere Reach - Average
- DePere to Green Bay Reach - Upperbound
- DePere to Green Bay Reach - Average
- Green Bay - Upperbound
- Green Bay - Average

Receptor: Recreational Angler

Exposure Scenario: RME Assumptions (with Upperbound Concentrations)

Areas Evaluated: Little Lake Butte des Morts Reach
Appleton to Little Rapids Reach
Little Rapids to DePere Reach
DePere to Green Bay Reach
Green Bay

Exposure Pathways: Ingestion of Fish
Incidental Ingestion of Surface Water
Dermal Contact with Surface Water
Inhalation of Volatiles in Outdoor Air

Receptor: Recreational Angler

Exposure Scenario: RME Assumptions (with Average Concentrations)

Areas Evaluated: Little Lake Butte des Morts Reach
Appleton to Little Rapids Reach
Little Rapids to DePere Reach
DePere to Green Bay Reach
Green Bay

Exposure Pathways: Ingestion of Fish
Incidental Ingestion of Surface Water
Dermal Contact with Surface Water
Inhalation of Volatiles in Outdoor Air

Receptor: Recreational Angler

Exposure Scenario: CTE Assumptions (with Average Concentrations)

Areas Evaluated: Little Lake Butte des Morts Reach
Appleton to Little Rapids Reach
Little Rapids to DePere Reach
DePere to Green Bay Reach
Green Bay

Exposure Pathways: Ingestion of Fish
Incidental Ingestion of Surface Water
Dermal Contact with Surface Water
Inhalation of Volatiles in Outdoor Air

Receptor: High Intake Fish Consumer

Exposure Scenario: RME Assumptions (with Upperbound Concentrations)

Areas Evaluated: Little Lake Butte des Morts Reach
Appleton to Little Rapids Reach
Little Rapids to DePere Reach
DePere to Green Bay Reach
Green Bay

Exposure Pathways: Ingestion of Fish
Incidental Ingestion of Surface Water
Dermal Contact with Surface Water
Inhalation of Volatiles in Outdoor Air

Receptor: High Intake Fish Consumer

Exposure Scenario: RME Assumptions (with Average Concentrations)

Areas Evaluated: Little Lake Butte des Morts Reach
Appleton to Little Rapids Reach
Little Rapids to DePere Reach
DePere to Green Bay Reach
Green Bay

Exposure Pathways: Ingestion of Fish
Incidental Ingestion of Surface Water
Dermal Contact with Surface Water
Inhalation of Volatiles in Outdoor Air

Receptor: High Intake Fish Consumer

Exposure Scenario: CTE Assumptions (with Average Concentrations)

Areas Evaluated: Little Lake Butte des Morts Reach
Appleton to Little Rapids Reach
Little Rapids to DePere Reach
DePere to Green Bay Reach
Green Bay

Exposure Pathways: Ingestion of Fish
Incidental Ingestion of Surface Water
Dermal Contact with Surface Water
Inhalation of Volatiles in Outdoor Air

Receptor: Hunter

Exposure Scenario: RME Assumptions (with Upperbound Concentrations)

Areas Evaluated: Little Lake Butte des Morts Reach
Appleton to Little Rapids Reach
Little Rapids to DePere Reach
DePere to Green Bay Reach
Green Bay

Exposure Pathways: Ingestion of Waterfowl
Incidental Ingestion of Surface Water
Dermal Contact with Surface Water
Inhalation of Volatiles in Outdoor Air

Receptor: Hunter

Exposure Scenario: RME Assumptions (with Average Concentrations)

Areas Evaluated: Little Lake Butte des Morts Reach
Appleton to Little Rapids Reach
Little Rapids to DePere Reach
DePere to Green Bay Reach
Green Bay

Exposure Pathways: Ingestion of Waterfowl
Incidental Ingestion of Surface Water
Dermal Contact with Surface Water
Inhalation of Volatiles in Outdoor Air

Receptor: Hunter

Exposure Scenario: CTE Assumptions (with Average Concentrations)

Areas Evaluated: Little Lake Butte des Morts Reach
Appleton to Little Rapids Reach
Little Rapids to DePere Reach
DePere to Green Bay Reach
Green Bay

Exposure Pathways: Ingestion of Waterfowl
Incidental Ingestion of Surface Water
Dermal Contact with Surface Water
Inhalation of Volatiles in Outdoor Air

Receptor: Drinking Water User

Exposure Scenario: RME Assumptions (with Upperbound Concentrations)

Areas Evaluated: Little Lake Butte des Morts Reach
Appleton to Little Rapids Reach
Little Rapids to DePere Reach
DePere to Green Bay Reach
Green Bay

Exposure Pathways: Ingestion of Surface Water
Dermal Contact with Surface Water
Inhalation of Volatiles in Indoor Air

Receptor: Drinking Water User

Exposure Scenario: RME Assumptions (with Upperbound Concentrations and Recent Mercury Data)

Areas Evaluated: Little Lake Butte des Morts Reach
Appleton to Little Rapids Reach
Little Rapids to DePere Reach
DePere to Green Bay Reach
Green Bay

Exposure Pathways: Ingestion of Surface Water
Dermal Contact with Surface Water
Inhalation of Volatiles in Indoor Air

Receptor: Local Resident

Exposure Scenario: RME Assumptions (with Upperbound Concentrations)

Areas Evaluated: Little Lake Butte des Morts Reach
Appleton to Little Rapids Reach
Little Rapids to DePere Reach
DePere to Green Bay Reach
Green Bay

Exposure Pathways: Inhalation of Volatiles in Outdoor Air

Receptor: Local Resident

Exposure Scenario: RME Assumptions (with Upperbound Concentrations and Recent Mercury Data)

Areas Evaluated: Little Lake Butte des Morts Reach
Appleton to Little Rapids Reach
Little Rapids to DePere Reach
DePere to Green Bay Reach
Green Bay

Exposure Pathways: Inhalation of Volatiles in Outdoor Air

Receptor: Recreational Water User: Swimmer

Exposure Scenario: RME Assumptions (with Upperbound Concentrations)

Areas Evaluated: Little Lake Butte des Morts Reach
Appleton to Little Rapids Reach
Little Rapids to DePere Reach
DePere to Green Bay Reach
Green Bay

Exposure Pathways: Incidental Ingestion of Surface Water
Dermal Contact with Surface Water
Inhalation of Volatiles in Outdoor Air
Incidental Ingestion of Sediments
Dermal Contact with Sediment Pore Water

Receptor: Recreational Water User: Wader

Exposure Scenario: RME Assumptions (with Upperbound Concentrations)

Areas Evaluated: Little Lake Butte des Morts Reach
Appleton to Little Rapids Reach
Little Rapids to DePere Reach
DePere to Green Bay Reach
Green Bay

Exposure Pathways: Incidental Ingestion of Surface Water
Dermal Contact with Surface Water
Inhalation of Volatiles in Outdoor Air
Incidental Ingestion of Sediments
Dermal Contact with Sediment Pore Water

Receptor: Marine Construction Worker

Exposure Scenario: RME Assumptions (with Upperbound Concentrations)

Areas Evaluated: Little Lake Butte des Morts Reach
Appleton to Little Rapids Reach
Little Rapids to DePere Reach
DePere to Green Bay Reach
Green Bay

Exposure Pathways: Incidental Ingestion of Surface Water
Dermal Contact with Surface Water
Inhalation of Volatiles in Outdoor Air
Incidental Ingestion of Sediments
Dermal Contact with Sediments

Appendix B5

Concentrations of Lead in Surface Sediment, Surface Water, Fish Tissue, and Waterfowl Tissue Samples

Table 1 Lead Concentrations in Surface Sediment Samples

LOCATION	SAMPLE NUMBER	SAMPLE TYPE	SAMPLE DATE	RESULT (mg/kg)
Little Lake Butte des Morts	D-RI-Comp1(0-2)	Surface Sediment		3.99 J
Little Lake Butte des Morts	D-RI-Comp2(0-2)	Surface Sediment		160 J
Little Lake Butte des Morts	E-RI-Comp1(0-2)	Surface Sediment		7.10
Little Lake Butte des Morts	E-RI-Comp2(0-2)	Surface Sediment		7.79
Little Lake Butte des Morts	P-RI-Comp1(0-2)	Surface Sediment		6.08
Little Lake Butte des Morts	2C2 (Tr)	Surface Sediment	1993	300
Little Lake Butte des Morts	POG (Tr)	Surface Sediment	1992	110
Little Lake Butte des Morts	2E8 (Tr)	Surface Sediment	1993	99
Little Lake Butte des Morts	SDC-C-1-P-S	Surface Sediment	06/05/1998	262
Little Lake Butte des Morts	SDC-C-3-P-S	Surface Sediment	06/05/1998	162
Little Lake Butte des Morts	SDC-E-1-P-S	Surface Sediment	06/05/1998	289
Little Lake Butte des Morts	SDC-E-3-P-S	Surface Sediment	06/05/1998	39
Appleton to Little Rapids	N-RI-Comp1(0-2)	Surface Sediment		5.43
Appleton to Little Rapids	N-RI-Comp2(0-2)	Surface Sediment		5.17
Appleton to Little Rapids	N-RI-Comp3(0-2)	Surface Sediment		7.25
Appleton to Little Rapids	N (Tr)	Surface Sediment	1992	280
Little Rapids to Depere	EGH-RI-Comp1(0-2)	Surface Sediment		6.15
Little Rapids to Depere	X (Tr)	Surface Sediment	1992	130
Little Rapids to Depere	HH (Tr)	Surface Sediment	1992	1400
Little Rapids to Depere	SDC-EE26-5-P-S	Surface Sediment	06/01/1998	297
Little Rapids to Depere	SDC-EE26-1-P-S	Surface Sediment	06/01/1998	123
Little Rapids to Depere	SDC-EE25-1-P-S	Surface Sediment	06/02/1998	148
Little Rapids to Depere	SDC-EE25-3-P-S	Surface Sediment	06/02/1998	72
Little Rapids to Depere	SDC-EE24-1-P-S	Surface Sediment	06/02/1998	62
Little Rapids to Depere	SDC-EE24-3-P-S	Surface Sediment	06/02/1998	70
Little Rapids to Depere	SDC-EE22-3-P-S	Surface Sediment	06/03/1998	126
Little Rapids to Depere	SDC-EE22-2-P-S	Surface Sediment	06/03/1998	68
Little Rapids to Depere	SDC-EE23-2-P-S	Surface Sediment	06/03/1998	74
Little Rapids to Depere	SDC-EE23-3-P-S	Surface Sediment	06/03/1998	68
Little Rapids to Depere	SDC-W-2-P-S	Surface Sediment	06/04/1998	60
Little Rapids to Depere	SDC-W-3-P-S	Surface Sediment	06/04/1998	57
Little Rapids to Depere	SDC-X-1-P-S	Surface Sediment	06/04/1998	84
Little Rapids to Depere	SDC-X-3-P-S	Surface Sediment	06/04/1998	71
DePere to Green Bay	95002-01	Surface Sediment		104.432
DePere to Green Bay	95004-01	Surface Sediment		90.64
DePere to Green Bay	95006-01	Surface Sediment		39.64
DePere to Green Bay	95007-01	Surface Sediment		75.44
DePere to Green Bay	95008-01	Surface Sediment		96.24
DePere to Green Bay	95010-01	Surface Sediment		104.406
DePere to Green Bay	95011-01	Surface Sediment		84.24
DePere to Green Bay	95013-01	Surface Sediment		76.84
DePere to Green Bay	95016-01	Surface Sediment		38.24
DePere to Green Bay	95018-01	Surface Sediment		85.04
DePere to Green Bay	95020-01	Surface Sediment		140.425
DePere to Green Bay	95022-01	Surface Sediment		4.44
DePere to Green Bay	95025-01	Surface Sediment		80.64
DePere to Green Bay	95028-01	Surface Sediment		80.54
DePere to Green Bay	95030-01	Surface Sediment		77.94
DePere to Green Bay	95035-01	Surface Sediment		166.429
DePere to Green Bay	95038-01	Surface Sediment		110.431
DePere to Green Bay	95041-01	Surface Sediment		73.8
DePere to Green Bay	95044-01	Surface Sediment		69.74
DePere to Green Bay	95047-01	Surface Sediment		85.64
DePere to Green Bay	95049-01	Surface Sediment		77.9
DePere to Green Bay	95051-01	Surface Sediment		84.1
DePere to Green Bay	95052-01	Surface Sediment		65.4

Table 1 Lead Concentrations in Surface Sediment Samples

LOCATION	SAMPLE NUMBER	SAMPLE TYPE	SAMPLE DATE	RESULT (mg/kg)
DePere to Green Bay	95054-01	Surface Sediment		76.74
DePere to Green Bay	95056-01	Surface Sediment		88.4
DePere to Green Bay	95058-01	Surface Sediment		73.3
DePere to Green Bay	95060-01	Surface Sediment		29.6
DePere to Green Bay	95061-01	Surface Sediment		83.2
DePere to Green Bay	95062-01	Surface Sediment		47.8
DePere to Green Bay	95064-01	Surface Sediment		9.3
DePere to Green Bay	95066-01	Surface Sediment		108
DePere to Green Bay	95068-01	Surface Sediment		76.2
DePere to Green Bay	95070-01	Surface Sediment		77.2
DePere to Green Bay	95071-01	Surface Sediment		80.8
DePere to Green Bay	95072-01	Surface Sediment		78.2
DePere to Green Bay	95074-01	Surface Sediment		88.5
DePere to Green Bay	95076-01	Surface Sediment		91.1
DePere to Green Bay	95077-01	Surface Sediment		85.4
DePere to Green Bay	95078-01	Surface Sediment		93.8
DePere to Green Bay	95079-01	Surface Sediment		74.9
DePere to Green Bay	95080-01	Surface Sediment		84.7
DePere to Green Bay	95081-01	Surface Sediment		98.5
DePere to Green Bay	95082-01	Surface Sediment		71.4
DePere to Green Bay	95084-01	Surface Sediment		83.8
DePere to Green Bay	95085-01	Surface Sediment		121
DePere to Green Bay	95086-01	Surface Sediment		85.6
DePere to Green Bay	95087-01	Surface Sediment		80.4
DePere to Green Bay	95088-01	Surface Sediment		89.8
DePere to Green Bay	95089-01	Surface Sediment		73.1
DePere to Green Bay	95090-01	Surface Sediment		128
DePere to Green Bay	95091-01	Surface Sediment		218
DePere to Green Bay	95092-01	Surface Sediment		96.5
DePere to Green Bay	95093-01	Surface Sediment		71.9
DePere to Green Bay	95094-01	Surface Sediment		52.1
DePere to Green Bay	95095-01	Surface Sediment		41.6
DePere to Green Bay	95096-01	Surface Sediment		17.2
DePere to Green Bay	95097-01	Surface Sediment		59.6
DePere to Green Bay	95098-01	Surface Sediment		41.9
DePere to Green Bay	95099-01	Surface Sediment		5.3
DePere to Green Bay	95100-01	Surface Sediment		40
DePere to Green Bay	95101-01	Surface Sediment		20.2
DePere to Green Bay	95102-01	Surface Sediment		79.6
DePere to Green Bay	95103-01	Surface Sediment		49
DePere to Green Bay	95104-01	Surface Sediment		19.1
DePere to Green Bay	95105-01	Surface Sediment		62.1
DePere to Green Bay	95106-01	Surface Sediment		62.1
DePere to Green Bay	95109-01	Surface Sediment		83.5
DePere to Green Bay	2FRB1 (Tr)	Surface Sediment	1993	99
DePere to Green Bay	2FRB22 (Tr)	Surface Sediment	1993	180
DePere to Green Bay	2FRB17 (Tr)	Surface Sediment	1993	27
DePere to Green Bay	FRB (Tr)	Surface Sediment	1992	350
DePere to Green Bay	SDC-DPD-1-P-S	Surface Sediment	06/03/1998	113
DePere to Green Bay	SDC-DPD-2-P-S	Surface Sediment	06/03/1998	89
DePere to Green Bay	SDC-DPD-3-P-S	Surface Sediment	06/03/1998	72
DePere to Green Bay	SDC-DPD-4-P-S	Surface Sediment	06/03/1998	20
DePere to Green Bay	SDC-DPD-5-P-S	Surface Sediment	06/03/1998	58
Reference	REF (Tr)	Surface Sediment	1993	20
Lake Winnebago	SDC-LW-1-P-S	Surface Sediment	06/08/1998	30

Table 1 Lead Concentrations in Surface Sediment Samples

LOCATION	SAMPLE NUMBER	SAMPLE TYPE	SAMPLE DATE	RESULT (mg/kg)
Lake Winnebago	SDC-LW-2-P-S	Surface Sediment	06/08/1998	36
Lake Winnebago	SDC-LW-3-P-S	Surface Sediment	06/08/1998	39

Table 2 Lead Concentrations in Surface Water Samples

LOCATION	SAMPLE NUMBER	SAMPLE TYPE	SAMPLE DATE	RESULT (ug/L)
Fox River at Princeton	Princeton	filtered water	Fall 91	0.066
Fox River at N. Lttl.Lk. Butte	NLLBDM	filtered water	Fall 91	0.117
Fox River at Wrightstown	Wrightstown	filtered water	Fall 92	0.118
Duck Creek at Oneida	Oneida	filtered water	Fall 92	0.0442
Fox River at Wrightstown	Wrightstown	filtered water	Spr. 93	0.124
Duck Creek at Oneida	Oneida	filtered water	Spr. 93	0.044
Fox River at Princeton	Princeton	unfiltered water	Fall 91	0.949
Fox River at N. Lttl.Lk. Butte	NLLBDM	unfiltered water	Fall 91	1.45
Fox River at Wrightstown	Wrightstown	unfiltered water	Fall 92	0.707
Duck Creek at Oneida	Oneida	unfiltered water	Fall 92	0.0733
Fox River at Wrightstown	Wrightstown	unfiltered water	Spr. 93	0.526
Duck Creek at Oneida	Oneida	unfiltered water	Spr. 93	0.264
Appleton Papers Intake	API_Intake	unfiltered water	3/1997	0.9
Green Bay Packaging Intake	GBPI_Intake	unfiltered water	8/1997	2.4
Green Bay Packaging Intake	GBPI_Intake	unfiltered water	8/1997	5.3
Nicolet Paper Intake	NP_Intake	unfiltered water	8/1997	1.1
Nicolet Paper Intake	NP_Intake	unfiltered water	8/1997	1.9
Thilmany Intake	T_Intake	unfiltered water	4/1997	1.49
Kerwin Paper Intake	KP_Intake	unfiltered water	3/1997	1.8
GBMSD River & Bay	GBMSD_Intake	unfiltered water	1993	1.45

Table 3 Lead Concentrations in Game Fish Tissue Samples

LOCATION	SAMPLE NUMBER	SPECIES	SAMPLE TYPE	SAMPDATE	RESULT (mg/kg)
Little Lake Butte des Morts	8602(d)	Carp	fillet and skin	09/04/1986	5 U
Little Lake Butte des Morts	8604(d)	Walleye	fillet and skin	09/04/1986	5 U
Little Lake Butte des Morts	8301(g)	Carp	whole fish	09/06/1983	5 U
Little Lake Butte des Morts	8601(e)	Walleye	whole fish	09/04/1986	5 U
Little Lake Butte des Morts	8603(c)	Carp	whole fish	09/04/1986	5 U
Little Lake Butte des Morts	7701(f)	Carp	whole fish	05/20/1977	5 U
Little Lake Butte des Morts	7702(f)	Carp	whole fish	05/20/1977	5 U
Little Lake Butte des Morts	7703(f)	Walleye	whole fish	05/20/1977	5 U
Little Lake Butte des Morts	7901(g)	Northern Pike	whole fish	08/20/1979	5 U
Little Lake Butte des Morts	7902(l)	White Sucker	whole fish	08/20/1979	5 U
Little Lake Butte des Morts	7903(k)	Carp	whole fish	08/20/1979	5 U
Little Lake Butte des Morts	8001(f)	Northern Pike	whole fish	09/02/1980	5 U
Little Lake Butte des Morts	8002(e)	Carp	whole fish	09/02/1980	5 U
Little Lake Butte des Morts	8003(e)	Walleye	whole fish	09/02/1980	5 U
Little Lake Butte des Morts	8004(d)	White Sucker	whole fish	09/02/1980	5 U
Little Lake Butte des Morts	8101(l)	Walleye	whole fish	08/17/1981	5 U
Little Lake Butte des Morts	8102(j)	White Sucker	whole fish	08/17/1981	5 U
Little Lake Butte des Morts	8103(h)	Carp	whole fish	08/17/1981	5 U
Little Lake Butte des Morts	8201(h)	Walleye	whole fish	09/10/1982	5 U
Little Lake Butte des Morts	8202(h)	White Sucker	whole fish	09/10/1982	5 U
Little Lake Butte des Morts	8203(g)	Carp	whole fish	09/10/1982	5 U
Little Lake Butte des Morts	781A	White Sucker	whole fish	09/06/1978	5 U
Little Lake Butte des Morts	781B	Walleye	whole fish	09/06/1978	5 U
Little Lake Butte des Morts	781C	Carp	whole fish	09/06/1978	5 U
DePere to Green Bay Reach	8405(a)	Walleye	fillet and skin	01/01/1984	5 U
DePere to Green Bay Reach	8406(a)	Walleye	fillet and skin	01/01/1984	5 U
DePere to Green Bay Reach	8407(a)	Carp	fillet and skin	01/01/1984	5 U
DePere to Green Bay Reach	8305(b)	Walleye	whole fish	06/13/1983	5 U
DePere to Green Bay Reach	8308(c)	Carp	whole fish	10/16/1983	5 U
DePere to Green Bay Reach	8403(a)	Carp	whole fish	01/01/1984	5 U
DePere to Green Bay Reach	8404(a)	Carp	whole fish	01/01/1984	5 U
DePere to Green Bay Reach	8601(j)	Walleye	whole fish	10/06/1986	5 U
DePere to Green Bay Reach	8602(h)	Carp	whole fish	10/06/1986	5 U
DePere to Green Bay Reach	8609(e)	Gizzard Shad	whole fish	10/06/1986	5 U
DePere to Green Bay Reach	7801(h)	Carp	whole fish	08/09/1978	5 U
DePere to Green Bay Reach	7802(h)	Carp	whole fish	08/09/1978	5 U
DePere to Green Bay Reach	7803(g)	Walleye	whole fish	08/11/1978	5 U
DePere to Green Bay Reach	7901(a)	Walleye	whole fish	04/04/1979	5 U
DePere to Green Bay Reach	7902(b)	White Sucker	whole fish	04/04/1979	5 U
DePere to Green Bay Reach	7903(b)	Carp	whole fish	04/04/1979	5 U
DePere to Green Bay Reach	8001(h)	Walleye	whole fish	10/02/1980	5 U
DePere to Green Bay Reach	8002(g)	Carp	whole fish	10/02/1980	5 U
DePere to Green Bay Reach	8003(h)	Carp	whole fish	10/02/1980	5 U
DePere to Green Bay Reach	8101(a)	Walleye	whole fish	03/13/1981	5 U
DePere to Green Bay Reach	8102(a)	Walleye	whole fish	03/13/1981	5 U
DePere to Green Bay Reach	8103(j)	White Sucker	whole fish	09/28/1981	5 U
DePere to Green Bay Reach	8104(i)	Walleye	whole fish	09/28/1981	5 U
DePere to Green Bay Reach	8105(h)	Carp	whole fish	09/28/1981	5 U
DePere to Green Bay Reach	8106(f)	Walleye	whole fish	09/28/1981	5 U
DePere to Green Bay Reach	8201(e)	Walleye	whole fish	08/03/1982	5 U
DePere to Green Bay Reach	8202(d)	Carp	whole fish	08/03/1982	5 U
DePere to Green Bay Reach	8203(d)	Carp	whole fish	08/03/1982	5 U
DePere to Green Bay Reach	8101(j)	Carp	whole fish	08/17/1981	5
DePere to Green Bay Reach	7901(b)	Yellow Perch	whole fish	04/04/1979	5 U
DePere to Green Bay Reach	7902(a)	Brown Bullhead	whole fish	04/04/1979	5 U
DePere to Green Bay Reach	7903(a)	Brown Bullhead	whole fish	04/04/1979	5 U

Table 3 Lead Concentrations in Game Fish Tissue Samples

LOCATION	SAMPLE NUMBER	SPECIES	SAMPLE TYPE	SAMPDATE	RESULT (mg/kg)
DePere to Green Bay Reach	7904(k)	Carp	whole fish	10/17/1979	5 U
DePere to Green Bay Reach	7905(j)	Carp	whole fish	10/17/1979	5 U
DePere to Green Bay Reach	7906(f)	Walleye	whole fish	10/17/1979	5 U
DePere to Green Bay Reach	8201(d)	Carp	whole fish	08/03/1982	5 U
DePere to Green Bay Reach	8202(f)	Walleye	whole fish	08/23/1982	5 U
DePere to Green Bay Reach	8307(e)	Carp	whole fish	10/01/1983	0.5 U
Green Bay	7901(c)	Alewife	whole fish	07/05/1979	5 U
Green Bay	7902(h)	Yellow Perch	whole fish	07/05/1979	5 U
Green Bay	7903(h)	White Sucker	whole fish	07/05/1979	5 U
Green Bay	7904(g)	Brown Bullhead	whole fish	07/05/1979	5 U
Green Bay	7905(e)	Carp	whole fish	07/05/1979	5 U
Green Bay	8102(b)	Carp	whole fish	05/14/1981	5
Green Bay	8103(a)	Carp	whole fish	05/14/1981	5
Green Bay	8104(a)	Carp	whole fish	05/14/1981	5
Green Bay	8105(a)	Carp	whole fish	05/14/1981	5
Green Bay	8304(b)	Carp	whole fish	06/01/1983	5 U
Green Bay	8305(a)	Carp	whole fish	06/01/1983	5 U
Green Bay	7901(a)	Lake Trout	whole fish	06/11/1979	5 U
Green Bay	7902(f)	Longnose Sucker	whole fish	06/11/1979	5 U
Green Bay	7903(f)	Burbot	whole fish	06/11/1979	5 U
Green Bay	7904(d)	Rainbow Smelt	whole fish	06/11/1979	5 U
Green Bay	7901(b)	Lake Trout	whole fish	06/25/1979	5 U
Green Bay	7902(g)	Burbot	whole fish	06/25/1979	5 U
Green Bay	7903(g)	Longnose Sucker	whole fish	06/25/1979	5 U
Green Bay	7904(f)	Alewife	whole fish	06/25/1979	5 U
Green Bay	7905(d)	Rainbow Smelt	whole fish	06/25/1979	5 U
Green Bay	7906(d)	Lake Whitefish	whole fish	07/16/1979	5 U
Green Bay	7901(f)	Lake Whitefish	whole fish	07/26/1979	5 U
Green Bay	7901(d)	Rainbow Trout	whole fish	05/17/1979	5 U
Green Bay	7902(d)	Brown Trout	whole fish	05/17/1979	5 U
Green Bay	7903(d)	Lake Whitefish	whole fish	05/17/1979	5 U
Green Bay	7904(e)	Alewife	whole fish	06/15/1979	5 U
Green Bay	8105(e)	Carp	whole fish	06/16/1981	5
Green Bay	7901(e)	Yellow Perch	whole fish	07/12/1979	5 U
Green Bay	7902(j)	Alewife	whole fish	07/12/1979	5 U
Green Bay	7903(j)	Troutperch	whole fish	07/12/1979	5 U
Green Bay	7904(i)	White Sucker	whole fish	07/18/1979	5 U
Green Bay	7905(g)	Black Bullhead	whole fish	07/18/1979	5 U
Green Bay	7901(e)	Brown Trout	whole fish	05/22/1979	5 U
Green Bay	7901(c)	Walleye	whole fish	04/30/1979	5 U
Green Bay	7902(c)	Lake Trout	whole fish	05/02/1979	5 U
Green Bay	7903(c)	Walleye	whole fish	05/08/1979	5 U
Green Bay	7904(b)	Brown Trout	whole fish	05/08/1979	5 U
Green Bay	7905(b)	Rainbow Smelt	whole fish	05/08/1979	5 U
Green Bay	7906(b)	Yellow Perch	whole fish	05/08/1979	5 U
Green Bay	7907(b)	Burbot	whole fish	05/08/1979	5 U
Green Bay	7908(b)	White Sucker	whole fish	05/08/1979	5 U
Green Bay	7909(a)	Northern Pike	whole fish	05/30/1979	5 U
Green Bay	7901(d)	Carp	whole fish	07/06/1979	5 U
Green Bay	7902(i)	Yellow Perch	whole fish	07/06/1979	5 U
Green Bay	7903(i)	Alewife	whole fish	07/06/1979	5 U
Green Bay	7904(h)	Brown Bullhead	whole fish	07/06/1979	5 U
Green Bay	7905(f)	White Sucker	whole fish	07/06/1979	5 U
Green Bay	8106(e)	Carp	whole fish	09/01/1981	5
Green Bay	8108(d)	Carp	whole fish	09/01/1981	5

Table 3 Lead Concentrations in Game Fish Tissue Samples

LOCATION	SAMPLE NUMBER	SPECIES	SAMPLE TYPE	SAMPDATE	RESULT (mg/kg)
Reference	8702(f)	Walleye	fillet and skin	05/12/1987	5 U
Reference	8704(f)	Walleye	fillet and skin	05/15/1987	5 U

Table 4 Lead Concentrations in Waterfowl Tissue Samples

LOCATION	SAMPLE NUMBER	SPECIES	SAMPLE TYPE	SAMPLE DATE	RESULT (mg/kg)
Dunbar Wildlife Area	31B,C (P)	Woodcock	muscle, no skin	09/05/1984	5.00 U
Green Bay	18B,C (P)	Canada Goose	muscle and skin	07/02/1984	5.00 U
Green Bay	11B,C (P)	Mallard	muscle and skin	08/16/1984	5.00 U
Green Bay	84B,C (P)	Mallard	muscle and skin	12/06/1984	5.00 U
Green Bay	18E,F (P)	Canada Goose	muscle, no skin	07/02/1984	5.00 U
Green Bay	11E,F (P)	Mallard	muscle, no skin	08/29/1984	5.00 U
Green Bay	84E,F (P)	Mallard	muscle, no skin	12/06/1984	5.00 U
Little Lake Butte des Morts	08B,C (P)	Mallard	muscle and skin	07/31/1984	5.00 U
Little Lake Butte des Morts	30B,C (P)	Ring-necked Pheasant	muscle, no skin	09/10/1984	5.00 U
Navarino Wildlife Area	10B,C (P)	Common Merganser	muscle and skin	09/07/1984	5.00 U
Navarino Wildlife Area	09B,C (P)	Mallard	muscle and skin	09/14/1984	5.00 U
Rush Lake	04B,C (P)	Mallard	muscle and skin	08/14/1984	5.00 U
Green Bay	96089 (P)	Canada Goose	unknown	06/19/1996	0.09
Green Bay	96092 (P)	Canada Goose	unknown	06/18/1996	0.05
Lincoln Park	96101 (P)	Canada Goose	unknown	06/24/1996	0.07
Oak	97003 (P)	Canada Goose	unknown	06/26/1996	0.13
Regner Park	96098 (P)	Canada Goose	unknown	06/20/1996	0.04
Rock River Golf Course	97016 (P)	Canada Goose	unknown	07/09/1996	0.03
Sheboygan River	96086 (P)	Canada Goose	unknown	06/19/1996	0.07
Spring Lake Park	97006 (P)	Canada Goose	unknown	06/26/1996	0.10
Villa Du Park	96095 (P)	Canada Goose	unknown	06/20/1996	0.04
Wilson Park	96104 (P)	Canada Goose	unknown	06/25/1996	0.06

Appendix C

Focused Ecological Risk Assessment Upper Green Bay Portion of the Fox River Site, Green Bay, Wisconsin

FOCUSED ECOLOGICAL RISK ASSESSMENT

UPPER GREEN BAY PORTION OF THE FOX RIVER SITE
GREEN BAY, WI
February 2000

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TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

LIST OF ACRONYMS

AHH	Aryl hydrocarbon hydroxylase
Ah receptor	Aryl hydrocarbon receptor
AUF	Area use factor
BW	Body weight
COPC	Contaminant of Potential Concern
DDD	Dichlorodiphenyl dichloroethane
DDE	Dichlorodiphenyl ethylene
DDT	Dichlorodiphenyl trichloroethane
DME	Drug-metabolizing enzyme
ERA	Ecological risk assessment
EROD	Ethoxyresorufin-O-deethylase
FT	Federal threatened
HQ	Hazard Quotient
kg/day	Kilograms per day
km	Kilometer
K_{ow}	Octanol-water partition coefficient
LD_{50}	Median lethal dose
LOAEC	Lowest observed adverse effect concentration
LOAEL	Lowest observed adverse effect level
LOD	Limit of detection
LOQ	Limit of quantification
µg/kg µg/kg	Micrograms per kilogram
µg/L	Micrograms per liter
mg/kg	Milligrams per kilogram
mg/kgBW/day	Milligrams per kilogram body weight per day
NRC	Natural Resource Council
NRDA	Natural Resource Damage Assessment
NOAEC	No observed adverse effect concentration
NOAEL	No observed adverse effect level
PCB	Polychlorinated biphenyl
PCDD	Polychlorinated dibenzo-*p*-dioxins
PCDF	Polychlorinated dibenzofurans
PHHs	Planar halogenated hydrocarbons
QA/QC	Quality assurance/quality control
SE	State endangered
TCDD	2,3,7,8-Tetrachloro-*p*-dibenzodioxin
TCDD-EQ	2,3,7,8-Tetrachloro-*p*-dibenzodioxin equivalents
TCDF	2,3,7,8-Tetrachloro-*p*-dibenzofuran
TEQ	Toxicity equivalents
TRV	Toxicity Reference Value
USGS	United States Geological Survey
UTM	Universal Tranverse Mercator

EXECUTIVE SUMMARY

This document presents the results of a focused ecological risk assessment for the upper Green Bay portion of the Fox River site at Green Bay, Wisconsin. This risk assessment serves as an initial ecological risk evaluation leading toward the baseline risk assessment being performed for the Lower Fox River/lower Green Bay (ThermoRetec Consulting Corporation). The objective of this assessment is to determine whether PCBs do not pose a risk to the upper Green Bay system, or whether further risk evaluation is needed. This risk assessment is not the baseline risk assessment for this site, and is not intended to be utilized to derive cleanup levels. This risk assessment was developed based on the eight-step process described in the Ecological Risk Assessment Guidance for Superfund (U.S. EPA 1997).

The baseline risk assessment conducted for the Lower Fox River/lower Green Bay concluded that polychlorinated biphenyls (PCBs) represented the greatest site-related threat to ecological receptors; calculated risks from PCB exposure were 10 to 1,000 times greater than predicted risk from all other chemicals of potential concern (COPCs; ThermoRetec Consulting Corporation). To focus this risk assessment, PCBs were the only contaminant of concern evaluated for the upper Green Bay. Existing PCB data on concentrations in sediment, water, fish tissue, and bird eggs were utilized to evaluate potential risk from PCBs to ecological receptors in the upper Green Bay. No empirical field or laboratory studies were conducted as part of this risk assessment.

Assessment endpoints selected for this risk assessment focused on upper trophic level receptors, based on the ability of PCBs to bioaccumulate in food chains. Direct toxicity of PCBs to benthic organisms was evaluated in the baseline risk assessment conducted for the Lower Fox River/lower Green Bay (ThermoRetec Consulting Corporation) using whole sediment toxicity tests. No acute or chronic toxicity was observed. Based on existing PCB data for lower and upper Green Bay sediment, and because the lower bay is the primary source of PCB-contaminated sediment to the upper bay (Stratus Consulting, Inc. 1999a), PCB concentrations in upper Green Bay are not expected to exceed levels in lower Green Bay. Therefore, assessment endpoints evaluated in the baseline risk assessment for the lower bay which focused on direct toxicity were not evaluated in this risk assessment. The following assessment endpoints were evaluated:

- Pelagic Fish Reproduction and Survival
- Piscivorous Bird Reproduction and Survival
- Piscivorous Mammal Reproduction and Survival

The risk to fish in the upper Green Bay was evaluated using two lines of evidence: 1) measured PCB concentrations in fish tissue; and 2) estimated PCB concentrations in fish eggs were compared with tissue levels published in the literature which have been shown to result in adverse effects to fish. Risk to piscivorous birds was evaluated using three lines of evidence: 1) Measured concentrations in bird eggs were compared to published adverse effect concentrations; 2) Food chain models were employed to calculate dietary exposure concentrations for piscivorous birds. Results of the food chain models were compared with values in the literature (chronic no observed adverse effect levels [NOAELs] or lowest observed adverse effect levels [LOAELs]) which have been associated with toxic effects in birds; 3) Field studies which have evaluated effects of PCBs on birds inhabiting the upper Green Bay were reviewed to supplement conclusions predicted by the first two lines of evidence. Risk to piscivorous mammals was evaluated using a single line of evidence, the results of a food chain model.

The most substantive risk indicated by this assessment are lines of evidence where hazard quotient (HQ) calculations exceeded 1.0 when the LOAEL was used as the effect level and mean PCB concentrations were used as the exposure concentrations. This occurred for the following lines of evidence:

- Caspian tern egg concentration; toxicity reference value (TRV) of 7.6 (decreased hatching success); HQ = 2.1
- Double-crested cormorant egg concentration; TRV of 7.6 (decreased hatching success); HQ = 1.4
- Caspian tern egg concentration; TRV of 8.0 (increased deformity rate); HQ = 1.9
- Double-crested cormorant egg concentration; TRV of 8.0 (increased deformity rate); HQ = 1.3

- Piscivorous mammal food chain model; Natural Resource Damage Assessment (NRDA) data set mean PCB concentration; HQ = 5.3
- Piscivorous mammal food chain model; combined data set mean PCB concentration; HQ = 2.1

Additionally, the food chain model for piscivorous birds utilizing the double-crested cormorant model resulted in a HQ of 1.6 when the LOAEL and maximum fish concentrations from the NRDA data set were used in risk calculations.

The risk characterization for the first assessment endpoint indicated potential risk to pelagic fish reproduction and survival. Hazard quotients calculated using the no observed adverse effect concentration (NOAEC) as the effect level exceeded 1.0 for both egg and whole-body fish tissue PCB concentrations (ranging from 3.9 to 9.5 and 4.0 to 9.4, respectively).

The weight of evidence used to evaluate risk to piscivorous bird reproduction and survival indicates that piscivorous birds are at risk from PCB exposure in the upper Green Bay. Results of the food chain models indicate greater risk using the double-crested cormorant model, which correlates with results observed in field studies (higher deformity rates in Green Bay cormorants than at reference sites).

The food chain model used to evaluate risk to piscivorous mammals indicates they are at risk from PCB exposure at concentrations measured in fish collected in the upper Green Bay. All hazard quotients calculated for the receptor species, mink, exceeded 1.0, and ranged from 2.1 (LOAEL as the effect concentration, mean overall fish PCB concentration) to 397.8 (NOAEL as the effect concentration, maximum fish PCB concentration from the NRDA data set).

1.0 INTRODUCTION AND SITE HISTORY

1.1 Introduction

This ecological risk assessment for the upper Green Bay portion of the Fox River site is a focused evaluation of risk to ecological receptors from PCBs present in the upper section of the bay. This risk assessment serves as an initial ecological risk evaluation leading toward the baseline risk assessment being performed for the Lower Fox River/lower Green Bay (ThermoRetec Consulting Corporation). The objective of this assessment is to determine whether polychlorinated biphenyls (PCBs) do not pose a risk to the upper Green Bay system, or whether further risk evaluation is needed. This risk assessment is not the baseline risk assessment for this site, and is not intended to be utilized to derive cleanup levels. The ecological risk assessment presented here was developed according to the eight step process described in the Ecological Risk Assessment Guidance for Superfund (U.S. EPA 1997).

1.2 Site History

The upper Green Bay portion of the Fox River system is located in Green Bay, Wisconsin (Figure 1). Contamination in the upper Green Bay originated from industrial activities, agricultural activities, and residential surface water runoff along the Lower Fox River, which flows into Green Bay. The Lower Fox River, at the time of this study, was one of the most industrialized rivers in Wisconsin. The Lower Fox River (from Lake Winnebago to Green Bay) had the greatest concentration of pulp and paper mills in the world (ThermoRetec Consulting Corporation). It had received discharges from 15 pulp and/or paper mills, 8 municipal wastewater treatment plants, and one electric generating facility (ThermoRetec Consulting Corporation). As a result of industrial activities, between 190,000 and 375,000 kilograms (kg) of PCBs were discharged into the Lower Fox River from 1954 to the present (ThermoRetec Consulting Corporation). In the mid-1960's, organochlorines were detected in herring gulls nesting on Sister Island in the upper Green Bay (Keith 1966). In the early 1970s, PCBs were detected in water and sediments from the Lower Fox River, and both the Fox River and lower Green Bay were found to contain fish and birds with detectable levels of PCBs in their tissues. The Fox River contributed approximately 92 percent of the PCB loading into the bay in 1989 (DePinto et al. 1994). Other studies have identified up to 362 total contaminants present in sediment, water and biota collected from the Lower Fox River and lower Green Bay (ThermoRetec Consulting Corporation). A more extensive description of the history of the Fox River site is available in the baseline risk assessment for the Lower Fox River and lower Green Bay, which is being prepared through the Wisconsin Department of Natural Resources (ThermoRetec Consulting Corporation).

The risk assessment for the Lower Fox River and lower Green Bay evaluated risks from multiple organic and inorganic contaminants in the Fox River from the outlet of Lake Winnebago to Green Bay and in Green Bay from the outlet of the Fox River to Chambers Island (lower Green Bay). However, contaminants from the Fox River site have been shown to have migrated into upper Green Bay (from Chambers Island up to Lake Michigan, Figures 1 and 2). For example, studies of PCB deposition in bay sediment have shown that PCB-contaminated sediment extends northward along the Door Peninsula for many miles beyond the boundaries of the lower Green Bay (Manchester-Neesvig et al. 1996). Hawley and Niester (1993) estimated that approximately 10 to 33 percent of tributary sediment (most of which is from the Fox River) discharged into the lower bay was transported to the upper bay annually. Based on particle settling velocities in the lower bay and around Chambers Island, Eadie et al. (1991) concluded that PCBs adsorbed to particulate matter can be transported many kilometers (km) before settling. Increased suspended sediment loads from the lower to upper bay were measured during a storm event in September of 1989 (Hawley and Niester 1993). Finally, statistical evaluation of PCB congener patterns in sediment from Lower Fox River, lower Green Bay, upper Green Bay, and Lake Michigan indicates that the

PCB congener pattern in upper Green Bay is more similar to that in lower Green Bay and is unlikely to have been derived from the transport and weathering of Lake Michigan sediment (Stratus Consulting, Inc. 1999a). These studies show that although lower Green Bay is the primary depositional zone for Fox River PCBs, PCBs adsorbed to sediment are being transported from the lower bay to the upper bay.

In addition, available data showed that fish from the upper Green Bay had elevated concentrations of PCBs in their tissue (U.S. EPA 1996; Stratus Consulting, Inc. 1999b and 1999c; Hagler Bailly Services, Inc. 1997). Therefore, since PCBs are one of the primary contaminants associated with the Fox River site, the upper Green Bay is within the extent of contamination of that site. To date, the ecological risks in the upper Green Bay have not been assessed. Therefore, the current risk assessment will evaluate ecological risks in the upper Green Bay from contaminants associated with the Fox River site.

2.0 PROBLEM FORMULATION (Steps 3 and 4, U.S. EPA 1997)

The purpose of problem formulation is to establish the goals, extent, and focus of the baseline ecological risk assessment for the upper Green Bay portion of the Fox River investigation. Problem formulation constitutes Steps 3 and 4 of the U.S. EPA guidance. In the problem formulation phase, the questions and issues that need to be addressed are defined based on potentially complete exposure pathways and ecological effects. Only after these questions and issues are carefully defined should the ecological risk characterization be initiated. The problem formulation presented here is developed according to the guidelines established in the Ecological Risk Assessment Guidance for Superfund (U.S. EPA 1997).

2.1 Contaminants of Potential Concern (COPCs)

A screening level risk assessment was conducted for the Lower Fox River and lower Green Bay (ThermoRetec Consulting Corporation). The contaminants of potential concern (COPCs) were identified based on their concentrations in sediment, surface water, and biota collected from the Fox River and/or lower Green Bay relative to benchmarks; concentration thresholds that represent little or no risk. The volume and spatial extent of each contaminant was also considered when selecting the contaminants to be evaluated. The COPCs selected for further evaluation in the Baseline Risk Assessment for the Lower Fox River/lower Green Bay site were:

- PCBs (total and/or Aroclor 1242)
- 2,3,7,8-Tetrachloro-*p*-dibenzodioxin (2,3,7,8-TCDD or dioxin)
- 2,3,7,8-Tetrachloro-*p*-dibenzofuran (2,3,7,8-TCDF or furan)
- 4,4' Dichlorodiphenyl trichloroethylene (DDT) and its metabolites (dichlorodiphenyl ethylene [DDE], dichlorodiphenyl dichloroethane [DDD])
- Dieldrin
- Arsenic
- Lead
- Mercury

Risk assessment of the above COPCs in the Lower Fox River and lower Green Bay provided a basis for the contaminant selection for the upper Green Bay risk assessment. The baseline risk assessment conducted for the Lower Fox River/lower Green Bay concluded that PCBs represented the greatest threat to ecological receptors. Calculated risks from PCB exposure were 10 to 1,000 times greater than predicted risk from all other COPCs (ThermoRetec Consulting Corporation). It has been well documented that PCBs are the most widespread and dominant contaminant in the Fox River/Green Bay system (ThermoRetec Consulting Corporation 1998); the sources, transport, and fate of PCBs within this system have been extensively characterized. Based on spatial and temporal distributions of PCBs and a statistical analysis of congener patterns in sediment, the Fox

River and the lower Green Bay have been identified as the primary source of PCBs to the upper Green Bay (Stratus Consulting, Inc. 1999a). While the possibility was not discounted that the other contaminants listed above could potentially pose an ecological risk in the upper Green Bay, it was assumed, for the purposes of this risk assessment, that the risk posed by those contaminants was sufficiently evaluated in the ecological risk assessment for the Lower Fox River/lower Green Bay. Therefore, PCBs (as total PCBs) were the only contaminant of concern (COC) evaluated in this risk assessment for the upper Green Bay.

2.2 Ecological Effects of PCBs

The most studied biochemical effect of PCBs in animals is the induction of hepatic mixed function oxidase systems, increasing an organism's capacity to biotransform or detoxify xenobiotic chemicals. Enzymes in this system are sometimes referred to as drug-metabolizing enzymes (DMEs) (Kluwe et al. 1979). Although the increased capacity to detoxify xenobiotic chemicals may appear to benefit an organism, the metabolism of the foreign chemicals can also produce metabolites that are more toxic than the parent compound (Mitchell et al. 1976). In addition, PCB-induced changes in enzyme activity may also alter enzyme substrate concentrations in other metabolic pathways (Montz et al. 1982). Polychlorinated biphenyls also induce microsomal hepatic enzyme systems that metabolize naturally occurring steroid hormones (Peakall 1975). The degree of this enzyme system response has been found to be positively dose-related (Linzey 1987). Polychlorinated biphenyl-induced effects to these hepatic enzyme systems can result in increased liver weight, fatty degeneration, hyalin degeneration, necrosis, hepatocyte formation, and increased hormone metabolism in animals (Batty et al. 1990, Lincer and Peakall 1970, Sanders and Kirkpatrick 1977, Sanders et al. 1974, Stotz and Greichus 1978, Vos 1972, Welsch 1985).

Chlorinated hydrocarbons such as PCBs have been documented as a cause of reproductive dysfunction and mortality in wildlife species (Heaton et al. 1995, Hoffman et al. 1986, Langford 1979). Exposure to PCBs has been found to reduce litter sizes at birth, reduce number of litters, induce longer birthing intervals in mice (Linzey 1987, Merson and Kirkpatrick 1976), and reduce plasma concentrations of estradiol and progesterone in female rats (Johnson et al. 1976). Transplacental movement of PCBs has been reported for humans, rabbits, monkeys, and rats (Storm et al. 1981) causing a dose-dependent reduction in the body weights and survival of exposed mammalian offspring (prenatally as well as postnatally) (Barsotti et al. 1976, Brezner et al. 1984, Fein et al. 1984, Heaton et al. 1995, Wren et al. 1987a,b). Transfer of PCBs to mammalian offspring continues via mother's milk (Wren et al. 1987a). Polychlorinated biphenyls have been implicated as the cause of low embryonic weight in black-crowned night herons (*Nycticorax nycticorax*) (Hoffman et al. 1986). Persistence in courtship behavior was reduced in PCB-fed mourning doves (*Zenaida macroura*) (Tori and Peterle 1983). Reduced sperm concentration, thin-shelled eggs, poor hatching success, and offspring born with teratogenic abnormalities have also been reported (Abrahamson and Allen 1973; Bird et al. 1983; Lowe and Stendell 1991; Scott 1977). Polychlorinated biphenyls have also been shown to transfer from the adult to eggs in fish (Niimi 1982; Mac and Schwartz 1992) and have been implicated in reduced hatching success, larval mortality, and larval growth of fish (Mac and Schwartz 1992; Hendricks et al. 1981; Mac and Edsall 1991; Mac et al. 1993). A more extensive review of the toxic effects caused by PCBs to fish, birds, and mammals can be found in Appendix A.

Much of the toxicity caused by PCBs has been attributed to the planar congeners that resemble TCDD (Geisy et al. 1994). The toxic nature of some prepared PCB mixtures may be associated with trace levels of compounds having four or more chlorine atoms at both the *para* and *meta* positions (Koslowski et al. 1994). This chlorine substitution pattern increases the structural similarity of the congeners to TCDD (Safe 1994). Planar PCBs have affinity for the same cellular receptor (the aryl hydrocarbon or Ah receptor) as TCDD. Dioxin-like PCBs elicit toxic biological responses in animals such as hepatic damage, weight loss, thymic atrophy, dermal disorder,

reproductive toxicity, immunosuppresion, teratogenicity, and functional effects to the spleen, adrenal gland and testes (Batty et al. 1990; Sanders et al. 1974).

The specific toxicity reference values (TRVs) selected were based, in part, on the measurement endpoints selected for the risk assessment. A brief discussion on the derivation of TRVs for this risk assessment is provided in Section 4.2; TRV selection is described in detail in Appendix A. The selection of TRVs for the upper Green Bay risk assessment mirrors, as much as possible, the TRVs selected for the Lower Fox River/lower Green Bay risk assessment (ThermoRetec Consulting Corporation).

2.3 PCB Fate and Transport

PCBs are a group of 209 synthetic halogenated aromatic hydrocarbons that are extremely stable, and are resistant to most chemical and biological degradation processes (Eisler 1986; Hornshaw et al. 1983). The persistence of PCBs in the environment is due to their stable carbon-halogen bonds (Risebrough et al. 1968). In general, PCBs have low aqueous solubility (Chou and Griffin 1986), and are lipophilic (Risebrough et al. 1968), allowing them to accumulate in fatty tissues (Hornshaw et al. 1983).

Upon entering an aquatic system, PCBs partition between the water, sediment, particulate matter, and biota (Koslowski et al. 1994). The more lipophilic and hydrophobic a substance, the more concentrated it will be in the sediment and phytoplankton of an aquatic system (Loizeau and Menesguen 1993); PCBs are highly lipophilic and hydrophobic. While it has been shown that transport of PCBs in the dissolved phase can be important during the warmer low flow periods of summer, PCBs generally sorb strongly to sediment particles. It has been shown that PCBs discharged to aquatic environments rapidly sorb to particles and are usually deposited in sediment, often close to the area of discharge (Kalmaz and Kalmaz 1979). After this, dispersal and movement of PCBs in aquatic systems depends largely on the movement of the associated sediment (Connell and Miller 1984).

The fact that sediment transport plays such a significant role in PCB transport in aquatic systems has direct consequences in Green Bay. Studies have indicated that a distinct depositional zone is located northeast of the mouth of the Fox River, along the eastern shore of Green Bay, and extends approximately 27 miles into the bay (Manchester-Neesvig et al.1996). However, based on measurements of suspended sediment mass flux, Hawley and Niester (1993) estimated that approximately 10 to 33 percent of tributary sediment discharged into the lower bay (the majority of which comes from the Fox River) is transported to the upper Green Bay annually. Similarly, in a study of particle settling velocities in the lower bay and around Chambers Island, Eadie et al. (1991) concluded that PCBs adsorbed to particulate matter can be transported many kilometers within Green Bay before settling. Surface water transport may also be important, as PCBs can be transported in the dissolved phase as well as the particulate phase. Movement of surface water from the lower bay to the upper bay has been documented; most of the flow from the lower to the upper bay occurs along the east side of Chambers Island (Stratus Consulting, Inc. 1999a).

Sedimentation rates in upper Green Bay are generally low, indicating less sediment accumulation than in the lower bay (Manchester-Neesvig et al. 1996). Sediment deposition in the middle of the upper bay may result from events such as storms that move contaminated sediments northward from the lower bay (Manchester-Neesvig et al, 1996). For example, Hawley and Niester (1993) detected an increase in suspended sediment loads from the lower to upper bay during a storm in September of 1989. These studies show that although lower Green Bay is the primary depositional zone for the Fox River, PCB-contaminated sediment is transported with surface water moving from the lower to the upper Green Bay.

Additional potential sources of PCBs to Green Bay include atmospheric deposition and influx from Lake Michigan. Atmospheric deposition has been identified as an important source of PCBs to Lake Superior (Eisenreich et al. 1981) and southern Lake Michigan (Murphy et al. 1981). However, Sweet et al. (1991) estimated that atmospheric deposition of PCBs accounts for less than 10 percent of the total input to Green Bay. The potential contribution from Lake Michigan is unknown; transport of PCBs within the lakes is generally via sediment or biota (Simmons 1984). DePinto et al. (1994) identified the Fox River as the major source of PCBs to Green Bay; they estimated the Fox River contributed 92 percent of the PCB loading to the bay in 1989. To focus this risk assessment, the assumption was made that the primary source of PCBs to Green Bay was the Lower Fox River.

Because PCBs are extremely lipid-soluble, they tend to accumulate in the lipid component, internal organs, and mesenteric fat of organisms (Eisler 1986). Optimum accumulation of PCBs by aquatic biota occurs when planar molecules are substituted with 5 to 7 chlorine atoms (Shaw and Connell 1984). Rapid gill uptake of PCBs has been observed in short-term laboratory experiments with fish (Bruggerman et al. 1981). Generally, when equally exposed, fish accumulate two to three times more PCBs than aquatic invertebrates (Eisler 1986). Once absorbed, PCBs generally partition into the fatty tissues of organisms (Ernst et al. 1976, Phillips 1980, Shaw and Connell 1984). Initially, PCBs concentrate in liver, blood, and muscle; eventually accumulations are highest in adipose tissue and skin. PCB concentrations in a salmonid population were found to be related to fish size as well as fish age (Madenjian et al. 1994).

Controversy exists regarding the relative contribution of food versus direct uptake in determining PCB levels in the tissues of aquatic biota (Rasmussen et al. 1990). Field-collected fish were found to have significantly greater PCB body burdens than laboratory specimens exposed to identical concentrations in water, suggesting that food-chain transfer of PCBs is an important mode of contaminant transfer for top predators (Thomann 1981). Thomann et al. (1992) suggest that PCBs with octanol-water partition coefficients (K_{ow}) greater than 10^5 seem to enter the biota via food-web transfer originating from sediment sources, as opposed to direct uptake from water. Madenjian et al. (1998) indicated that lake trout retain 80 percent of the PCBs that are contained in their food, and concluded that most of the PCB body burden accumulated by lake trout is from their food. Based on the above studies, it was assumed for this risk assessment that dietary uptake was the major route of exposure for upper trophic level organisms.

2.4 Ecological Setting

Green Bay is located in Lake Michigan, in northeastern Wisconsin, within the eastern ridges and lowlands of the state. Green Bay extends 192 km from the mouth of the Fox River northeast to Lake Michigan. Rock Island, Washington Island, and St. Martin's Island mark the separation between Green Bay and Lake Michigan (Figure 2). The largest width of Green Bay is 37 km. The Fox River is the primary tributary to lower Green Bay. Green Bay drains approximately 40,470 square km, which is one-third of the total drainage of Lake Michigan (ThermoRetec Consulting Corporation). The total surface area of Green Bay is 4213 km^2, of which the upper Green Bay comprises 3260 km^2 (Gaude 1998). Lower Green Bay is fairly shallow and provides habitat for warm-water fish; half of this area is less than 9.1 m deep. Upper Green Bay is characterized by deeper water, with about 85% of the area more than 9.1 m deep; the upper bay provides mostly deep, cold-water habitat (Stratus Consulting, Inc. 1999b). The deepest part of the bay is 53.6 m deep and is located 6.4 km west of Washington Island.

The benthic community in the bay is expected to consist of a variety of invertebrates, including insects, annelids, molluscs, and crustaceans. A variety of wildlife species are also known or expected to inhabit Green Bay (ThermoRetec Consulting Corporation; Heinz et al. 1984; Ankley et al. 1992). Some of the wildlife species that are expected to use the bay for food or habitat are

listed below (ThermoRetec Consulting Corporation):

Fish

Common name	Scientific Name
Walleye	*Stizostedion vitreum*
Lake Trout	*Salvelinus namaycush*
Brown Trout	*Salmo trutta*
Rainbow Trout	*Oncorhynchus mykiss*
Brook Trout	*Salvelinus fontinalis*
Chinook Salmon	*Oncorhynchus tshawytscha*
Coho Salmon	*Oncorhynchus kisutch*
White Bass	*Morone chrysops*
White Sucker	*Catostomus commersoni*
Carp	*Cyprinus carpio*
Channel Catfish	*Ictalurus punctatus*
Brown Bullhead	*Ameiurus nebulosus*
Shortnose Sturgeon	*Acipenser brevirostrum*
Lake Sturgeon	*Acipenser fulvescens*
Rainbow Smelt	*Osmerus mordax*
Yellow Perch	*Perca flavescens*
Black Crappie	*Pomoxis nigromaculatus*
American Gizzard Shad	*Dorosoma cepedianum*
Emerald Shiner	*Notropis atherinoides*
Alewife	*Alosa pseudoharengus*
Minnow spp.	Family *Cyprinidae*
Darter spp.	*Etheostoma spp., Percina spp., Ammocrypta spp.*

Birds

Common Name	Scientific Name
Mallard	*Anas platyrhynchos*
Double-crested Cormorant	*Phalacrocorax auritus*
Forster's Tern	*Sterna forsteri* (SE)
Common Tern	*Sterna hirundo* (SE)
Caspian Tern	*Sterna caspia* (SE)
Black Tern	*Chlidonias niger*
Herring Gull	*Larus argentatus*
Little Gull	*Larus minutus*
Ring-billed Gull	*Larus delawarensis*
Bald Eagle	*Haliaeetus leucocephalus* (FT)
Red-breasted Merganser	*Mergus serrator*
Common Merganser	*Mergus merganser*
Black-crowned Night Heron	*Nycticorax nycticorax*
Green Heron	*Butorides striatus*
Red-winged Blackbird	*Agelaius phoeniceus*
Tree Swallow	*Tachycineta bicolor*

Mammals

Common Name	Scientific Name
Mink	*Mustela vison*
River Otter	*Lutra canadensis*
Muskrat	*Ondatra zibethicus*
Beaver	*Castor canadensis*

SE = Endangered according to the state of Wisconsin
FT = Threatened according to the U.S. Fish and Wildlife Service

2.5 Complete Exposure Pathways

As discussed previously, a large volume of sediment is transported from the Fox River and the inner depositional zone of the lower Green Bay to the upper Green Bay each year. Since PCBs have very low water solubility and a high octanol-water partition coefficient, they are likely to sorb strongly to sediment and thus be transported with the sediment into the upper Green Bay. A variety of organisms reside in and around the upper Green Bay and use the bay for food and/or habitat. It is possible that these organisms are exposed to the contaminants that have been transported into the upper Green Bay from the lower Green Bay.

Benthic invertebrates inhabit upper Green Bay, and are in constant contact with sediment. They are potentially exposed to contaminants via direct contact with sediment and sediment interstitial water. In addition, some benthic invertebrates consume sediment to obtain food. They are also potentially exposed to contaminants by ingesting contaminated food items. Some benthic invertebrates may also be exposed to contaminants via direct contact with surface water since some of these organisms inhabit the top layer of sediment, while others inhabit burrows which are constructed to allow for circulation of surface water throughout the burrow.

Upper Green Bay is inhabited by numerous fish species which occupy different regions of the bay. Benthic fish, such as catfish, feed primarily on the bottom substrate of the bay, ingesting relatively large quantities of sediment, periphyton, and benthic invertebrates. These fish may be exposed to contaminants in the bay via ingestion of contaminated food and water, incidental ingestion of contaminated sediment, and direct contact with contaminated sediment and water. Other species of fish inhabit the open water of the bay and feed on phytoplankton and zooplankton. These fish may be exposed to contaminants via ingestion of contaminated food and water and via direct contact with contaminated water. Since some of these open water fish may also feed on benthic organisms, they may also be exposed to contaminants by ingesting contaminated sediment and benthic invertebrates. Upper trophic level fish, such as walleye, feed on other fish and also inhabit the upper Green Bay. These upper trophic level fish may be exposed to contaminants in the bay by consuming other fish that have accumulated contaminants in their tissues. In addition, these fish may be exposed to contaminants via direct contact with contaminated water or sediment, ingestion of contaminated water, or incidental ingestion of contaminated sediment. Finally, some of these upper trophic level fish may also obtain all or a portion of their diet from benthic invertebrates, and thus may be exposed to contaminants via ingestion of benthic invertebrates or incidental ingestion of sediment. Fish that inhabit the open water of the bay are not expected to spend significant time in contact with the sediment. Therefore, direct contact with contaminated sediment is not expected to be a significant exposure pathway for these types of fish.

Other organisms which utilize the bay for food include a variety of bird species. These birds may potentially be exposed to contaminants by ingesting contaminated food items. They may also be exposed to contaminants via ingestion of contaminated water, incidental ingestion of contaminated sediment, direct contact with contaminated sediment and water, and inhalation.

Mammals, such as mink, also utilize the bay for food. Such mammals may inhabit the islands or shores of the bay and feed on fish or invertebrates. Fish, migrating upstream in the tributaries from Green Bay to spawn, may be consumed by terrestrial mammals utilizing the banks of these rivers. Therefore, mammals inhabiting the banks of Green Bay and its tributaries may potentially be exposed to contaminants by ingesting contaminated food items. They may also be exposed to contaminants by ingestion of contaminated water, incidental ingestion of contaminated sediment, direct contact with contaminated sediment and water, and inhalation.

Particularly lipophilic contaminants are known to adsorb to sediments. Since it has been shown that sediment is transported into the upper Green Bay from the lower Green Bay, lipophilic contaminants in the lower Green Bay are also likely to be transported into the upper Green Bay. Another characteristic of lipophilic contaminants is that they are transported across biological membranes more readily than non-lipophilic contaminants and thus are absorbed by biological organisms readily. Once absorbed, they tend to be stored in fatty tissues, allowing for the accumulation of lipophilic contaminants in these tissues via bioaccumulation. As these contaminants are transferred through the food chain, the concentrations in higher trophic level organisms become greater than concentrations in lower trophic level organisms. This process is known as biomagnification and is of particular importance when evaluating the effects of lipophilic contaminants on upper trophic level receptors. PCBs have high octanol-water partition coefficients and are very lipophilic. Therefore, PCBs are expected to accumulate in receptor tissues and to biomagnify through the food chain. This underscores the significance of the potential exposure pathway through the food chain for upper trophic level fish, birds, and mammals in the upper Green Bay.

It should be noted that the dermal contact and inhalation pathways of exposure were not evaluated. Exposure via these routes is difficult to quantify because little information is available in the literature on exposure rates and contaminant effects via these pathways. For this risk assessment, these exposure pathways were assumed to be insignificant compared to ingestion, due to the ability of PCBs to biomagnify through the food chain.

2.6 Assessment Endpoints

Assessment endpoints are explicit expressions of the actual ecological resources that are to be protected. Valuable ecological resources include those without which ecosystem function would be significantly impaired, or those providing critical resources (e.g., habitat). Appropriate selection and definition of assessment endpoints is critical to the utility of a risk assessment as they focus risk assessment design and analysis. It is not practical or possible to directly evaluate risks to all of the individual components of the ecosystem at the site, so assessment endpoints are used to focus the risk assessment on particular components of the ecosystem that could be adversely affected by the contaminants associated with the site. In general, the assessment endpoints selected for the site were aimed at aquatic and terrestrial organism reproduction and survival.

As discussed in Section 1.1, this risk assessment is an extension of the risk assessment conducted for the Lower Fox River/lower Green Bay (ThermoRetec Consulting Corporation). Therefore, the assessment endpoints for the upper Green Bay stem from those used for the Lower Fox River/lower Green Bay study, which were:

- Functioning Water Column Invertebrate Communities
- Functioning Benthic Invertebrate Communities
- Benthic Fish Reproduction and Survival
- Pelagic Fish Reproduction and Survival
- Insectivorous Bird Reproduction and Survival
- Piscivorous Bird Reproduction and Survival
- Omnivorous Bird Reproduction and Survival
- Piscivorous Mammal Reproduction and Survival

A subset of these assessment endpoints was evaluated for the upper Green Bay. Since the only contaminant to be evaluated in the upper Green Bay risk assessment was PCBs (as discussed in Section 2.1) the assessment endpoints for the upper Green Bay focused on upper trophic level receptors (fish, birds, and mammals). This is because exposure to PCBs in the upper Green Bay is

primarily an issue of bioaccumulation and biomagnification rather than direct toxicity; PCBs are not acutely toxic at levels generally found in the environment. For example, information in the literature indicates that PCBs are not expected to have direct toxic effects on benthic invertebrates at levels found in the upper Green Bay. In one study, sediment from the lower Fox River and lower Green Bay were tested in whole sediment toxicity tests using four different test species. The sediment was aerated first in order to dissipate ammonia. No acute or chronic toxicity was observed for any of the test species in any of the whole sediment toxicity tests (Ankley et al. 1992). PCBs were measured in the sediment and levels as high as 6.57 milligrams per kilogram (mg/kg) total PCBs were detected (Ankley et al. 1992). Since the sediment PCB concentrations in upper Green Bay are not anticipated to exceed those found in the lower Green Bay, any direct threat to the benthic community was assumed to be sufficiently evaluated through the ecological risk assessment (ERA) of the Lower Fox River/lower Green Bay. In the risk assessment for the Lower Fox River/lower Green Bay, two assessment endpoints were evaluated for direct toxic effects to lower trophic level organisms: functioning water column invertebrate communities, and functioning benthic invertebrate communities. As stated above, since direct toxicity is not the primary concern with regard to PCBs and no toxicity was observed in toxicity tests conducted with lower Green Bay sediment, these assessment endpoints were not evaluated in the risk assessment for the upper Green Bay.

Three of the assessment endpoints used for the Lower Fox River/lower Green Bay ecological risk assessment were selected for evaluation in the risk assessment for the upper Green Bay. These assessment endpoints represent upper trophic level receptors that would be expected to be exposed to PCBs which have bioaccumulated and biomagnified in a PCB-contaminated ecosystem. The three assessment endpoints were selected in light of the open, deep water habitat of the upper bay, and were a subset of the assessment endpoints selected for the Lower Fox River/lower Green Bay risk assessment. Assessment endpoints not evaluated in this risk assessment include insectivorous and omnivorous birds. Since the selected assessment endpoint trophic groups feed at higher trophic levels and PCBs bioaccumulate, PCB exposure of the selected trophic groups should be higher. By evaluating and protecting these endpoints which are expected to have the greatest exposure and be most sensitive to potential adverse impacts from exposure to site-related contaminants, the upper bay ecosystem as a whole should also be protected. The specific assessment endpoints that were evaluated in this risk assessment are listed below.

2.6.1 Assessment Endpoint #1: Pelagic Fish Reproduction and Survival

The first assessment endpoint was aimed at pelagic fish reproduction and survival in the upper Green Bay. Fish serve a vital role in nutrient and energy transfer within the bay. Specifically, fish act as a link between aquatic and terrestrial ecosystems and between the benthic and pelagic environments. Fish that consume benthic organisms are consumed by other fish, who are in turn consumed by terrestrial organisms such as mammals and birds. These predator-prey interactions represent a transfer of energy from and within the aquatic ecosystem. Since the number of organisms supported at any position in a food chain depends upon the limits of the energy supply available, the role of energy transfer played by fish is integral to the productivity of an aquatic ecosystem. Furthermore, since energy and nutrient cycles are delicately balanced, even a small decline in the fish population can have detrimental impacts on the balance of energy within an ecosystem.

Fish typically comprise a large proportion of the biomass in an aquatic ecosystem and fill a wide range of trophic positions (e.g., predatory, bottom feeders). Fish serve as predators of zooplankton, periphyton, benthic invertebrates, and other fish. Some fish also serve as food items for predators that inhabit aquatic ecosystems. Also, some fish themselves are piscivorous and consume lower trophic level forage fish. A viable fish population is therefore imperative for the maintenance of viable populations of organisms

that feed on them and upon which they feed.

Fish are also important recreationally and commercially. It has been shown that declines in fish populations associated with chemical contamination have adversely affected commercial and recreational fishing industries in many areas of the country (NRC 1992, Miller et al. 1993). In some areas this has had a major impact on local economies due to losses from decreased tourism and decreased revenues from the commercial sale of fish.

Fish populations are of particular concern due to their role in energy transfer, their role in regulating populations, and their role in maintaining a productive commercial and recreational fishery. Therefore, the first assessment endpoint was aimed at pelagic fish reproduction and survival in the upper Green Bay.

2.6.2 Assessment Endpoint #2: Piscivorous Bird Reproduction and Survival

The second assessment endpoint was aimed at piscivorous bird reproduction and survival in the upper Green Bay. Piscivorous birds are upper trophic level organisms that rely primarily on fish as food. Foraging behavior of piscivorous birds represents a pathway by which nutrients and energy are transferred from aquatic to terrestrial ecosystems. There is a close relationship between terrestrial and aquatic systems due to the nutrient and energy flow between these systems. Nutrients enter lake ecosystems via surface water runoff, input via streams, and water infiltration through the soil. Energy enters lake ecosystems via sunlight and other biological input such as detritus and leaves. Nutrients and energy are transferred from aquatic to terrestrial ecosystems via biological output. An example of a biological output is the act of a piscivorous bird consuming fish. Nutrient and energy cycles between aquatic and terrestrial systems are delicately balanced. Since nutrients and energy are limiting factors in the production of an ecosystem, the transfer of energy from an aquatic to a terrestrial system is essential. Piscivorous birds provide one mechanism by which nutrients and energy are transferred from aquatic to terrestrial ecosystems and are therefore important in the maintenance of balanced nutrient and energy cycles.

Predators are often required to keep prey numbers in check, and impacts on predators could cause detrimental population explosions in prey species. Such population explosions result in an imbalance in the energy and nutrient allocations among the organisms inhabiting the same ecosystem, resulting in declines of affected populations. In an aquatic ecosystem, piscivorous birds help to keep populations of the fish, upon which they feed, in check. By keeping fish populations in check, piscivorus birds indirectly impact population fluctuations of invertebrates and other aquatic organisms. The result is balanced populations of fish and invertebrates, which has commercial, recreational, and ecological benefits.

Piscivorous birds can also be preyed upon by other organisms at even higher trophic levels, such as other birds and mammals. By serving as a food source for these higher trophic level organisms, piscivorous birds also function to maintain the population balance of these higher trophic levels. If the populations of piscivorous bird species declined, the populations of the organisms that prey on piscivorous birds might also decline.

Since piscivorous birds are upper trophic level predators, they are especially susceptible to exposure to contaminants that have accumulated in the organisms upon which they feed. In a freshwater system, birds are common predators of fish. Fish have been shown to accumulate contaminants that are present in aquatic ecosystems. Therefore, birds that

consume fish have the potential to accumulate large concentrations of contaminants in their tissue.

Some birds are resident year-round and some are migratory. The variable mobility of potential avian receptors, the relatively large home range, varied diet, and the often seasonal residency, suggest that the potential for exposure, and the identification of specific exposure routes and concentrations is associated with some uncertainty. Nonetheless, the avian piscivore community is of particular concern due to the potential for exposure and adverse effects in a higher trophic level organism, their role in regulating populations, and their role in energy transfer. Therefore, the second assessment endpoint was aimed at the reproduction and survival of piscivorous birds in the upper Green Bay.

2.6.3 Assessment Endpoint #3: Piscivorous Mammal Reproduction and Survival

The third assessment endpoint was aimed at piscivorous mammal reproduction and survival in the upper Green Bay. Piscivorous mammals are upper trophic level organisms that rely primarily on fish as forage. Foraging behavior of piscivorous mammals represents another pathway by which nutrients and energy are transferred from aquatic to terrestrial ecosystems. As stated above, there is a close relationship between terrestrial and aquatic systems due to the nutrient and energy flow between these systems, and piscivorous mammals provide one mechanism by which nutrients and energy are transferred between ecosystems. Piscivorous mammals can be preyed upon by other organisms at even higher trophic levels, such as other mammals and birds. By serving as a food source for these higher trophic level organisms, piscivorous mammals also function to maintain the population balance of these higher trophic levels. If the populations of piscivorous mammal species declined, the populations of the organisms that prey on piscivorous mammals might also decline.

Since piscivorous mammals are upper trophic level predators, they are especially susceptible to exposure to contaminants that have accumulated in the organisms upon which they feed. In a freshwater system, mammals are common predators of fish. Fish have been shown to accumulate contaminants that are present in aquatic ecosystems. Therefore, mammals that consume fish have the potential to accumulate large concentrations of contaminants in their tissues.

Although the shore area of upper Green Bay is limited relative to the area of the bay itself, piscivorous mammals foraging along the shoreline may be exposed to PCB-contaminated fish. In addition, it is possible that fish migrating upstream in the tributaries of Green Bay to spawn may be consumed by terrestrial mammals utilizing the banks of these rivers. Therefore, mammals inhabiting these upstream areas have the potential to be exposed to significant levels of contaminants originating from the upper Green Bay. The mammalian piscivore community is of particular concern due to the potential for exposure and adverse effects in a higher trophic level organism and their role in energy transfer. Therefore, the third assessment endpoint was aimed at the reproduction and survival of piscivorous mammals in the upper Green Bay.

2.7 Conceptual Model

The conceptual model utilizes contaminant and habitat characteristics to identify critical exposure pathways to the selected assessment endpoints. At the site, contaminants in the water and sediment may come in contact with the aquatic and terrestrial receptors inhabiting the upper Green Bay and its islands and surrounding areas. The potentially complete exposure pathways are described in

detail in Section 2.5. The assessment endpoints selected for this risk assessment are described in section 2.6. The site conceptual model is illustrated in Figure 3.

It should be noted that selection of exposure pathways evaluated in this risk assessment was partially dependent on the availability of existing site-specific information. No site-specific data on PCB concentrations in phytoplankton, aquatic plants, or benthic organisms were available for the upper bay. Exposure pathways not evaluated due to lack of site-specific tissue PCB concentrations include ingestion of phytoplankton; ingestion of sediment, aquatic invertebrates and plants by dabbling ducks; and ingestion of insects by insectivorous birds. However, the selected receptor species feed at higher trophic levels than the receptors in the pathways not being evaluated. Since PCBs bioaccumulate and biomagnify through the food chain, PCB exposure of the selected receptors should be higher than for herbivorous or planktivorous species. Therefore, protection of selected receptor species should be protective of organisms with lower exposure levels.

Exposure pathways that were evaluated in this risk assessment are as follows:

I. Aquatic Vertebrates (Fish)

- Direct contact with surface water
- Direct contact with sediment
- Ingestion of water
- Incidental ingestion of sediment
- Ingestion of fish

II. Piscivorous Birds

- Ingestion of surface water
- Incidental ingestion of sediment
- Ingestion of fish

III. Piscivorous Mammals

- Ingestion of surface water
- Incidental ingestion of sediment
- Ingestion of fish

2.8 Selection of Receptor Species

Receptor species were selected as representative of organisms within the complete exposure pathways identified above. Selection was based on potential for exposure to PCBs due to feeding habits or habitat use, sensitivity to adverse effects of PCBs, availability of toxicological data, and consistency with receptors selected for the Lower Fox River/lower Green Bay risk assessment.

2.8.1 Pelagic Fish

Lake trout are top level predators with a high fat content and are therefore likely to accumulate large concentrations of PCBs. Information on the life history of lake trout can be found in Appendix B. Historically, lake trout spawned in Green Bay, utilizing spawning grounds mostly located in the upper bay (Thibodeau 1990). Since the Lake Michigan lake trout population crash in the 1940s and 1950s, lake trout have not spawned in Green Bay, although reproduction is occurring in Lake Michigan. However, although successful reproduction of hatchery-reared trout has occurred in Lake Michigan, sustainable recruitment of lake trout into a fishery has not developed (Holey et al. 1995).

Lake trout have been shown to accumulate PCBs to higher concentrations than any other salmonid species in western Lake Michigan, with mean fillet concentrations approximately two times greater than those in brown trout, chinook salmon, brook trout, rainbow trout, or coho salmon (Miller et al. 1993). Madenjian et al. (1998) indicated that lake trout retain 80 percent of the PCBs that are contained in their food. The authors estimate a net trophic transfer efficiency of 0.73 to 0.89 for lake trout between the ages of 5 and 10 years old. This study also indicated that most of the PCB body burden accumulated by lake trout was from their food. Furthermore, among fish species studied, lake trout have been found to be the most sensitive to PCB-caused fry mortality (Walker et al. 1991).

Given the high degree of accumulation of PCBs in lake trout and their sensitivity to PCB reproductive effects, lake trout are an appropriate receptor species to evaluate pelagic fish reproduction and survival in upper Green Bay.

2.8.2 Piscivorous Birds

Two piscivorous bird species were selected as receptor species representative of piscivorous birds which utilize upper Green Bay: Caspian tern and double-crested cormorant. Information on the life history and an exposure profile for the Caspian tern is provided in Appendix B. Terns may be one of the more sensitive avian species to PCB toxicity (Mineau et al. 1984). Caspian terns generally feed on fish, but will also consume eggs and young of other bird species. In addition, the Caspian tern is currently classified as endangered according to the state of Wisconsin. Based on sensitivity to PCBs and the potential for high exposure to PCBs based on feeding habits, Caspian terns were considered to be an appropriate receptor species representative of piscivorous birds for this risk assessment.

The second species selected as representative of piscivorous birds was the double-crested cormorant. Information on the life history and an exposure profile for the double-crested cormorant is provided in Appendix B. Double-crested cormorants are strict piscivores and have the potential for exposure to PCBs in the upper Green Bay via the consumption of fish. Since they are upper trophic level consumers, they have the potential to accumulate PCBs in their tissues to high concentrations. The concentration of PCBs in eggs of double-crested cormorants has been positively correlated with deformities in hatchlings (Giesy et al. 1994), indicating that a mechanism of toxicity leading to adverse effects from exposure to PCBs may exist in double-crested cormorants. Therefore, double-crested cormorants were also considered to be an appropriate receptor species representative of piscivorous birds for this risk assessment.

2.8.3 Piscivorous Mammals

Mink were selected as receptor species representative of piscivorous mammals which utilize the upper Green Bay area. Information on their life history and an exposure profile for the mink are provided in Appendix B. Life history parameters selected for use in the exposure model are conservative (e.g., highest reported ingestion rate and lowest reported body weight); the objective of this risk assessment is to determine whether no ecological risk is present, or whether further evaluation is needed. The use of conservative assumptions minimizes the possibility of concluding risk is not present when a threat actually does exist.

The habitat of mink includes coastal marshes such as those along the western shore of Green Bay (Chapman and Felhamer 1982). Since a large proportion of the mink's diet is

fish, mink would be expected to accumulate PCBs in their tissues via the consumption of PCB-contaminated fish. Furthermore, of the wildlife species tested, mink are the most sensitive species to the toxicity of PCBs (Eisler 1986). For these reasons, mink were considered to be an appropriate receptor species representative of piscivorous mammals for this risk assessment.

2.9 Testable Hypotheses

Testable hypotheses are specific risk questions that are based upon the assessment endpoints. For this risk assessment, the testable hypotheses were as follows:

2.9.1 Assessment Endpoint #1: Pelagic Fish Reproduction and Survival.

Are levels of site-related contaminants sufficient to cause toxic effects or reproductive impairment in fish that inhabit the upper Green Bay?

2.9.2 Assessment Endpoint #2: Piscivorous Bird Reproduction and Survival.

Are levels of site-related contaminants sufficient to cause toxic effects or reproductive impairment in piscivorous birds that utilize the upper Green Bay?

2.9.3 Assessment Endpoint #3: Piscivorous Mammal Reproduction and Survival.

Are levels of site-related contaminants sufficient to cause toxic effects or reproductive impairment in piscivorous mammals that utilize the upper Green Bay?

2.10 Measurement Endpoints

Each of the testable hypotheses was evaluated using one or more measurement endpoints. The number of measurement endpoints chosen for each assessment endpoint was determined by the type of habitat, the mechanism(s) of toxicity, and the availability of existing data. When more than one measurement endpoint was used to evaluate a single assessment endpoint, a weight-of-evidence approach was employed, whereby the measurement endpoints were treated as lines of evidence. The overall risk to each assessment endpoint was then determined based on the results of the evaluation of each line of evidence, having taken into consideration the degree of importance of each line of evidence.

The measurement endpoints were selected to represent the mechanisms of toxicity and exposure pathways for the assessment endpoints, and to answer questions posed by the testable hypotheses for each assessment endpoint. Similar to the assessment endpoints, the measurement endpoints for this study stemmed from the measurement endpoints selected for the risk assessment of the Lower Fox River/lower Green Bay (ThermoRetec Consulting Corporation). The following measurement endpoints, or lines of evidence, were identified for each of the assessment endpoints in this risk assessment:

2.10.1 Measurement Endpoint for Assessment Endpoint #1: Pelagic Fish Reproduction and Survival.

Two lines of evidence were used to assess whether PCBs are likely to adversely affect survival and reproduction of pelagic fish in the upper Green Bay:

First, data on whole-body concentrations of PCBs in upper trophic level fish collected from the upper Green Bay were obtained from the U.S. Fish and Wildlife Service

database that was used for the Natural Resources Damage Assessment (NRDA), and data collected for the Green Bay Mass Balance Model (Hagler Bailly Services, Inc. 1997; Stratus Consulting, Inc. 1999b; U.S. EPA 1996). These concentrations were assumed to be representative of whole-body tissue concentrations of lake trout inhabiting the upper Green Bay. Measured fish tissue concentrations were compared to values cited in the literature which have been shown to result in toxic effects or reproductive impairment of fish.

Second, estimates of fish egg PCB concentrations were calculated from the whole-body fish tissue concentrations using a ratio calculated based on the data presented in Mac et al. (1993). These estimated fish egg concentrations were then Compared with fish egg concentrations of PCBs, derived from the literature, that have been associated with adverse effects in fish.

2.10.2 Measurement Endpoints for Assessment Endpoint #2: Piscivorous Bird Reproduction and Survival

Three lines of evidence were used to evaluate piscivorous bird reproduction and survival in upper Green Bay

First, PCB concentrations measured in bird eggs collected from islands in or near upper Green Bay were ThermoRetec Consulting Corporation concentrations of PCBs in bird eggs cited in the literature which are associated with adverse effects on bird reproduction and survival.

Second, a food chain model for each receptor species was used to estimate daily dietary exposure to PCBs in the upper Green Bay. Data on fish tissue PCB concentrations in the upper Green Bay were obtained from the U.S. Fish and Wildlife Service database used for the NRDA (Hagler Bailly Services, Inc. 1997; Stratus Consulting, Inc. 1999a) and the database compiled for the development of the Green Bay Mass Balance Model (U.S. EPA 1996). Sediment and surface water concentrations to be entered into the food chain models were also obtained from the database compiled for the development of the Green Bay Mass Balance Model. Using the food chain models, a predicted daily PCB dosage was calculated for both receptors. These dosages were then Compared with dietary PCB dosages derived from the literature that were associated with toxic effects in birds.

Third, results from published studies in which the effects of PCBs on birds inhabiting the upper Green Bay were evaluated. This information was used to supplement the conclusions drawn from the first two lines of evidence.

2.10.3 Measurement Endpoint for Assessment Endpoint #3: Piscivorous Mammal Reproduction and Survival.

A food chain model for mink was selected as an appropriate measurement endpoint to assess the risk to piscivorous mammal reproduction and survival in the upper Green Bay from exposure to PCBs. Data on fish tissue concentrations of PCBs in the upper Green Bay were obtained from the U.S. Fish and Wildlife Service database used for the NRDA, and the database compiled for the development of the Green Bay Mass Balance Model (U.S. EPA 1996). Sediment and surface water concentrations entered into the food chain models were obtained from the database compiled for the development of the Green Bay Mass Balance Model. Using the food chain model, a predicted daily PCB dosage was calculated for the mink. This dosage was compared with dietary PCB dosages derived from the literature that are associated with toxic effects in mink.

3.0 ASSUMPTIONS

An attempt was made to utilize conservative assumptions throughout this risk assessment due to the uncertainty associated with the risk assessment process. The use of consistently conservative assumptions minimizes the possibility of concluding that risk is not present when a threat actually does exist (i.e., the elimination of false negatives). While there is uncertainty associated with each conservative assumption used, this consistent selection process assures that the uncertainty associated with this type of error will err on the side of a protective outcome. In some cases, there was sufficient information available to justify the use of less conservative assumptions. The assumptions utilized in this risk assessment are described below.

The following conservative assumptions were made to conduct this risk assessment:

- Maximum contaminant levels measured in tissue and sediment were used in the risk calculations and assumed to be representative of concentrations present site-wide.

- To calculate total PCB concentrations in fish tissue for the upper Green Bay Mass Balance Model, the concentration of each of the PCB congeners measured for each sample were summed. If a particular congener was not detected in a sample, it was assumed to be present as one-half of either its limit of detection (LOD) or its limit of quanitification (LOQ), whichever was reported.

- Contaminants in food items were assumed to be 100 percent bioavailable and not metabolized and/or excreted during the life of the receptor. Most dietary toxicity reference values (TRVs) are based on administered doses in toxicity tests rather than the resulting absorbed doses. Therefore, this assumption probably does not greatly influence the results of the analysis.

- For calculations of an area use factor[1] (AUF) for the mink, the minimum reported home range was used.

- Since most dietary TRVs were derived using dosing intervals shorter than seasonal life history events, it was deemed appropriate to not consider seasonal factors in the life histories of avian receptors for the purposes of this risk assessment. Therefore, breeding territories rather than full migratory ranges were used to calculate AUFs for the Caspian tern and the double-crested cormorant. The portion of the year that these birds have migrated elsewhere and are therefore not utilizing the upper Green Bay was not accounted for in the estimation of their AUFs. It was assumed that these birds are present year round in the upper Green Bay.

- A literature search was conducted to determine the chronic toxicity of PCBs for use in the food chain models. If no toxicity values could be located for the receptor species, values reported for a closely related species were used. Studies were critically reviewed to determine whether study design and methods were appropriate. If values for chronic toxicity were not available, LD_{50} (median lethal dose) values were used. For this study, a factor of 100 was used to convert the reported LD_{50} to a No Observable Adverse Effect Level (NOAEL). A factor of 10 was used to convert a reported Lowest Observable Adverse Effect Level (LOAEL) to a NOAEL. No other safety factors were incorporated into the TRVs selected for this risk assessment. If several toxicity values were reported for a receptor species, the most conservative value was used in the risk calculations as long as the study design, exposure route, mechanism, and species tested were deemed appropriate. For the chronic toxicity endpoints, values obtained from long-term feeding

[1] An area use factor is the ratio of an organism's home range, breeding range, or feeding/foraging range to the area of contamination of a site.

studies were used in preference to those obtained from single dose oral studies.

- A sediment ingestion rate could not be located for mink; estimated sediment ingestion rates were based on those reported in the literature for a similar species, the raccoon. It was assumed that the sediment ingestion rate of the raccoon, as a percentage of dietary intake, was representative of the sediment ingestion rate for the mink.

- In the food chain model, the lowest reported body weights and the highest reported ingestion rates for adults were assumed in each case.

The following assumptions were also made to conduct this risk assessment. Some are not conservative (e.g., mean contaminant levels) while others are realistic (e.g., an area use factor of 1.0 for piscivorous birds).

- Mean contaminant levels measured in tissue and sediment were also used in the risk calculations and assumed to be representative of concentrations present site-wide.

- PCB concentrations measured in walleye and brown trout were assumed to be representative of concentrations in lake trout. Although lake trout have been found to accumulate the highest concentrations of PCBs found in open-water fish of the Great Lakes (Mac and Schwarz 1992), lake trout data collected under rigorous QA/QC procedures were not available for use in this risk assessment.

- Dietary composition information was obtained from the literature for the receptor species evaluated using the food chain models. However, simplifications of complex diets were assumed for the receptors. Since fish were the only food items for which PCB residue data existed, the receptors evaluated using the food chain model were assumed to consume 100 percent fish. Fish were assumed to be appropriate surrogates for all other prey species potentially consumed by receptors.

- It was assumed that Caspian terns and double-crested cormorants could obtain all of their food within the study area.

- Sediment ingestion rates for the Caspian tern and double-crested cormorant could not be found in the literature. However, due to the open water feeding habits of the Caspian tern and the double-crested cormorant, these receptors were assumed to not incidentally ingest sediment.

- Numerous studies have documented greater sensitivity of chickens to TCDD-like toxicity compared with other species. Other species tested include pheasants, mallards, goldeneyes, herring gulls, black-headed gulls, common tern and kestrels (Brunstrom 1988, Brunstrom and Reutergardh 1986, Hoffman et al. 1998); all species tested to date have been considerably less sensitive than chickens (Hoffman et al 1998). Dietary LOAELs reported for chickens ranged from 0.0414 to 0.9 milligrams per kilogram body weight per day (mg/kgBW/day), whereas dietary LOAELs reported for other bird species ranged from 1.12 to 36 mg/kgBW/day (Appendix A, Table A-1). We felt a sufficient number of studies had been conducted with other avian species to conclude that effect levels reported for chickens were an anomaly relative to other bird species. Studies in which chickens were the test species were not selected for derivation of the NOAEL and LOAEL in this risk assessment.

- In some cases, toxicity values in the literature were reported as mg/kg in the diet. These were converted to daily intake (mg/kg BW/day) by using the following formula:

$$\text{Daily Intake (mg/kg BW/day)} = \text{Contaminant Dose (mg/kg diet)} \times \text{Ingestion Rate}$$

(kg/day) x 1/Bodyweight (kg)

This conversion allowed dietary toxicity levels cited to be converted to a daily dose based on body weight. Contaminant doses are exposure levels utilized in studies which evaluated dietary toxicity of PCBs. All studies evaluated to derive the TRVs utilized in this risk assessment are described in detail, including contaminant dose, in Appendix A, Section A.3.1 and A.4. Life history profiles used to derive exposure parameters (ingestion rates and body weights) for receptor species are presented in Appendix B. Values used for this conversion are summarized in Table A-1 (Appendix A).

4.0 METHODS

4.1 Data Compilation (Exposure Characterization)

Data used in support of the ecological risk assessment was obtained from three original sources. First, fish whole-body PCB concentration data were obtained from a database that was developed for use in the NRDA. Second, additional fish whole-body PCB data as well as surface water and sediment PCB data were obtained from the database developed in support of the Green Bay Mass Balance Model (U.S. EPA 1996). Third, bird egg concentrations as well as information on the success of field populations were obtained from studies in the literature. Of these data sets, the one developed for the NRDA was developed under the most rigorous quality assurance/quality control (QA/QC) procedures. The NRDA data were also the most recent data available and thus was given the most weight in this risk assessment. The data collected in support of the Green Bay Mass Balance Model was older data and has not been fully validated using strict QA/QC procedures. Therefore, these data have a higher level of uncertainty than the NRDA data and were therefore given less weight in the risk assessment. The Mass Balance Model data, however, are the most comprehensive data set available for the upper Green Bay and therefore were considered important supporting data for inclusion in the risk assessment. The QA/QC procedures used to validate the bird data collected from the literature are not known. Therefore, these data were considered the least rigorous data set, but they were also considered to be important information in support of the conclusions of the ecological risk assessment because these types of data do not exist elsewhere. In light of the varying degrees of confidence in the different data sets, the data sets were used both separately and combined in the risk assessment to be able to assign a qualitative level of certainty to each of the conclusions.

4.1.1 Surface Water PCB Data

Surface water data were obtained from the Green Bay Mass Balance data set incorporated into the Fox River Database (http://www.ecochem.net/FoxRiverDatabaseWeb/default.asp). This database contains Green Bay Mass Balance data which has been reviewed to eliminate duplicate entries or other anomalies. All data reported for Green Bay Zone IV were utilized in this risk assessment. Green Bay Zone IV, as defined for the Green Bay Mass Balance Model (U.S.EPA 1996), includes the portion of Green Bay north of a line which intersects Chambers Island (Figure 2). Data were reported as “dissolved” and “particulate”; these two fractions were summed to obtain a total PCB concentration for each sample. Any duplicate samples were first averaged to calculate a mean dissolved, particulate and total PCB concentration for that location. Finally, an overall mean and maximum total PCB concentration for surface water in the upper Green Bay was calculated.

Mean and maximum total PCB concentrations in surface water were entered into the food chain models to estimate the expected dosage of PCBs from ingestion of surface water for the Caspian tern, double-crested cormorant, and mink, as described in Section 4.3.2.

4.1.2 Sediment PCB Data

Surface sediment data (0 to 12 inches) were also obtained from the Green Bay Mass Balance data set incorporated into the Fox River Database. All data reported for Green Bay Zone IV were utilized in this risk assessment. Mean and maximum PCB concentrations in surface sediment for the upper Green Bay were calculated. These numbers were used to estimate the expected incidental sediment dosage in the food chain model for the mink.

4.1.3 Fish Whole-Body PCB Data

Fish whole-body PCB data were obtained from two different sources. First, data were available for upper trophic level fish (walleye and brown trout) from the database developed by the U.S. Fish and Wildlife Service and used for the NRDA (Hagler Bailly Services, Inc. 1997; Stratus Consulting, Inc. 1999a). From this database, the total PCB concentration in each sample was calculated by summing the concentrations of each congener and subtracting congener 85 for each sample. Congener 85 was subtracted from the total because the analytical laboratory performing the analysis determined that there was analytical interference with DDE. Due to this interference, it was the opinion of the analytical laboratory, that a sum of all the congeners would have resulted in a gross overestimation of the total PCB concentrations, while the sum of the congener concentrations minus congener 85 was believed to be only a slight underestimation of the total PCB concentrations. Therefore, the congener sum minus congener 85 was determined to be the most appropriate calculation of total PCBs for this data set and was selected for use in this risk assessment.

An overall mean and a maximum total PCB concentration was then calculated for whole-body fish tissue from the NRDA data set. Tissue data from this database are composite samples comprised of three to six fish. The maximum concentration obtained from this data set may underestimate the maximum PCB concentration in individual fish. The resulting concentrations were used both in the food chain models, and in comparisons with fish whole-body PCB concentrations identified in the literature to be associated with adverse effects.

Data for PCB concentrations in fish tissue collected for the Green Bay Mass Balance Model were obtained from a data set extracted from the original Mass Balance Model database and compiled by Stratus Consulting, Inc. in Boulder CO, and from the Mass Balance Model data incorporated into the Fox River Database. Samples from this data set are composite samples comprised of five fish each. In this data set, PCB congener data were available for both upper trophic level fish (walleye and brown trout) as well as forage fish (alewife, carp, and smelt) for the upper Green Bay, corresponding to Region IV of Green Bay for the Mass Balance Model (U.S. EPA 1996). Total PCBs were calculated for each sample by summing the concentrations of each PCB congener detected in each sample. If a particular congener was not detected, it was assumed to be present at one-half of either its LOD or its LOQ, whichever was reported. An overall mean and maximum was calculated separately for forage fish (alewife, carp, and smelt only) and was used in the food chain models. In addition, the data for the upper trophic level fish (walleye and brown trout) were combined with the walleye and brown trout data from the NRDA data set, and an overall mean and maximum total PCB whole-body concentration was calculated from this combined data set. The resulting mean and maximum concentrations were used, as described below, to compare with fish whole-body PCB concentrations that have been associated with adverse effects in the literature.

4.1.4 Fish Egg PCB Data

Estimated concentrations of PCBs in fish eggs were calculated using an egg to whole-body ratio of 0.209 calculated for lake trout using data presented in Mac et al. (1993). Miller (1993) reported mean tissue and egg PCB concentrations in lake trout collected from Lake Michigan; the egg to whole-body ratio calculated from this mean is 0.223. The ratio calculated using the Mac et al. (1993) data was utilized in this risk assessment, as individual fish and egg concentrations were reported rather than means. Similar to the calculation of whole-body data, described in Section 4.1.3, the data obtained from the NRDA database was first taken alone to calculate a mean and a maximum estimated fish egg concentration. To do this, the mean and maximum whole-body PCB concentrations calculated from the NRDA data set were multiplied by 0.209 to obtain the estimated mean and maximum fish egg concentrations for upper trophic level fish from the NRDA data set. A similar calculation (multiplication by 0.209) was performed on upper trophic level fish data from the Mass Balance Model. All estimated fish egg PCB concentrations from the two databases were then combined to obtain an overall mean and maximum estimated PCB concentration in fish eggs. The resulting estimated fish egg PCB concentrations for the NRDA data alone and the combined data were compared with fish egg PCB concentrations in the literature that have been associated with adverse effects in fish, as described in Section 4.3.1.

4.1.5 Bird Egg PCB Data

A variety of published studies have been performed in which bird eggs were collected from islands in and around the upper Green Bay and analyzed for PCBs (e.g., Ewins et al. 1994, Custer et al. in press). These studies were reviewed and data on mean and maximum PCB concentrations reported for bird eggs were compiled. Because these studies were conducted over a broad time span, and since the concentrations of PCBs in bird eggs in the upper Green Bay have generally declined over time (Stratus Consulting, Inc. 1999d), the most recent data available was used to evaluate the present risk from PCBs in the upper Green Bay. Since bird egg PCB data were available for both of the selected receptor species (Caspian tern and double-crested cormorant), data for only these two species were considered.

4.2 Effects Characterization

A comprehensive literature search was conducted to locate studies which evaluated the toxicity of PCBs to ecological receptors. Toxicity reference values (TRVs) were derived based on the results of the literature search. A TRV is a contaminant dose level that is compared with an exposure dose to assess the presence and degree of risk to a receptor or group of receptors from that contaminant. Usually, two TRVs are used to predict ecological risk: a no observable adverse effect level (NOAEL) and a lowest observable adverse effect level (LOAEL). The NOAEL is the highest dose at which adverse effects are not expected to occur, and the LOAEL is the lowest dose at which adverse effects are expected to occur.

Studies located in the literature search were critically evaluated to determine whether they were appropriate to use to derive a TRV. Criteria used to appraise studies included suitability of the test result for evaluating the assessment endpoint, similarity of test organism to selected receptor species, duration of exposure, life stage tested, and ecological relevance of the measured effect. The TRVs selected for this risk assessment were based on high-quality studies which satisfied many or all of the evaluation criteria; they are presented in Table 1. Studies which reported both a LOAEL and NOAEL were selected over studies which reported only one effect level, due to the

uncertainty associated with an unbounded effect level[2]. If only a LOAEL could be identified from the studies, an uncertainty factor of 10 was used to calculate a NOAEL (Dourson and Stara 1983). If a LOAEL could not be located for a receptor, the highest NOAEL was selected, and a factor of 10 was used to calculate a LOAEL. Additional discussion on the TRVs selected for this risk assessment is provided in Section 6.0 (Risk Characterization). The studies used to derive TRVs for this risk assessment are described in detail in Appendix A.

4.3 Methods Used to Evaluate Risk

The hazard quotient (HQ) method (Barnthouse et al. 1986; U.S. EPA 1997) was employed to predict the effects of PCB contamination within the upper Green Bay. This method compares exposure concentrations to ecological endpoints such as mortality, reproductive failure or reduced growth. This is done using chronic toxicity values derived from the literature that are intended to represent a lower dose over a longer duration of exposure, resulting in subtle effects that would be expected to manifest themselves at the population level over the longer term.

The comparisons are expressed as ratios of potential intake values to population effect levels, as follows:

$$\text{Chronic Hazard Quotient} = \frac{\text{Exposure Concentration (Mean or Maximum)}}{\text{Chronic Effect Level (e.g., NOAEL or LOAEL)}}$$

The effect level values for toxicity of PCBs were obtained from published studies, and are summarized in Appendix A. The exposure concentrations and toxicity values were entered into the HQ equation and a HQ was calculated.

If the calculated HQ is greater than one based on a chronic NOAEL, it is an indication that there is a potential chronic risk from that contaminant to the ecological receptor in question. The most significant potential risk is indicated if the HQ exceeds one using mean measured PCB concentrations. It should be noted that the maximum concentration is an actual measured potential exposure concentration; a HQ which exceeds one using the maximum measured PCB concentration is still an indication of potential risk.

A LOAEL is an exposure concentration at which an adverse effect has observed; exposure at this concentration is likely to produce an adverse effect in a receptor. If the HQ is greater than one based on a chronic LOAEL for a particular contaminant, it is an indication that the site levels of that contaminant are likely to produce an adverse effect on survival, reproduction, or growth of the ecological receptor in question. As stated above, the most significant risk is indicated if the HQ exceeds one using mean measured PCB concentrations. In addition, the HQ should be interpreted based on the severity of the effect reported.

4.3.1 Comparisons of Measured Tissue Concentrations to Literature Values

The literature was reviewed to identify fish whole-body, fish egg, and bird egg PCB concentrations that are associated with toxicity. The literature on toxicity-associated tissue levels is summarized in Appendix A. Based on the studies found in the literature,

[2] A study which reports both a NOAEL and LOAEL (a “bounded”effect level) was considered preferable to studies which reported only one effect level. If an unbounded LOAEL is reported, this does not mean that the concentration is the lowest concentration at which an adverse effect may be observed; it is simply the lowest concentration tested in a particular study.

a no observed adverse effect concentration (NOAEC) and a lowest observed adverse effect concentration (LOAEC) for effects associated with PCB concentrations in fish whole bodies, fish eggs, and bird eggs were developed. The mean and maximum PCB concentrations for each tissue matrix were divided by the toxic threshold tissue concentrations (NOAECs and LOAECs) derived from the literature for each tissue matrix, resulting in a HQ. An HQ greater than 1.0 indicates a potential ecological risk.

Due to the differing degrees of confidence in the two sources of fish tissue data, the fish data were treated in two ways. First, the maximum and mean whole-body PCB concentrations from the database used to support the NRDA was used. Since this database contained whole-body PCB concentrations for two upper trophic level fish species (brown trout and walleye), thus representing the upper trophic level measurement endpoint species (lake trout), and since the data collected for the NRDA were collected under rigorous QA/QC procedures, the resulting HQ is associated with a high level of confidence. However, the NRDA database was comprised of only eight composite samples of fish, and maximum concentrations in individual fish may be underestimated. Therefore, a separate evaluation was conducted in which data from the Green Bay Mass Balance Model database was combined with the NRDA database in order to calculate overall maximum and mean PCB concentrations. To do this, only data for upper trophic level fish (brown trout and walleye) from the Mass Balance Model were used, since this was expected to represent whole-body PCB concentrations in the measurement endpoint species (lake trout) better than whole-body concentrations of forage fish, which were also available in the Mass Balance Model database. Since the Mass Balance Model database was comprised of twelve composite whole-body fish samples, this combined data set decreases the uncertainty derived from having only eight data points upon which to base an evaluation, as would have been the case if only the NRDA data were used.

This information obtained from comparing measured fish tissue and bird egg concentrations and estimated fish egg concentrations to literature values contributed to the risk characterization for the following assessment endpoints:

- Pelagic fish reproduction and survival
- Piscivorous bird reproduction and survival

4.3.2 Food Chain Models

Food chain models were used to characterize risk for the following assessment endpoints:

- Piscivorous bird reproduction and survival
- Piscivorous mammal reproduction and survival

The effect level values for dietary toxicity of PCBs were based on published studies, and are summarized in Appendix A. The exposure concentrations were estimated by employing a food chain model for each measurement endpoint (e.g., the mink) associated with an assessment endpoint (e.g., piscivorous mammals). In these food chain models, ingestion rates of PCBs for each receptor species were determined based on measured concentrations of PCBs in water, sediment, and food items collected from the upper Green Bay as well as known or estimated water, sediment, and food ingestion rates and body weights of each receptor species (Appendix B).

For this risk assessment, both maximum and mean contaminant exposure scenarios were modeled for each receptor. To model the maximum contaminant exposure scenario, the maximum water, sediment, and fish PCB concentrations were entered into the food chain

models to estimate a maximum contaminant dose for each receptor species. Likewise, to model the mean exposure scenario, the mean measured PCB concentrations in water, sediment, and fish were entered into the food chain models to estimate a mean contaminant dose for each receptor species.

Sediment and fish tissue PCB concentrations were entered into the models as wet weight concentrations to be compared with the toxicity values derived from the literature, which were also entered into the models on a wet weight basis. In addition, the water concentrations entered into the models were for the sum of the dissolved plus particulate PCBs because this represents a more realistic estimate of exposure via oral ingestion of water.

The fish data from the NRDA and the Mass Balance Model databases were treated in two ways in the food chain models. First, the maximum and mean whole-body PCB concentrations from the NRDA database were calculated and entered separately into the food chain models. Since the NRDA database contained whole-body PCB concentrations for only upper trophic level fish species (brown trout and walleye) rather than forage fish species, an overestimation of the PCB dosage from the ingestion of fish is expected since upper trophic level fish are expected to accumulate greater concentrations of PCBs than forage fish. However, since the NRDA database is the only source of data collected using rigorous QA/QC procedures, it was deemed appropriate to use these data in the food chain models. It should be noted that the resulting HQs may be higher than if PCB concentrations for forage fish were used. The Mass Balance Model data set, on the other hand, comprised data for three forage fish species (alewife, carp, and smelt). Therefore, a mean and maximum PCB concentration for forage fish only were calculated from the Mass Balance Model database and were also entered separately in the food chain models. The use of the Mass Balance Model forage fish data helps to address the uncertainty derived from using upper trophic level fish PCB concentrations from the NRDA database to represent forage fish PCB concentrations in the food chain models.

Uncertainty was also associated with the surface water (birds and mink) and sediment (mink only) PCB concentrations that were entered into the food chain models, since these data were also obtained from the Mass Balance Model database. To address this uncertainty in the food chain models, an HQ was calculated for the ingestion of fish alone as well as for the ingestion of fish, sediment, and water together. As a result, the influence of the sediment and water data on the final HQs could be determined, and the uncertainty derived from using the Mass Balance Model sediment and water data in the food chain models could be qualitatively evaluated.

4.3.3 Nesting Colony Studies

The results from published nesting colony studies were used as a third line of evidence to evaluate the risk to piscivorous birds inhabiting the upper Green Bay. Studies that have examined reproductive injuries in bird colonies in the upper Green Bay were summarized and used to support the conclusions regarding risk to the following assessment endpoint:

- ♦ Piscivorous bird reproduction and survival

5.0 RESULTS OF DATA COMPILATION

5.1 Surface Water PCB Data

Fifty-seven surface water samples were collected in the Upper Green Bay study area in support of

the Green Bay Mass Balance Model (U.S. EPA 1996). Total PCB concentrations in surface water ranged from 0.00028 to 0.00311 micrograms per liter (µg/L), with a mean concentration of 0.001 µg/L (Table 2).

5.2 Sediment PCB Data

Twenty-eight surface sediment samples were collected in the upper Green Bay Study area. Sediment concentrations ranged from 2.4 to 27.07 micrograms per kilogram (µg/kg) wet weight, with a mean PCB concentration of 11.33 µg/kg wet weight (Table 3).

5.3 Fish Whole-Body PCB Data

Fish tissue samples collected for the NRDA data set were composite samples comprised of three to six individual fish. Overall, a total of 10 walleye and 25 brown trout were included in the composite samples. PCB concentrations ranged from 1.17 to 1.98 mg/kg wet weight in brown trout, and 4.61 to 7.26 mg/kg wet weight in walleye (Table 4). Mean and maximum fish tissue concentrations from this data set were 3.23 and 7.26 mg/kg wet weight, respectively. These concentrations were used as the measurement endpoint for Assessment Endpoint #1, and also in food chain models for piscivorous birds and mammals.

PCB concentration ranges measured in fish collected in support of the Mass Balance Model were as follows: 0.11 to 4.20 mg/kg wet weight in forage fish, 1.70 to 3.90 mg/kg wet weight in brown trout, and 0.62 to 5.90 mg/kg wet weight in walleye (Table 5). The mean PCB concentration in forage fish was 1.28 mg/kg wet weight, while mean PCB concentration in upper trophic level fish was 2.98 mg/kg wet weight. Forage fish concentrations were used in food chain models for piscivorous birds and mammals.

To evaluate Assessment Endpoint #1, brown trout and walleye tissue data from both data sets was combined to obtain an overall mean and maximum PCB concentrations in upper trophic level fish of 3.04 and 7.26 mg/kg wet weight, respectively (Table 6).

5.4 Fish Egg PCB Data

Estimated concentrations of PCBs in fish eggs were calculated using an egg to whole-body ratio of 0.209 calculated for lake trout using data presented in Mac et al. (1993). Lake trout whole-body and egg PCB concentrations were reported; the egg concentrations (wet weight) were divided by the whole-body PCB concentrations (wet weight) to calculate the above ratio. Using mean and maximum PCB concentrations from the NRDA data set, a mean and maximum egg PCB concentration of 0.68 and 1.52 mg/kg wet weight was calculated (Table 7). When upper trophic level fish data from the NRDA and Mass Balance data set were combined, the estimated mean egg concentration is 0.64 mg/kg wet weight, and maximum egg concentration is 1.52 mg/kg wet weight.

5.5 Bird Egg PCB Data

Several studies were located which reported PCB concentrations measured in Caspian tern and double-crested cormorant eggs from the upper Green Bay study area (Table 8). The most recent data available was selected for use in this risk assessment. Ewins et al. (1994) reported a mean concentration of 15.8 mg/kg wet weight in Caspian tern eggs collected on Gravelly Island in 1991 (Table 8). Maximum and individual egg concentrations were not reported. Custer et al. (in press) reported mean and maximum PCB concentrations of 10.4 and 20.1 mg/kg wet weight, respectively, in double-crested cormorant eggs collected on Spider Island in 1994 and 1995.

6.0 RISK CHARACTERIZATION (Step 7)

6.1 Assessment Endpoint # 1: Pelagic Fish Reproduction and Survival

6.1.1 Comparisons of Estimated Fish Egg PCB Concentrations to Literature Values

Numerous studies have demonstrated that the early life stages of fish are most sensitive to PCB toxicity, and that PCBs are transferred from maternal tissue to eggs (Ankley et al. 1992, Newsted et al. 1995, Larsson et al. 1993). These studies are summarized in Appendix A. Reported NOAEC and LOAEC concentrations ranged from 0.17 to 3.7 mg/kg wet weight, and 0.31 to 5.1 mg/kg wet weight, respectively. Based on study characteristics (e.g., study design, presence of contaminants other than PCBs), a reported LOAEC of 1.6 mg/kg wet weight (Hendricks et al. 1981) and an estimated NOAEC of 0.16 mg/kg wet weight were selected as the most appropriate TRVs for this risk assessment.

Using data from the NRDA database, estimated mean PCB concentrations in eggs were 0.68 mg/kg, wet weight and maximum egg PCB concentrations were 1.52 mg/kg, wet weight. When data from the NRDA database and the Green Bay Mass Balance Model were combined, mean and maximum egg PCB concentrations were 0.64 and 1.52 mg/kg wet weight, respectively. All HQs calculated using the NOAEC exceeded 1.0, and ranged from 4.0 to 9.5 (Table 9). None of the HQs calculated using the LOAEC exceeded 1.0.

Results of risk calculations for fish egg concentrations indicate potential risk to pelagic fish reproduction and survival in the upper Green Bay.

6.1.2 Comparisons of Measured Fish Whole-Body Concentrations to Literature Values

Numerous studies have been conducted with fish in which adverse effects on reproductive endpoints have been observed, and whole-body concentrations of PCBs in adults have been measured. These studies are summarized in Appendix A. Reported NOAEC and LOAEC concentrations ranged from 1.6 to 11.6 mg/kg wet weight, and 9.3 to 429 mg/kg wet weight. No effect concentrations reported in studies in which growth was the measured endpoint ranged from 32 to 645 mg/kg wet weight.

An alternative way to determine whole-body concentrations at which adverse effects would be expected is to estimate a whole-body concentration based on an egg concentration that is associated with adverse effects. This method was derived based on the fact that whole-body concentrations are often available, while fish egg concentrations are not. Early life-stages are most sensitive to adverse effects of PCBs, therefore it is important to identify maternal whole-body concentrations which result in critical egg/fry PCB concentrations. Mac et al. (1993) reported lake trout whole-body and egg concentrations of PCBs; when the egg PCB concentrations (wet weight) were divided by the whole body PCB concentrations (wet weight), a mean ratio of 0.209 was calculated. Using this ratio, an expected lake trout whole-body concentration can be calculated based on a lake trout egg concentration. When the egg LOAEC concentration of 1.6 mg/kg wet weight, cited above, is divided by 0.209, a whole-body concentration that would be expected to elicit adverse effects of 7.7 mg/kg wet weight was calculated. Since this method provided the lowest LOAEC for whole-body fish PCB concentrations, a LOAEC of 7.7 mg/kg wet weight, and a calculated NOAEC of 0.77 mg/kg wet weight were used to evaluate the effects of PCBs on fish survival and reproduction in the upper Green Bay.

Because the LOAEC selected for fish egg concentrations was used to derive a whole-body concentration that would be expected to elicit adverse effects, these two lines of evidence are functionally the same. However, whole-body PCB concentrations are easier to measure than egg concentrations (sample collection is not seasonally limited); therefore use of this method to identify a common measurement (whole-body PCB concentration) that targets the most sensitive life stage was determined to be valid.

Using data from the NRDA database, mean whole-body fish PCB concentrations were 3.23 mg/kg wet weight and maximum whole-body fish PCB concentrations were 7.26 mg/kg wet weight. When data from the NRDA database and the Green Bay Mass Balance Model were combined, mean and maximum whole-body PCB concentrations in upper trophic level fish were 3.04 and 7.26 mg/kg wet weight, respectively. All HQs calculated using the NOAEC exceeded 1.0, and ranged from 3.9 to 9.4 (Table 9). None of the HQs calculated using the LOAEC exceeded 1.0.

Because HQs calculated for fish tissue concentration using the NOAEC as the effect level exceed 1.0, pelagic fish reproduction and survival in the upper Green Bay is potentially at risk from PCB exposure.

6.2 Assessment Endpoint #2: Piscivorous Bird Reproduction and Survival

6.2.1 Comparisons of Measured Bird Egg Concentrations to Literature Values

Field and laboratory studies have been published which correlate concentrations of PCBs in bird eggs with adverse effects on survival, growth, or reproduction. Observed effects include reduction in hatching success, eggshell production and female fertility (Scott 1977, Platonow and Reinhart 1973, McLane and Hughes 1980, Hoffman et al. 1993). These studies are summarized in Appendix A. Reported NOAEC and LOAEC concentrations of PCBs in bird eggs ranged from 0.36 to 39 mg/kg wet weight, and 1.5 to 105 mg/kg wet weight, respectively. The lowest exposure concentrations at which adverse effects were observed were reported in studies conducted with chickens. Numerous studies have documented the greater sensitivity of chickens to TCDD-like toxicity compared with other bird species. Other species tested include pheasants, mallards, goldeneyes, herring gulls, black-headed gulls, common tern and kestrels (Brunstrom 1988, Brunstrom and Reutergardh 1986, Hoffman et al. 1998); all species tested to date have been considerably less sensitive than chickens (Hoffman et al. 1998). A possible explanation for this difference in sensitivity is a difference in concentration of the Ah receptor or its binding affinity for TCDD. This receptor is present in the early stages of chick embryo development but was not found in turkey embryos (Brunstrom and Lund 1988). Because of their greater sensitivity, studies in which chickens were the test species were not selected for derivation of the NOAEC and LOAEC in this risk assessment. The NOAEC of 4.7 mg/kg wet weight and LOAEC of 7.6 mg/kg wet weight reported by Hoffman et al. (1993) for common terns were selected for use in this risk assessment; the adverse effect observed was decreased hatching success.

Measured mean PCB concentrations in Caspian tern and double-crested cormorant eggs were 15.8 mg/kg, wet weight, and 10.4 mg/kg wet weight, respectively (Table 10). All HQs calculated for mean PCB concentrations in bird eggs and NOAEC or LOAEC values exceeded 1.0. The maximum concentration measured in cormorant eggs was 20.1 mg/kg wet weight (Custer et al. in press). Hazard quotients calculated using the NOAEC and LOAEC were 4.3 and 2.6, respectively. No maximum concentration was reported by Ewins et al. (1994) for tern eggs, however hazard concentrations calculated using the mean and both effect levels exceeded 1.0, indicating potential risk. Use of the maximum

concentration in risk calculations would only increase the magnitude of the calculated HQ. All HQs calculated for bird egg PCB concentrations exceeded 1.0, indicating that piscivorous bird species utilizing the upper Green Bay are at risk.

Ludwig et al. (1996) reported a NOAEC of 0.8 mg/kg; the adverse effect measured in this study was deformity rate. This concentration was also evaluated in this risk assessment for comparative purposes, however it should be noted that this is an unbounded NOAEC and it was not selected as the sole TRV for this reason. All HQs calculated using this NOAEC exceeded 1.0, and ranged from 13 (mean concentration in double-crested cormorant eggs) to 25.1 (maximum concentration in double-crested cormorant eggs; Table 10). Use of this NOAEC does not change the conclusions of this risk assessment, namely that piscivorous birds utilizing the Upper Green Bay are at risk based on measured egg PCB concentrations.

6.2.2 Food Chain Models for Piscivorous Birds

A literature search was conducted to evaluate dietary toxicity of PCBs to bird species. The results of the literature search are presented in Appendix A. No studies were found in which dietary toxicity of PCBs to either of the selected receptor species (Caspian tern and double-crested cormorant) was tested. Reported NOAEL and LOAEL concentrations for other avian species ranged from 0.0158 to 2.0 mg/kg BW/day, and 0.0414 to 275 mg/kg BW/day, respectively. As before, studies in which chickens were the test species were not selected for derivation of the NOAEL and LOAEL in this risk assessment due to the documented greater sensitivity of this species to adverse effects from PCB exposure. A TRV was selected for this risk assessment based on the ecological significance of the observed adverse effects (reproductive success and behavior), and study design where PCBs were the only dietary contaminant present. A LOAEL of 1.12 mg/kg BW/day reported in studies using ring doves (Peakall and Peakall 1973, Peakall et al. 1972) and mourning doves (Tori and Peterle 1983) was selected as the TRV for this risk assessment. A NOAEL of 0.112 mg/kg BW/day was calculated from this LOAEL using an accepted conversion factor of 10 (Dourson and Stara 1983).

Dietary exposure concentrations for the two piscivorous bird receptor species were calculated using life history parameters summarized in Appendix B. For each species, the following exposure scenarios were evaluated:

- Ingestion of fish with mean and maximum PCB concentrations from the NRDA database (upper trophic level species)
- Ingestion of fish with mean and maximum PCB concentrations from the Green Bay Mass Balance Model database (forage species only)
- Ingestion of fish and ingestion of surface water (water data from the Green Bay Mass Balance Model)

Hazard quotient calculations were done using the NOAEL and LOAEL as the effect level for each of the above scenarios. Results of the food chain model calculations are presented in Table 11 (Caspian Tern) and Table 12 (Double-crested cormorant).

6.2.2.1 Caspian Tern

Hazard quotients calculated using the NOAEL and mean and maximum PCB concentrations in fish from the NRDA database, and maximum concentrations in fish from the Mass Balance Model database exceeded 1.0 (2.0, 4.6 and 2.6, respectively). None of the HQs obtained utilizing the LOAEL in the calculation exceeded 1.0. Adding ingestion of surface water to the exposure calculations had no impact on the results of the HQ calculations (HQs of 2.0, 4.6 and 2.6 for

mean and maximum PCB concentration from the NRDA data set and maximum concentration from the Mass Balance data set), indicating food ingestion is the primary source of contaminant exposure for this species.

Results of the HQ calculations indicate Caspian terns utilizing the upper Green Bay as a foraging area may potentially be at risk from dietary exposure to PCBs.

6.2.2.2 Double-Crested Cormorant

All HQs calculated using the NOAEL as the effect level exceeded 1.0 for this species (Table 12). Ingestion of fish with mean and maximum PCB concentrations from the NRDA database resulted in HQs of 7.2 and 16.2. Calculations using mean and maximum fish PCB concentrations from the Mass Balance Model resulted in HQs of 2.9 and 9.4, respectively. An HQ of 1.6 was calculated using the LOAEL as the effect level and maximum fish concentrations from the NRDA database. None of the other calculations done using the LOAEL resulted in an HQ which exceeded 1.0. As with the Caspian tern, including water ingestion in the exposure scenario had no impact on calculated HQs, indicating that food ingestion is the major exposure route for this species.

Because some HQs calculated for this species exceeded 1.0 when either effect level was evaluated, a food chain exposure using the double-crested cormorant model indicates piscivorous birds utilizing the upper Green Bay are at risk from PCB exposure.

6.2.3 Nesting Colony Studies

6.2.3.1 Caspian Tern

Ludwig and Ludwig (undated report) performed a field study during the 1986 nesting season and looked at rates of deformities and reproductive success in Caspian terns nesting on Gravelly and Gull Islands in upper Green Bay as well as islands in Lake Michigan, Lake Superior, and Lake Huron, the latter of which served as a reference site. The authors found no evidence of developmental defects in Caspian terns nesting in the upper Green Bay. However, they did observe the lowest hatching rate of all the study areas to be in Saginaw Bay and the upper Green Bay, with hatching success on Gravelly and Gull Islands measured to be 72 percent and 71 percent, respectively, compared with a range of 81 to 84 percent in the remaining colonies.

A similar study (Kurita and Ludwig 1988) was performed in 1988 in which Caspian tern eggs were collected from colonies nesting on Gravelly and Gull Islands in the upper Green Bay as well as in Lake Huron, Lake Superior, and Lake Michigan. Eggs were examined for viability and developmental deformities and grouped into four categories: live-normal, dead-normal, infertile, and deformed. The deformed category included both dead- and live-deformed. Unclassifiable and rotten eggs were classified as dead-normal. In the upper Green Bay, 13 Caspian tern eggs were classified as live-normal, 3 as infertile, and 2 as deformed. Organochlorine residues were examined in conjunction with these results, but unlike the cormorants, no trends could be established between PCB residues and rates of deformities in Caspian terns.

In 1990, Mora et al. (1993) examined productivity and colony site tenacity in

relation to PCB concentrations in blood samples collected from Caspian terns nesting in the Great Lakes, including Gravelly and Gull Islands in upper Green Bay. They found that productivity, as measured by the number of eggs laid, hatching success, and fledging success, was not significantly different between the upper Green Bay and the other colonies, even though PCB concentrations measured in the blood samples were greater in Caspian terns collected in upper Green Bay and Saginaw Bay compared with the other colonies. However, the authors report that the hatching success rates observed in this study, which ranged from 74 to 82 percent for all of the colonies studied, were less than the hatching success of Caspian tern colonies nesting in Texas where 85 percent success has been observed and in Finland where 85 to 95 percent success has been noted. Colony site tenacity was exceptionally low in the upper Green Bay colonies (56.5 percent) compared with the other colonies studied (81.2 to 100 percent). The authors explain that Caspian terns are less likely to return to their original breeding area if they experience poor reproduction during the previous year. When natal site tenacity is examined, a correlation is observed with PCB concentrations in blood samples by region, where natal site tenacity decreases with increasing PCB concentrations. However, this correlation is based on a small number of data points. Therefore, more data is needed to confirm this relationship.

Ludwig et al. (1996) summarized a variety of studies conducted from 1987 to 1991, in which field observations of Caspian tern egg death rates and deformity rates were made and either total PCBs or toxicity equivalents (TEQs) were measured in eggs for colonies in the Great Lakes, including Green Bay. The Green Bay colonies had the highest deformity and egg death rates of all the Great Lakes colonies studied except for Saginaw Bay, another region that is known to contain high levels of contamination. However, data specific to the upper Green Bay could not be deciphered from the data presented. Nonetheless, the authors found a significant correlation between TEQs and deformity rates in hatched tern chicks and dead eggs as well as egg death rates, although only egg death rates exhibited a strong correlation ($r^2 = 0.68$). Poor correlations were observed between total PCBs and the observed adverse effects.

Ewins et al. (1994) present the results of a 1991 study on Caspian terns nesting in colonies in the Great Lakes, including two islands (Gravelly and Gull Islands) in the upper Green Bay. Although observations were performed on both islands, eggs were only taken from Gravelly Island. Reproductive output was measured by determining the number of active nests per colony, and by monitoring the nests for numbers of eggs, hatching success, and number of young fledged per nest. Average rates of population change were determined by comparing nest counts for the 1991 study with a count that was conducted in 1980. The results indicated that even though the concentrations of PCBs and dichlorodiphenylethylene (DDE) in the eggs were highest on Gravelly Island and Saginaw Bay, there was no evidence of an overall adverse reproductive effect on Caspian terns in the upper Green Bay, since the number of young per pair was well above the minimum value of 0.6 established by Ludwig (1965) to maintain population stability. Furthermore, a dramatic increase in the number of active Caspian tern nests on Gravelly and Gull Islands in the upper Green Bay was observed from 1980 to 1991. The authors caution in basing definitive conclusions on this study in light of the results of the study by Mora et al. (1993) that indicate that PCBs may be affecting certain reproductive parameters such as natal region fidelity (tendency to return to their original breeding area) in the

upper Green Bay.

The results of the above studies are not conclusive that Caspian terns are at risk from PCBs in the upper Green Bay. The data presented suggest that PCBs are not associated with adverse effects on endpoints such as hatching success and deformities, but one study found a strong negative correlation between Caspian tern site tenacity and PCBs. This indicates that some subtle reproductive effects may be manifesting themselves in the upper Green Bay as a result of exposure to PCB contamination.

6.2.3.2 Double-Crested Cormorant

Ludwig and Ludwig (undated report) performed a field study during the 1986 nesting season and looked at rates of deformities and reproductive success in double-crested cormorants nesting on islands in upper Green Bay (Gravelly and Little Gull Islands) as well as in Lake Michigan, Lake Superior, and Lake Huron; Lake Huron was used as the reference site. They found that the rates of deformities were higher in the upper Green Bay compared with all other sites. Nine cormorants were observed with deformities, including crossed bill, chick edema, unabsorbed yolk sac, dwarfism, and an opaque covering over the eye. It is unclear whether the last deformity is chemically-induced, but the other deformities are similar to those observed in the laboratory as a result of exposure to PCBs (Ludwig et al. 1996). In addition, the lowest hatching rates were also observed in the upper Green Bay, with 63 percent hatchability in upper Green Bay versus 74 percent observed in the reference area (Lake Huron).

A similar study (Kurita and Ludwig 1988) was performed in 1988 in which double-crested cormorant eggs were collected from colonies nesting on Little Gull Island in the upper Green Bay as well as on islands in Lake Huron, Lake Superior, Lake Michigan. Eggs were examined for viability and developmental deformities and grouped into four categories: live-normal, dead-normal, infertile, and deformed. The deformed category included both dead- and live-deformed. Unclassifiable and rotten eggs were classified as dead-normal. In the upper Green Bay, a high rate of reproductive abnormalities was observed. Specifically, 18 cormorant eggs were classified as live-normal, 15 as infertile, and 8 as deformed. Organochlorine residues were examined in conjunction with these results, and it was found that total PCBs were correlated with the numbers of live deformities in cormorant chicks, while rates of dead-normal, dead-deformed, and infertile eggs were better correlated with coplanar PCBs and other chlorinated hydrocarbons.

Fox et al. (1991) performed a review of all studies conducted between 1979 and 1987 in which double-crested cormorants were examined for bill deformities in colonies in the Great Lakes, including Green Bay, as well as four reference areas. They found that the prevalence of chicks with bill defects in Green Bay was markedly greater than all other regions during this time interval. These differences were statistically significant ($p < 0.05$) between Green Bay and the North Channel, Alpena, and Lake Erie, and the difference approached significance ($p < 0.1$) for all other regions. The study also determined that the probability of observing a cormorant chick in Green Bay with a malformed bill was 10 to 32 times greater than for colonies in the reference areas. The incidence of bill defects was significantly greater in Green Bay compared with all other regions studied except for Lake Ontario. Bill defects were observed in

73 percent of the colonies observed in Green Bay, as compared with only 6 percent of the colonies observed in the reference areas. The authors suggest a chemical etiology for the observed bill defects, since an investigation into the cause of similar bill defects in Forster's terns indicated that the defects were associated with increased liver-to-body mass ratios and elevated aryl hydrocarbon hydroxylase (AHH) activity. Furthermore, the authors stated that all three of the more toxic non-ortho PCB congeners have been isolated from tissues of cormorant chicks with crossed bills collected from Green Bay. Two of these congeners are known to cause craniofacial abnormalities in laboratory animals. Although the data presented in this study do not allow one to distinguish between the upper and lower Green Bay colonies, the data presented clearly demonstrate that craniofacial abnormalities were high in double-crested cormorants nesting in Green Bay as a whole between 1979 and 1987 and that these defects may have been caused by exposure to polychlorinated aromatic hydrocarbons such as PCBs.

Tillitt et al. (1992) examined reproductive success of double-crested cormorants from 1986 to 1988 in colonies in and around the Great Lakes. They found that egg mortality was significantly greater in all of the Great Lakes nesting colonies, including the upper Green Bay colonies (Little Gull, Snake, and Gravelly Islands), where egg mortality ranged from 32 to 39 percent. At the reference area (Lake Winnipegosis), egg mortality was only 8 percent. Total PCB concentrations in eggs ranged from 0.05 and 14.8 µg/g wet weight. The authors found a significant correlation between total PCB concentrations in eggs and egg mortality ($p=0.045$). However, the coefficient of determination (r^2) was only 0.319, indicating that much of the variance in egg mortality was not explained by this general linear model. A significant correlation was also observed between egg mortality and the H4IIE rat hepatoma bioassay-derived 2,3,7,8-tetrachloro-*p*-dibenzodioxin equivalents (TCDD-EQ) concentrations ($p \leq 0.0003$, $r^2 = 0.703$). The eggs were analyzed for total PCBs, polychlorinated dibenzo-*p*-dioxins (PCDD), and polychlorinated dibenzofurans (PCDF)-type planar halogenated hydrocarbons (PHHs), and only PCBs were detected. This indicates that PCBs are the main contaminant associated with the observed egg mortality in double-crested cormorants in the Great Lakes colonies, including upper Green Bay.

Ludwig et al. (1996) summarized a variety of studies conducted from 1986 to 1991, in which field observations of double-crested cormorant egg death rates and deformity rates were observed and either total PCBs or TCDD-EQs were measured in eggs for colonies in the Great Lakes, including the upper Green Bay. Deformity rates were higher in all Great Lakes colonies than at a reference colony. Of all the Great Lakes colonies studied, the upper Green Bay had the highest rate of egg deformities (6.14 per thousand for upper Green Bay versus a range of 0.69 to 3.6 per thousand for the other Great Lakes colonies). Similarly, the egg death rate for Green Bay was higher than any other colony studied, although data specific to the upper Green Bay could not be deciphered from the data presented for Green Bay. PCB concentrations ranged from 0.8 mg/kg wet weight at the reference colony to 7.3 mg/kg in eggs collected from Green Bay. The authors found a significant correlation between hatching and deformity rates and both PCBs and TCDD-EQs, indicating that PCBs are playing a large role in the cormorant egg death and deformity rates observed in the upper Green Bay.

The weight of evidence based on the results presented in the studies summarized

above indicate that double-crested cormorants are experiencing adverse reproductive effects in the upper Green Bay. Deformities such as crossed bills, edema, unabsorbed yolk sac, and dwarfism as well as embryo mortality are characteristic of abnormalities observed as a result of exposure to polychlorinated aromatic hydrocarbons such as PCBs. This indicates that double-crested cormorants are at risk from PCBs in the upper Green Bay.

6.3 Assessment Endpoint #3: Piscivorous Mammal Reproduction and Survival

6.3.1 Food Chain Model

A literature search was conducted to evaluate dietary toxicity of PCBs to mammals, and results are presented in Appendix A. Numerous studies were located in which mink were the test species. Because mink are the selected receptor species for this risk assessment, and have been shown to be particularly sensitive to PCBs, these studies were the only mammal studies reviewed to derive the TRV. Reported LOAEL concentrations ranged from 0.055 to 1.1 mg/kg BW/day. The reported effect observed at the 0.055 mg/kgBW/day concentration was decreased kit growth. Reproductive effects (kit survival) were observed at exposure concentrations of 0.5 and 0.72 mg PCB/kg diet (Restum et al. 1998 and Heaton et al. 1995, respectively). Statistically, these two concentrations are effectively the same[3]. Food consumption was measured in the Heaton et al (1995) study; the reported exposure concentrations of 0.134 and 0.004 mg/kgBW/day was selected as the LOAEL and NOAEL to be utilized in this risk assessment.

The exposure scenarios evaluated for mink were the same as those evaluated for piscivorous birds, except that incidental sediment ingestion was added to the fish and water ingestion scenario. All HQs calculated for mink exceeded 1.0, and ranged from 2.1 (LOAEL as the effect level and mean PCB concentrations from the Mass Balance data set) to 397.8 (NOAEL as the effect level and maximum fish PCB concentrations from the NRDA data set; Table 13). Adding sediment and surface water ingestion to the exposure scenario had almost no effect on calculated HQs, indicating food ingestion is the primary exposure route for this species.

Exposure of mink is limited to feeding along the shoreline of the bay and along tributaries; mink may obtain a significant portion of their diet from tributaries. Although PCB concentrations from bay fish were used to model mink exposure, limited data are available for PCB concentrations in fish collected from tributaries to Green Bay (WI DNR 1999, Appendix C) . Whole-body PCB concentrations in walleye collected from the Peshtigo River ranged from 3.25 to 7.3 mg/kg, and from 0.36 to 13.0 mg/kg in walleye collected from the Menominee River. The range of whole-body PCB concentrations in walleye collected from the upper Green Bay (range 0.62 to 7.26 mg/kg) are comparable to those measured in tributary fish, and are a reasonable estimate of mink exposure levels.

The calculated HQs for this species indicates piscivorous mammals utilizing the upper Green Bay area are at risk from exposure to measured PCB concentrations in fish.

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[3] Based on the reported mean and standard deviation for total PCB concentration in diets used in the two studies.

7.0 UNCERTAINTY ANALYSIS

7.1 General Uncertainty Analysis

There are factors inherent in the risk assessment process that contribute uncertainty and must be considered when interpreting results. Major sources of uncertainty arise from natural variability in biological systems, the introduction of error in the risk assessment process, and the presence of data gaps.

Natural variability is an inherent characteristic of ecological receptors, their stressors, and their combined behavior in the environment. Biotic and abiotic parameters in these systems may vary to such a degree that the exposure of similar ecological receptors within the same system may differ temporally and spatially. Factors that contribute to temporal and spatial variability may be differences in an individual organism's behavior (within the same species), changes in the weather or ambient temperature, unanticipated interference from other stressors, differences between microenvironments, and numerous other factors.

Uncertainty associated with natural variability also arises from the use of literature toxicity values in which a study has examined a single species/single contaminant system under controlled conditions. If conducted in a laboratory, these studies do not take into account the effects of the environmental factors and other stressors that are present in natural systems. These factors may have synergistic, antagonistic, or neutral effects upon the receptor-contaminant interaction.

Point estimates of exposure such as NOAELs, LOAELs, LD_{50}s, and mathematical means that are presented in the literature also have inherent variability, which is incorporated into the risk assessment. Additionally, because these values are statistically determined, they do not represent absolute thresholds; they are reflective of the experimental design. A reported LOAEL may not represent the lowest toxicity threshold for a species simply because lower concentrations were not tested in a study.

In addition, uncertainty associated with variability is introduced from the use of literature values for soil, sediment, water, and food ingestion rates, dietary compositions, and body weights. These values reported in the literature are from studies that may have been conducted at a time of year or in a location that does not necessarily give an accurate representation of the life histories of the receptor species in the upper Green Bay.

Error may be introduced into the risk assessment through the use of invalid assumptions in the conceptual model. Conservative assumptions were made in light of the uncertainty associated with the risk assessment process (e.g., natural variability). Consistent conservative assumptions were used to minimize the possibility of concluding that risk is not present when a threat actually does exist (i.e., the elimination of false negatives). While there is uncertainty associated with each conservative assumption used, this consistent selection process assures that the uncertainty associated with this type of error will err on the side of a protective outcome.

This risk assessment did not examine the contribution of dermal absorption or inhalation exposure as part of the exposure pathway. In contrast to the use of conservative assumptions, the error introduced into this risk assessment by the omission of these routes of exposure may err on the side of a less protective outcome. The relative contribution of this error to alter the outcome of the risk assessment is unknown at this time.

Methodological problems in the literature reviewed for obtaining life history and toxicity information also introduce uncertainty into a risk assessment. Attempts were made to avoid using literature that was questionable. The process used to select appropriate studies on which to base

TRV derivation and life history parameter selection is described in Appendices A and B. However, if limited sources of information existed, potential error due to questionable study design was incorporated into the risk assessment if these data were used.

Data gaps were defined here as the incompleteness of data or information upon which the risk assessment was based. Specifically, these may be an incomplete contaminant data set, missing pieces of life history information, the absence of toxicity-based literature for the receptor of concern, or unknown or questionable QA/QC procedures.

Life history information and literature values for the toxicity of the contaminants of concern were not always available for all of the receptor species. By using closely related species, it was possible to make risk estimates. In reality, however, the information may vary substantially among species, thereby introducing another source of uncertainty.

In cases where a toxicity value has been converted by a factor of 10, the uncertainty associated with the absence of a directly relevant literature value was compounded by the uncertainty associated with a subjective mathematical adjustment.

7.2 Site-Specific Uncertainty Analysis

7.2.1 Selection of Contaminants of Potential Concern

The contaminant of concern evaluated in this risk assessment was selected based on the risk assessment conducted for the Lower Fox River/lower Green Bay (ThermoRetec Consulting Corporation). Of the eight COPCs retained for the above assessment, only PCBs were selected as a COPC for this risk assessment. It is well documented that PCBs are the most widespread contaminant in the Fox River/Green Bay system (ThermoRetec Consulting Corporation 1998). In addition, the above cited risk assessment concluded that risks to ecological receptors from PCB exposure were 10 to 1,000 times greater than predicted risk from the other seven COPCs (ThermoRetec Consulting Corporation). It should be recognized that other contaminants could potentially pose a risk to ecological receptors which utilize the Upper Green Bay. However, to focus this risk assessment, it was assumed that risks from exposure to other contaminants were sufficiently evaluated in the risk assessment conducted for the Lower Fox River/lower Green Bay.

Many of the toxic effects of PCBs are produced by coplanar PCB congeners that have a structure similar to TCDD. Dioxin-like toxic effects include edema, deformities, and early life stage mortality (Safe 1994). One method often used to evaluate toxicity of PCBs is the TCDD toxicity equivalence approach, where the toxic potency of each PCB congener is expressed relative to the potency of TCDD. A reason for utilizing this method is to incorporate the data available for toxicity of TCDD into the data reviewed for TRV derivation. Numerous studies evaluating PCB toxicity to the selected receptors were located in our literature search. Some uncertainty may result from not extending the literature search to include TCDD toxicity, however appropriate TRVs were located for all receptors based on results of the search which was conducted. An underlying assumption of the TEF approach is that toxicity of PCBs is solely related to their TCDD-like toxicity. Theoretically, any NOAEL for PCBs should incorporate dioxin-like toxicity, therefore this method was not utilized in this risk assessment.

7.2.2 Conceptual Model Limitations

Components of the conceptual model which potentially introduce uncertainty into this risk

assessment include transport and fate of PCBs, selected assessment endpoints and receptor species, and identification of complete exposure pathways.

Transport and fate of PCBs was modeled based on contaminant and ecosystem characteristics. Studies have shown that PCBs discharged into aquatic systems rapidly sorb to sediment (Kalmaz and Kalmaz 1979); movement of PCBs in aquatic systems depends mainly on movement of the associated sediment (Connell and Miller 1984). The lower Green Bay is the primary depositional zone for Fox River PCBs, however several studies conducted within Green Bay have documented sediment transport from the lower to upper Bay (Eadie et al. 1991, Manchester-Neesvig et al. 1996, Hawley and Niester 1993). In addition, fish and birds collected from the upper Green Bay have accumulated elevated concentrations of PCBs in their tissues (U.S. EPA 1996; Hagler Bailly Services, Inc. 1997; Stratus Consulting, Inc. 1999a). Although the above studies indicate transport of contaminated sediment from the lower to the upper Green Bay, and transfer of PCBs from sediment to ecological receptors, the magnitude of both transfer processes is uncertain.

An additional source of uncertainty is that other potential sources of PCBs are not considered in this risk assessment. Atmospheric deposition has been identified as an important source of PCBs to Lake Superior (Eisenreich et al. 1981) and southern Lake Michigan (Murphy et al. 1981). However, Sweet et al. (1991) estimated that atmospheric deposition of PCBs accounts for less than 10 percent of the total input to Green Bay. The potential contribution via influx from Lake Michigan is unknown. DePinto et al. (1994) identified the Fox River as the major source of PCBs to Green Bay; they estimated the Fox River contributed 92 percent of the PCB loading to the bay in 1989. To focus this risk assessment, the assumption was made that the primary source of PCBs to Green Bay was the Fox River.

The assessment endpoints selected for this risk assessment are a subset of those evaluated in the risk assessment conducted for the Lower Fox River/lower Green Bay. The only contaminant evaluated in this risk assessment was PCBs; adverse effects from exposure to PCBs are related to bioaccumulation rather than direct toxicity. Therefore, the assessment endpoints selected for this risk assessment focus on bioaccumulation of PCBs and upper trophic level receptors. By evaluating and protecting these assessment endpoints which are most sensitive to potential impacts from exposure to site-related contaminants, the upper bay ecosystem as a whole should also be protected.

Receptor species were selected for this risk assessment based on the complete exposure pathways identified in the conceptual model. The selected receptors act as surrogates for other species which are similar in terms of feeding habits and habitat use, and should be representative of potential risk to other species within the system. Mink and lake trout were selected as receptors based on their sensitivity to PCB effects. Numerous studies have documented the reproductive toxicity of dietary PCBs to mink at low exposure concentrations (Restum et al. 1998, Den Boer 1984, Heaton et al. 1995, Platanow and Karstad 1973). Among fish species studied to date, lake trout have been found to be most sensitive to PCB-caused fry mortality (Walker et al. 1991). In addition, lake trout females from Lake Michigan produce eggs which are deficient in thiamine; some studies have shown that an interaction exists between thiamine and dioxin-like embryo toxicity (Wright et al. 1998, Fisher et al. 1996, Wright and Tillit 1998). Selection of the most sensitive receptors (mink and lake trout) should adequately protect less sensitive species. Bird receptor species were selected based on potential sensitivity (e.g., observed deformities in field studies), complete exposure pathways, and to be consistent with

receptor species selected for the Lower Fox River/lower Green Bay risk assessment. Although there is uncertainty associated with limiting the number of species evaluated, the primary exposure pathway identified (dietary exposure) was sufficiently evaluated in this risk assessment by selecting high trophic level species as receptors.

Some exposure pathways not evaluated in this risk assessment include ingestion of plankton and exposure of dabbling ducks. No site-specific data on PCB concentrations in plankton, aquatic plants, or benthic organisms was available. The receptor species evaluated in this risk assessment feed at higher trophic levels than receptors within pathways not evaluated, therefore PCB exposure of selected receptors should be higher than for herbivorous or planktivorous species. Protection of the selected receptor species should be protective of organisms with lower exposure levels.

7.2.3 Estimates of Exposure Concentration

Uncertainty can be introduced into the risk assessment process by low quality, limited, or missing site-specific data. As discussed in Section 4.1, data utilized in this risk assessment was obtained from three sources. The data set developed for the NRDA was the most recent, and was developed under the most rigorous QA/QC procedures. Data collected in support of the Green Bay Mass Balance Model was older and has not been validated using strict QA/QC procedures. The QA/QC procedures used to validate bird data from published studies are unknown, however all studies cited have been peer-reviewed.

Fish tissue data from both the NRDA and Mass Balance data sets were used for this risk assessment. Although the confidence level in the quality of data from the NRDA model is high, no forage fish were collected. Piscivorous birds and mammals are not likely to consume fish the size of the upper trophic level fish collected, and PCB concentrations tend to increase with increasing fish size. Therefore, use of upper trophic level fish data to estimate dietary exposure of piscivorous birds and mammals may overestimate exposure concentrations. Therefore, forage fish data from the Mass Balance data set was also utilized for this risk assessment, although these data have not been validated at this time. Separate HQ calculations were done for each data source, so that the uncertainty associated with the different data sets could be evaluated.

Another source of uncertainty associated with use of both the NRDA and Mass Balance data sets is that composite samples were analyzed; each sample was comprised of three to six fish, and five fish each, respectively. Maximum PCB concentrations measured may underestimate maximum PCB concentrations for individual fish.

Fish species analyzed for both the NRDA and Mass Balance data set were walleye and brown trout. The selected receptor species, lake trout, tend to accumulate the highest concentrations of PCBs found in open-water fish of the Great Lakes (Mac and Schwartz 1992). Species-specific traits that contribute to this are:

- Lake trout possess a large amount of body fat (average of 12 percent);
- They have a long life span (8 to 10 years), and are exposed to PCBs for a longer period of time than many fish species;
- They grow slowly, leading to a higher PCB body burden (Jensen et al. 1982);
- Alewife, one of their main prey species, contain significant amounts of PCBs (St. Amant et al. 1984).

Because tissue data utilized in this risk assessment are from walleye and brown trout, actual tissue concentrations found in lake trout may be underestimated, therefore potential risk for this receptor species may be underestimated.

Fish collected within Green Bay were used to estimate dietary exposure of mink to PCBs. It should be noted that exposure of mink will be limited to feeding along the shoreline of the bay and along tributaries. Use of fish concentrations from the bay may overestimate fish concentrations in tributaries to the bay. However, comparison of bay fish data (Table 6) with limited data available on PCB concentrations in walleye collected from tributaries to Green Bay (Appendix C) indicate bay fish concentrations may be a reasonable estimate of tributary fish concentrations.

All sediment and water data were obtained from the Mass Balance data set. As stated above, the quality of these data is unknown, as it has not been validated according to strict QA/QC procedures. In addition, only one water sample collected in the Upper Green Bay was located. A separate exposure scenario was evaluated which incorporated sediment and water ingestion into the food chain model for each receptor, so that the uncertainty associated with use of these data could be evaluated. Sediment and water ingestion had no impact on calculated HQs for any receptor (Tables 11, 12, and 13). Therefore the uncertainty associated with use of one data point was not significant within the risk calculations performed for this risk assessment.

The bird egg and tissue data used in this risk assessment was obtained from studies conducted in the Upper Green Bay which were published in peer-reviewed literature. The QA/QC procedures used to evaluate these data and the associated uncertainty are unknown.

A final limitation of the data utilized in this risk assessment is that the most recent samples were collected in 1996. It is recognized that the Upper Green Bay is not a static system, therefore use of old data to characterize present conditions is another source of uncertainty. However, it is known that many of the primary sources of PCBs to this system have been eliminated; the principal current source of PCBs to ecological receptors is a secondary source, the sediment. Several long-term studies have been conducted within this system. The Canadian Wildlife Service has collected herring gull eggs from Big Sister Island almost every year since 1972 (Bishop et al. 1992, Pettit et al. 1994, Pekarik et al. 1998, Hughes et al. 1998). This data set is the most complete data set available to evaluate temporal trends in PCB exposure of birds that utilize the Green Bay system. The highest input of PCBs to the Green Bay system occurred in the early 1970s. After primary sources (e.g., discharges from paper companies related to use of PCB emulsion) were eliminated, PCB concentrations in herring gull eggs declined rapidly until 1982 (mean concentration approximately 142 and 62 mg/kg, wet weight, in 1971and 1982, respectively). Since 1983 the decline has reached a plateau (mean concentration approximately 27 and 15 mg/kg, wet weight, in 1983and 1996, respectively), although there is an almost significant negative trend ($r = 0.5$, $P = 0.07$; Stratus Consulting Inc. 1999d). A temporal PCB pattern similar to that seen in herring gulls has been observed in Lake Michigan fish (Stow et al. 1995, Lamon et al. 1998). Based on limited data available for Green Bay fish, the following trends were described: a decline in alewife PCB concentrations from the late 1970s to 1989; a decline consistent with an exponential decrease in yellow perch from 1976 to 1993 in Zone II, and from 1975 to 1984 in Zone III; and a slight linear decline in PCB concentrations in walleye in Zone III from 1976 to 1996 (Stratus Consulting, Inc. 1999a). Based on the above trends, the uncertainty associated with use of old data is that current risk to receptors may be overestimated.

7.2.4 Selection of TRVs

A comprehensive literature search was conducted to locate studies in which the toxicity of PCBs to wildlife receptors was evaluated. These studies were reviewed to evaluate the appropriateness of using a particular study to derive a TRV. Criteria used to evaluate studies are described in Appendix A, Section A.1; two important factors were study design and species tested. Very few toxicological studies have been conducted using wildlife species. Many TRVs were selected from studies in which the test organism was closely related taxonomically to a selected receptor species. It may be more appropriate to select effect levels derived from test organisms which are closely related trophic-wise (e.g., using an effect level for a carnivorous species such as a kestrel to derive a TRV for a piscivorous species). However, an attempt was made to use consistently conservative assumptions where possible in this risk assessment. Conservative assumptions were used to minimize the possibility of concluding that risk is not present when a threat actually does exist (i.e., the elimination of false negatives). If an acceptable study reported an effect level for a dietary exposure route to a taxonomically related species, the lowest reported LOAEL and NOAEL were selected as the TRV.

An exception to this is the selection of LOAELs and NOAELs for bird species. Effect levels reported for chickens were consistently much lower than effect levels reported for other bird species (Appendix A, Table A-1). Numerous studies have documented the greater sensitivity of chickens to TCDD-like toxicity as compared with wild bird species (Eisler and Belisle 1996, Hoffman et al. 1998, Bosveld and van den Berg 1994, Lorenzen et al. 1997). Dietary LOAELs reported for chickens ranged from 0.0414 to 0.9 mg/kgBW/day, whereas dietary LOAELS reported for other bird species ranged from 1.12 to 36 mg/kgBW/day. We felt a sufficient number of studies had been conducted with other avian species to conclude that effect levels reported for chickens were an anomaly relative to other bird species. Therefore, studies in which chickens were the test species were not selected for the derivation of the NOAEC and LOAEC in this risk assessment. However, if any bird species in the Green Bay area have PCB sensitivities similar to that of the chicken, this risk assessment will underestimate potential effects on that species.

In addition to effect levels reported in the literature as critical body concentrations for fish species, an alternative method was used to determine whole-body concentrations at which adverse effects would be expected. This method was derived based on the observation that whole-body concentrations are often measured, while fish egg concentration measurements are rare. Early life-stages are most sensitive to adverse effects of PCBs, therefore it is important to identify maternal whole-body concentrations which result in critical egg/fry PCB concentrations. The TRV for whole-body concentrations was derived using the TRV identified for fish egg concentrations and an egg to whole-body ratio reported by Mac et al. (1993), and resulted in the lowest LOAEC for fish body concentrations. This LOAEC was selected as the TRV for whole-body concentrations because it addresses the sensitivity of early life stages to PCBs. However, the method used to derive this LOAEC results in the loss of two independent lines of evidence to evaluate toxicity of PCBs to fish. Use of a weight-of evidence approach to evaluate risk reduces uncertainty when all lines lead to similar conclusions about potential risk. However, it was determined that the risk assessment should focus on the most susceptible receptors (early life-stage fish); therefore the most conservative LOAEC was selected as the TRV.

8.0 FISH AND WILDLIFE SERVICE INJURY REPORTS

The assessment area defined for the Natural Resource Damage Assessment (NRDA) conducted by the U.S. Fish and Wildlife Service includes the Lower Fox River and all of Green Bay. Several NRDA reports have been released that assess injuries to natural resources of the Lower Fox River/Green Bay system that have resulted from releases of PCBs to the Lower Fox River. The injury reports for fishery resources (Stratus Consulting, Inc. 1999b) and avian resources (Stratus Consulting, Inc. 1999d) are summarized below so that the conclusions of the NRDA can be compared with the results of this risk assessment.

8.1 Fish and Wildlife Service Injuries to Fishery Resources Report

As part of the larger Great Lakes ecosystem, Green Bay provides important fish habitat and supports a diverse and productive fishery. Although the historic fish community composition has changed due to overfishing and the introduction of exotic species, the fishery resource continues to provide valuable ecological services. The injury report describes PCB transport and exposure pathways in the assessment area.

Pathways by which the fishery resources of Green Bay have been exposed to PCBs released from Lower Fox River paper companies were described based on transport processes (water circulation patterns and sediment transport and deposition patterns) and the spatial and temporal distribution of PCBs in sediment, water and biota in relation to the primary source. Elevated concentrations of PCBs have been document in surface water, sediment, plankton, and fish within the assessment area.

Laboratory and field studies have shown that exposure of fish to PCBs results in adverse effects which meet the NRDA definition of injury. Effects include mortality, promotion or enhanced formation of tumors initiated by other factors, deformities, and impairment of immune and endocrine systems. Early life stages in fish are more sensitive to PCB-related mortality than adult fish (Eisler 1986).

Impacts to fish in the assessment area were evaluated based on measured concentrations of PCBs in fish tissue, and presence of adverse effects associated with PCB exposure. Two general types of changes to fish viability were assessed: adverse effects on fish health, and adverse effects on fish reproduction.

Fish health was evaluated using a suite of tests designed to measure parameters that can be adversely affected by PCB exposure. These included examination of tissues for bacterial, viral and parasitic infections, immunological evaluation of kidney and blood samples, evaluation of liver lesions, and measurement of ethoxyresorufin-O-deethylase (EROD) activity and tissue PCB concentration. Walleye were collected from five locations within the assessment area and two reference locations. Tissue PCB concentrations were significantly higher in assessment area walleye than in fish collected from the reference areas. Assessment area fish also had a significantly higher incidence of liver tumors and pre-tumors. It has been documented that PCBs promote or enhance liver tumor formation (Hendricks et al. 1990); therefore the injury report concluded walleye health has been adversely impacted by PCB exposure.

Adverse effects on reproduction were assessed for lake trout based on historical data, information from the scientific literature, and reproduction and laboratory toxicity studies conducted for the NRDA by the United State Geological Survey (USGS). The toxicity equivalence approach was used to compare historic PCB concentrations in lake trout eggs with toxicity thresholds for embryomortality. Mean egg total PCB concentrations over time were modeled and compared with LD_{10} and LD_{50} concentrations. The analysis concluded that in the mid-1970s egg PCB

concentrations were sufficient to cause sac fry mortality to some Green Bay lake trout eggs; by 1980, concentrations in less than one percent of Lake Michigan lake trout eggs are estimated to have been sufficient to cause mortality. Limited PCB data were available for Green Bay and western Lake Michigan lake trout; analysis of these data suggest PCB concentrations were higher in Green Bay lake trout.

Results of the toxicity studies conducted by the USGS for the NRDA indicate that thiamine deficiency rather than exposure to PCBs or other TCDD-like compounds is currently the primary causal factor for fry mortality in Lake Michigan lake trout. The Trustees concluded that current data do not support concluding that lake trout in Green Bay and Lake Michigan are injured by the PCBs released from Fox River paper companies.

The report concluded that the most significant injury to fishery resources in the Lower Fox River and Green Bay is the presence of extensive fish consumption advisories. Walleye within the assessment area are experiencing increased liver tumors compared with fish from reference areas. Available information does not support concluding that other PCB-related injuries assessed (brown trout and lake trout health, lake trout reproduction) are currently occurring, although they may have in the past.

8.2 Fish and Wildlife Service Avian Injury Report

The Lower Fox River/Green Bay area is an important site within the Great Lakes Ecoregion for breeding and migratory birds (Robbins 1991, Jacobs 1991). The assessment area, due to its comparatively undisturbed nature and the quality and extent of habitats it provides, supports bird populations and communities more diverse than those found in many other areas of the Great Lakes. Because the majority of the PCBs released into the assessment area are concentrated in the aquatic systems of the Fox River and Green Bay (Connolly et al. 1992), the NRDA focused on bird species which utilize aquatic habitats. Critical habitats identified in the NRDA were wetlands and small uninhabited islands in Green Bay that provide nesting sites for colonial waterbirds.

Exposure to a hazardous substance can be characterized by direct measurement of that substance in biota tissue [43 CFR § 11.63(f)(4)(I)]. Numerous studies have been conducted which evaluate PCB concentrations in assessment area birds. For all species and studies where a statistical comparison was made between PCB concentrations in assessment and reference area tissues, PCB concentrations were significantly higher in tissues from the assessment area. Based on evaluation of foraging areas and analysis of PCB concentrations in prey species, the NRDA report concluded that the primary route of exposure for most assessment area bird species is dietary.

Laboratory and field studies have shown that exposure of birds to PCBs results in numerous adverse effects that meet the NRDA definition of injury. These effects include death, behavioral abnormalities, physiological malfunctions and physical deformities. Avian embryos are the life stage most sensitive to PCB toxicity, followed by nestlings, then adults (Hoffman et al. 1998).

Two lines of evidence were used to evaluate injury to avian species utilizing the Fox River/Green Bay assessment area: comparison of egg PCB concentrations to concentrations of PCBs in bird eggs cited in the literature associated with adverse effects on bird reproduction and survival; and field studies conducted in Green Bay which evaluated PCB effects on bird populations.

Based on a literature search, PCB concentrations in eggs ranging from 3 to 20 mg/kg wet weight were identified as a toxic effect concentration range. Mean total PCB concentrations measured in eggs of five assessment area species (double-crested cormorants, Caspian terns, common terns, red-breasted mergansers, and Forster's terns) from 1983 to 1996 were within or exceeded the range where adverse reproductive effects have been shown to occur.

Field studies conducted with eight species (Forster's, common, and Caspian tern; double-crested cormorant; bald eagle; black-crowned night heron; tree swallow; and red-breasted merganser) were evaluated to determine whether sufficient evidence existed to conclude that birds in the assessment area have been injured by exposure to PCBs. Observed effects (decreased hatching success, deformities, edema) and their relationship to measured egg PCB concentrations provide strong evidence that Green Bay Forster's terns have been adversely affected by PCB exposure. Contaminants other than PCBs measured in eggs did not appear to be significant contributors to the observed toxicity. In a single field study conducted with common tern, observed effects were consistent with those observed in Forster's tern and with those caused by PCBs. Available studies do not provide strong evidence that reproductive success of Caspian terns has been adversely affected by PCB exposure, however there is some evidence of increased deformity rates. Two studies concluded that hatch success rates in Green Bay cormorant nests were significantly lower than in control areas; one found no difference between Green Bay and reference site nests. All studies that have compared bill deformity rates in embryos and nestlings between Green Bay and reference sites have found higher rates in Green Bay cormorants. Two studies conducted on bald eagles have found that productivity of Green Bay eagles is significantly lower than at inland sites where eagles are not exposed to point source releases of PCBs. Although studies conducted with black-crowned night heron, tree swallow, and red-breasted merganser conclude that these species have been exposed to PCBs at levels that exceed background concentrations, no significant adverse effects were observed. The conclusion from this evaluation was that sufficient evidence exists to conclude that Forster's, common and Caspian terns, double-crested cormorants, and bald eagles have been injured by PCBs, and that the occurrence of PCB-induced injuries has been widespread throughout the assessment area.

9.0 CONCLUSIONS

A LOAEL is an exposure concentration at which an adverse effect has been observed in a toxicological study; therefore a HQ greater than 1.0 based on a chronic LOAEL indicates that site levels of that contaminant may produce an adverse effect on the ecological receptor in question. The most substantive risk indicated by this risk assessment are lines of evidence where HQ calculations exceeded 1.0 when the LOAEL was used as the effect level and mean PCB concentrations were used as the exposure concentration. This occurred for the bird egg concentrations and the mink food chain model (Table 14).

Although the most substantive risk is indicated if the HQ exceeds one using mean measured PCB concentrations, the maximum concentration is an actual site-specific measured potential exposure concentration. A HQ which exceeds 1.0 using the maximum measured PCB concentration is still an indication of potential risk. The food chain model utilizing double-crested cormorant exposure parameters resulted in a HQ greater than 1.0 when the LOAEL and maximum fish concentrations from the NRDA data set were used in risk calculations.

A calculated HQ greater than 1.0 based on a chronic NOAEL indicates there is a potential chronic risk from that contaminant to the ecological receptor in question. Because concentrations of a contaminant on-site exceed the observed no-effect level for that contaminant, it can not be concluded that there is not risk associated with measured on-site concentrations. This occurred for the fish egg and tissue concentrations, and the food chain models for piscivorous birds.

Lines of evidence evaluated for this risk assessment for each individual assessment endpoint and conclusions based on the risk characterization for each are discussed below.

9.1 Pelagic Fish Reproduction and Survival

Two lines of evidence were used to estimate risk to pelagic fish reproduction and survival in the upper Green Bay: comparison of fish tissue and egg concentrations to adverse effect levels cited in

published studies. Hazard quotients calculated using the NOAEC exceeded 1.0 for both egg and upper trophic level fish tissue PCB concentrations (ranging from 3.9 to 9.5 and 4.0 to 9.4, respectively), indicating potential risk to pelagic fish reproduction and survival.

Although the conclusion of this risk assessment is potential risk to the selected receptor species, (lake trout) and the NRDA fish injury assessment concluded no actual adverse effects to lake trout reproduction are currently occurring, the two reports are not inconsistent. Lake trout were utilized in this risk assessment as a representative pelagic fish species; the risk characterization indicated potential risk to pelagic fish based on contaminant concentrations in eggs and fish tissue which exceed concentrations at which no adverse impacts have been documented. Although the reported LOAEC was not exceeded, a LOAEC derived from the literature is not necessarily the lowest concentration at which an adverse effect will occur, it is simply the lowest concentration that has been tested. Because concentrations of a contaminant on-site exceed the observed no-effect level for that contaminant, it can not be concluded that there is not risk associated with measured on-site concentrations. The NRDA fish injury assessment did find actual adverse effects which are consistent with effects observed after PCB exposure in another pelagic fish, walleye (increased incidence of liver tumors).

9.2 Piscivorous Bird Reproduction and Survival

Three lines of evidence were used to evaluate risk to piscivorous birds utilizing the upper Green Bay area: comparison of bird egg concentrations to adverse effect levels published in the literature; modeled food chain exposure and comparison of estimated dietary exposure concentrations to published adverse effect levels; and published studies on birds utilizing the upper Green Bay.

Comparison of bird egg concentrations to adverse effect levels cited in the literature indicates that piscivorous birds utilizing the upper Green Bay area are at risk from exposure to PCBs. Measured concentrations of PCBs in Caspian tern and double-crested cormorant eggs exceed levels shown to cause adverse reproductive effects (hazard quotients ranging from 1.3 to 25.1).

Food chain exposure models indicate that piscivorous birds are potentially at risk from dietary PCB exposure levels; all except one HQ calculated using the NOAEL as the effect level exceeded 1.0 (range between 0.8 and 16.2). The HQ calculated using a double-crested cormorant exposure model exceeded 1.0 (HQ = 1.6) when the LOAEL was used as the effect level and maximum fish concentrations from the NRDA data set were used.

Published studies in which the effects of PCBs on birds inhabiting the upper Green Bay were reviewed as the third line of evidence for this assessment endpoint. This line of evidence also indicates that piscivorous birds may be at risk from PCB exposure. Adverse effects associated with PCB exposure (decreased hatching success, embryo deformities) were not observed in studies conducted with Caspian terns. One study found a strong negative correlation between nest site tenacity and PCB concentrations, however population-level implications of subtle behavioral changes are not known. Studies conducted on double-crested cormorants in the upper Green Bay indicate this species has experienced adverse reproductive effects. Hatch success rates were lower and physical deformity rates were higher in Green Bay cormorants than at reference sites. PCBs have been shown in laboratory experiments to cause deformities in avian embryos similar to those seen in Green Bay cormorants (crossed bills, edema, dwarfism).

The weight of evidence used to evaluate risk to piscivorous birds indicates these species are potentially at risk from PCB exposure. Results from food chain exposure models indicate greater risk using the double-crested cormorant model, which correlates with results observed in field studies.

Double-crested cormorants were utilized in this risk assessment as a model for piscivorous birds. Although cormorant populations in the Great Lakes are doing well, risk calculations indicate that other species within this feeding guild may be at risk for experiencing adverse effects. Factors contributing to the cormorant population increase observed in the Great Lakes since 1973 include a rise in the numbers of prey fish, decreased levels of toxic chemicals, a decrease in commercial fishing, and legislation which protects cormorants (Environment Canada 1995). An additional point which should be noted is that the decline in PCB concentrations measured in bird eggs in the late 1970s reached a plateau in the mid-1980s; relatively little decline has occurred since. Although the primary source of PCBs to the upper bay has been eliminated, exposure concentrations for birds appear to have remained similar for the last decade.

9.3 Piscivorous Mammal Reproduction and Survival

The food chain model used to evaluate risk to piscivorous mammals indicates mink are at risk from PCB exposure in the upper Green Bay area. All HQs calculated for this species exceeded 1.0, and ranged from 2.1 (LOAEL as the effect concentration, mean overall fish PCB concentration) to 397.8 (NOAEL as the effect concentration, maximum fish PCB concentration from the NRDA data set).

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Table 1. TRVs Selected for Use in the Upper Green Bay Risk Assessment
Upper Green Bay Portion of the Fox River Site
Green Bay, WI
February 2000

RECEPTOR	MEDIA	TRV		UNITS	EFFECT	REFERENCE
		NOAEL	LOAEL			
Fish	Egg	0.16	1.6	mg/kg, ww	Decreased fry growth	Hendricks et al. 1981
Fish	Whole-body	0.77	7.7	mg/kg, ww	Estimated based on egg LOAEL, egg:body ratio	Mac et al. 1993
Piscivorous Bird	Egg	4.7	7.6	mg/kg, ww	Decreased hatching success	Hoffman et al. 1993
Piscivorous Bird	Egg	0.8	8	mg/kg, ww	Decreased hatching success, increased deformity rate	Ludwig et al. 1996
Piscivorous Bird	Diet	0.112	1.12	mg/kgBW/day	Reproductive success and nesting behavior	Tori and Peterle 1983, Peakall and Peakall 1973, Peakall et al. 1972
Piscivorous Mammal	Diet	0.004	0.13	mg/kgBW/day	Kit survival	Heaton et al. 1995

TRV = Toxicity reference value

NOAEL = No observed adverse effect level

LOAEL = Lowest observed adverse effect level

mg/kg, ww = milligrams per kilogram, wet weight

mg/kgBW/day = milligrams per kilogram body weight per day

Table 2. PCB Concentrations in Surface Water,Upper Green Bay
Green Bay Mass Balance Model Data Set
Green Bay, WI
February 2000

		PCB Concentrations (µg/L)		
Sample Number	SampleDate	Dissolved	Particulate	Total
89GG20S23	4/30/89	0.00095	0.00099	0.00195
89GG20S43	5/1/89	0.00073	0.00238	0.00311
89GG20S63	5/1/89	0.00081	0.00044	0.00125
89GG20S83	4/30/89	0.00083	0.00057	0.00140
89GG21S03	4/30/89	0.00070	0.00207	0.00277
89GG21S23	5/1/89	0.00101	0.00131	0.00232
89GG32S63	6/9/89	0.00097	0.00040	0.00137
89GG32S65	6/9/89	0.00051	0.00024	0.00074
89GG32S83	6/8/89	0.00080	0.00019	0.00098
89GG33S03	6/9/89	0.00066	0.00035	0.00101
89GG33S05	6/10/89	0.00049	0.00025	0.00074
89GG33S23	6/10/89	0.00045	0.00027	0.00072
89GG33S43	6/10/89	0.00049	0.00027	0.00076
89GG33S63	6/10/89	0.00057	0.00037	0.00094
89GG33S65	6/10/89	0.00057	0.00047	0.00104
89GG33S83	6/11/89	0.00053	0.00035	0.00088
89GG33S85	6/11/89	0.00058	0.00038	0.00096
89GG42S63	7/28/89	0.00044	0.00024	0.00068
89GG42S65	7/28/89	0.00039	0.00024	0.00062
89GG42S83	7/28/89	0.00048	0.00035	0.00083
89GG43S03	7/29/89	0.00045	0.00029	0.00074
89GG43S05	7/29/89	0.00054	0.00029	0.00083
89GG43S23	7/29/89	0.00040	0.00021	0.00061
89GG43S25	7/29/89	0.00038	0.00030	0.00068
89GG43S43	7/29/89	0.00035	0.00022	0.00057
89GG43S45	7/29/89	0.00051	0.00030	0.00081
89GG43S63	7/29/89	0.00041	0.00023	0.00064
89GG43S65	7/29/89	0.00000 U	0.00028	0.00028
89GG43S83	7/30/89	0.00050	0.00000 U	0.00050
89GG50S43	9/13/89	0.00045	0.00017	0.00062
89GG50S45	9/13/89	0.00069	0.00012	0.00081
89GG50S63	9/13/89	0.00048	0.00016	0.00064
89GG50S83	9/14/89	0.00055	0.00018	0.00073
89GG50S85	9/14/89	0.00052	0.00034	0.00085
89GG51S03	9/14/89	0.00038	0.00019	0.00056
89GG51S05	9/14/89	0.00048	0.00019	0.00067
89GG51S23	9/14/89	0.00076	0.00021	0.00096
89GG51S25	9/14/89	0.00066	0.00038	0.00104
89GG51S43	9/15/89	0.00063	0.00020	0.00083
89GG51S45	9/15/89	0.00061	0.00033	0.00094
89GG51S63	9/15/89	0.00055	0.00021	0.00076
89GG51S65	9/15/89	0.00049	0.00024	0.00072
90GG02S63	10/20/89	0.00073	0.00037	0.00110
90GG02S83	10/21/89	0.00062	0.00075	0.00136
90GG03S03	10/21/89	0.00067	0.00038	0.00105
90GG03S23	10/21/89	0.00065	0.00043	0.00108
90GG03S43	10/21/89	0.00088	0.00051	0.00140
90GG03S63	10/21/89	0.00089	0.00059	0.00148
90GG03S83	10/22/89	0.00083	0.00123	0.00206
90GG10S63	2/17/90	0.00083	0.00052	0.00135
90GG20S43	4/26/90	0.00045	0.00025	0.00070
90GG20S63	4/27/90	0.00042	0.00032	0.00074
90GG20S83	4/27/90	0.00052	0.00032	0.00085
90GG21S03	4/27/90	0.00041	0.00046	0.00087
90GG21S23	4/27/90	0.00038	0.00029	0.00067
90GG21S43	4/27/90	0.00056	0.00040	0.00096
90GG21S63	4/28/90	0.00035	0.00040	0.00075
Mean Water Concentration, Total PCBs				0.00100
Maximum Water Concentration, Total PCBs				0.00311

µg/L = micrograms per liter
U = Not detected

Table 3. Total PCB Concentrations in Sediment Collected from the Upper Green Bay
Green Bay Mass Balance Model Data Set
Green Bay, WI
February 2000

Sample #	Year Collected	Total PCB Concentrations (µg/kg, ww)
E052B09A	1987	18.61
E326B07A	1988	20.13
E309B11A	1988	19.54
E339B02A	1988	23.86
D342B08A	1989	18.68
D344B02A	1989	5.07
D342B02A	1989	12.83
D344B05A	1989	4.63
D344B08A	1989	3.24
E054B08A	1989	6.53
E063B07A	1989	11.47
E071B03A	1989	13.74
E148B02A	1989	16.92
D342B05A	1989	5.78
E184B02A	1989	27.07
E054B02A	1989	10.5
E054B05A	1989	7.98
E304B18A	1989	18.86
E148B07A	1989	19.18
E197B09A	1990	5.22
E204B02A	1990	9.73
E284B08A	1990	3.95
E204B05A	1990	9.31
E284B03A	1990	5.08
E191B05A	1990	4.65
E319B17A	1990	8.74
E191B02A	1990	2.4
E284B17A	1990	3.67
Mean Sediment Concentration:		11.33
Maximum Sediment Concentration:		27.07

µg/kg, ww = micrograms per kilogram, wet weight

Table 4. Total PCB Concentrations in Fish Collected in 1996 from Upper Green Bay
Natural Resource Damage Assessment Data Set
Upper Green Bay
Green Bay, WI
February 2000

Sample #	Species	Number of Individual Fish per Sample	Total PCBs (mg/kg,ww)
UPPER TROPHIC LEVEL FISH:			
BTUG02CP	Brown Trout	5	1.75
BTUG04CP	Brown Trout	5	1.75
BTUG03CP	Brown Trout	6	1.17
BTUG05CP	Brown Trout	4	1.98
BTUG01CP	Brown Trout	5	1.70
WEUG02CP	Walleye	3	4.61
WEUG03CP	Walleye	3	7.26
WEUG01CP	Walleye	4	5.65
Mean:			3.23
Maximum:			7.26

mg/kg, ww = milligrams per kilogram, wet weight

Table 5. Total PCB Concentrations in Fish Collected in 1989 from Upper Green Bay
Green Bay Mass Balance Model Data Set
Upper Green Bay
Green Bay, WI
February 2000

Sample Number	Date Collected	Species	Total PCBs (mg/kg, ww)
FORAGE FISH			
WDI119001BC1	09/11/89	Alewife	0.11
WDJ049008BC1	10/04/89	Alewife	2.00
WDF199001BC1	06/19/89	Alewife	0.25
WDG189001BC1	07/18/89	Alewife	0.98
WDG189002BC1	07/18/89	Alewife	0.85
WDG189003BC1	07/18/89	Alewife	0.90
WDJ049009BC1	10/04/89	Alewife	1.80
WDJ049010BC1	10/04/89	Alewife	1.40
WDF199027BC1	06/19/89	Carp	3.70
WDI129011BC1	09/12/89	Carp	2.90
WDF199025BC1	06/19/89	Carp	4.10
WDK099005BC1	11/09/89	Carp	2.40
WDJ039031BC1	10/03/89	Carp	1.70
WDJ039028BC1	10/03/89	Carp	3.20
WDK089003BC1	11/08/89	Carp	1.90
WDI129015BC1	09/12/89	Carp	4.20
WDI129014BC1	09/12/89	Carp	1.80
WDJ039026BC1	10/03/89	Carp	2.50
WDI069008BC1	09/06/89	Rainbow Smelt	0.84
WDI069006BC1	09/06/89	Rainbow Smelt	0.26
WDI069005BC1	09/06/89	Rainbow Smelt	0.52
WDI069001BC1	09/06/89	Rainbow Smelt	0.78
WDJ049018BC1	10/04/89	Rainbow Smelt	1.60
WDI069004BC1	09/06/89	Rainbow Smelt	0.29
WDI069003BC1	09/06/89	Rainbow Smelt	0.33
WDF199007BC1	06/19/89	Rainbow Smelt	0.47
WDF199006BC1	06/19/89	Rainbow Smelt	0.53
WDF199005BC1	06/19/89	Rainbow Smelt	0.44
WDF199004BC1	06/19/89	Rainbow Smelt	0.43
WDJ049024BC1	10/04/89	Rainbow Smelt	0.22
WDJ049021BC1	10/04/89	Rainbow Smelt	0.19
WDJ049022BC1	10/04/89	Rainbow Smelt	0.81
WDJ049023BC1	10/04/89	Rainbow Smelt	1.10
WDF199003BC1	06/19/89	Rainbow Smelt	0.15
WDJ049025BC1	10/04/89	Rainbow Smelt	0.34
WDF199002BC1	06/19/89	Rainbow Smelt	0.16
Mean for Forage Fish:			1.28
Maximum for Forage Fish:			4.20

Table 5. Total PCB Concentrations in Fish Collected in 1989 from Upper Green Bay
Green Bay Mass Balance Model Data Set
Upper Green Bay
Green Bay, WI
February 2000

Sample Number	Date Collected	Species	Total PCBs (mg/kg, ww)
UPPER TROPHIC LEVEL FISH			
WDF079018BC1	06/07/89	Brown Trout	1.80
WDJ099001BC1	12/30/99	Brown Trout	2.20
WDJ099003BC1	12/30/99	Brown Trout	3.90
WDJ189006BC1	12/30/99	Brown Trout	2.80
WDJ189007BC1	12/30/99	Brown Trout	2.40
WDJ189010BC1	12/30/99	Brown Trout	2.70
WDJ099002BC1	12/30/99	Brown Trout	2.30
WDF069003BC1	06/06/89	Brown Trout	1.70
WDG079001BC1	07/07/89	Brown Trout	2.30
WDG209001BC1	07/20/89	Brown Trout	2.30
WDF069002BC1	06/06/89	Brown Trout	2.90
WDG209003BC1	07/20/89	Brown Trout	3.70
WDG209002BC1	07/20/89	Brown Trout	3.10
WDI219001BC1	12/30/99	Walleye	3.80
WDF139004BC1	06/13/89	Walleye	4.80
WDJ149001BC1	12/30/99	Walleye	5.90
WDF139003BC1	06/13/89	Walleye	3.30
WDF139002BC1	06/13/89	Walleye	3.20
WDF139001BC1	06/13/89	Walleye	2.50
WDJ229002BC1	12/30/99	Walleye	0.62
WDJ229003BC1	12/30/99	Walleye	5.70
WDJ319001BC1	12/30/99	Walleye	2.10
WDK089001BC1	12/30/99	Walleye	3.70
WDI209002BC1	12/30/99	Walleye	3.20
WDG209004BC1	07/20/89	Walleye	3.30
WDK149001BC1	12/30/99	Walleye	1.30
Mean for Upper Trophic Level Fish:			2.98
Maximum for Upper Trophic Level Fish:			5.90

mg/kg, ww = milligrams per kilogram, wet weight

Table 6. Total PCB Concentrations in Upper Trophic Level Fish Collected from Upper Green Bay
NRDA and Mass Balance Data Sets Combined
Upper Green Bay
Green Bay, WI
February 2000

Sample #	Database	Species	Total PCBs (mg/kg, ww)
BTUG02CP	NRDA	Brown Trout	1.75
BTUG05CP	NRDA	Brown Trout	1.98
BTUG01CP	NRDA	Brown Trout	1.70
BTUG03CP	NRDA	Brown Trout	1.17
BTUG04CP	NRDA	Brown Trout	1.75
WDF079018BC1	Mass Balance Model	Brown Trout	1.80
WDJ099001BC1	Mass Balance Model	Brown Trout	2.20
WDJ099003BC1	Mass Balance Model	Brown Trout	3.90
WDJ189006BC1	Mass Balance Model	Brown Trout	2.80
WDJ189007BC1	Mass Balance Model	Brown Trout	2.40
WDJ189010BC1	Mass Balance Model	Brown Trout	2.70
WDJ099002BC1	Mass Balance Model	Brown Trout	2.30
WDF069003BC1	Mass Balance Model	Brown Trout	1.70
WDG079001BC1	Mass Balance Model	Brown Trout	2.30
WDG209001BC1	Mass Balance Model	Brown Trout	2.30
WDF069002BC1	Mass Balance Model	Brown Trout	2.90
WDG209003BC1	Mass Balance Model	Brown Trout	3.70
WDG209002BC1	Mass Balance Model	Brown Trout	3.10
WEUG02CP	NRDA	Walleye	4.61
WEUG03CP	NRDA	Walleye	7.26
WEUG01CP	NRDA	Walleye	5.65
WDI219001BC1	Mass Balance Model	Walleye	3.80
WDF139004BC1	Mass Balance Model	Walleye	4.80
WDJ149001BC1	Mass Balance Model	Walleye	5.90
WDF139003BC1	Mass Balance Model	Walleye	3.30
WDF139002BC1	Mass Balance Model	Walleye	3.20
WDF139001BC1	Mass Balance Model	Walleye	2.50
WDJ229002BC1	Mass Balance Model	Walleye	0.62
WDJ229003BC1	Mass Balance Model	Walleye	5.70
WDJ319001BC1	Mass Balance Model	Walleye	2.10
WDK089001BC1	Mass Balance Model	Walleye	3.70
WDI209002BC1	Mass Balance Model	Walleye	3.20
WDG209004BC1	Mass Balance Model	Walleye	3.30
WDK149001BC1	Mass Balance Model	Walleye	1.30
Overall Mean:			**3.04**
Overall Maximum:			**7.26**

NRDA = Natural Resource Damage Assessment Data Set

Mass Balance Model = Green Bay Mass Balance Model Data Set

mg/kg, ww = milligrams per kilogram, wet weight

Table 7. Estimated Total PCB Concentrations in Fish Eggs
Based on Fish Whole Body PCB Concentrations
Upper Green Bay
Green Bay, WI
February 2000

Whole Body Sample #	Database	Species	Estimated Total PCBs in Fish Eggs (mg/kg, ww)
BTUG02CP	NRDA	Brown Trout	0.37
BTUG05CP	NRDA	Brown Trout	0.41
BTUG01CP	NRDA	Brown Trout	0.36
BTUG03CP	NRDA	Brown Trout	0.24
BTUG04CP	NRDA	Brown Trout	0.37
WEUG02CP	NRDA	Walleye	0.96
WEUG01CP	NRDA	Walleye	1.18
WEUG03CP	NRDA	Walleye	1.52
NRDA Database Mean:			0.68
NRDA Database Maximum:			1.52
WDF079018BC1	Mass Balance Model	Brown Trout	0.38
WDJ099001BC1	Mass Balance Model	Brown Trout	0.46
WDJ099003BC1	Mass Balance Model	Brown Trout	0.82
WDJ189006BC1	Mass Balance Model	Brown Trout	0.59
WDJ189007BC1	Mass Balance Model	Brown Trout	0.50
WDJ189010BC1	Mass Balance Model	Brown Trout	0.56
WDJ099002BC1	Mass Balance Model	Brown Trout	0.48
WDF069003BC1	Mass Balance Model	Brown Trout	0.36
WDG079001BC1	Mass Balance Model	Brown Trout	0.48
WDG209001BC1	Mass Balance Model	Brown Trout	0.48
WDF069002BC1	Mass Balance Model	Brown Trout	0.61
WDG209003BC1	Mass Balance Model	Brown Trout	0.77
WDG209002BC1	Mass Balance Model	Brown Trout	0.65
WDI219001BC1	Mass Balance Model	Walleye	0.79
WDF139004BC1	Mass Balance Model	Walleye	1.00
WDJ149001BC1	Mass Balance Model	Walleye	1.23
WDF139003BC1	Mass Balance Model	Walleye	0.69
WDF139002BC1	Mass Balance Model	Walleye	0.67
WDF139001BC1	Mass Balance Model	Walleye	0.52
WDJ229002BC1	Mass Balance Model	Walleye	0.13
WDJ229003BC1	Mass Balance Model	Walleye	1.19
WDJ319001BC1	Mass Balance Model	Walleye	0.44
WDK089001BC1	Mass Balance Model	Walleye	0.77
WDI209002BC1	Mass Balance Model	Walleye	0.67
WDG209004BC1	Mass Balance Model	Walleye	0.69
WDK149001BC1	Mass Balance Model	Walleye	0.27
Overall Mean:			0.64
Overall Maximum:			1.52

mg/kg, ww = milligrams per kilogram, wet weight

Egg concentration = Whole-body PCB concentration times 0.209

0.209 = Egg to whole body ratio calculated for lake trout; Mac et al. 1993

NRDA = Natural Resource Damage Assessment Data Set

Table 8. Total PCB Concentrations in Bird Eggs Collected from Islands In or Near Upper Green Bay
Upper Green Bay
Green Bay, WI
February 2000

Collection Location	Collection Year	Species	Mean Total PCBs (mg/kg, ww)	Standard Deviation	# of eggs	Reference
Gravelly Island	1980	Caspian tern	36.2	9.2	10	Struger and Weseloh 1985
Gravelly/Gull Islands	1988	Caspian tern	11	nd	18	Yamashita et al. 1993
Gravelly Island	1991	Caspian tern	15.8	nd	10	Ewins et al. 1994
Little Gull Island	1986	Double-crested cormorant	14.8	0.1	nd	Tillett et al. 1992
Gravelly/Little Gull Islands	1987	Double-crested cormorant	12.3	0.6	nd	Tillett et al. 1992
Spider Island	1988	Double-crested cormorant	5.3	0.3	nd	Tillett et al. 1992
Little Gull Island	1988	Double-crested cormorant	7.2	nd	41	Yamashita et al. 1993
Spider/Hog/Fish Islands	1988	Double-crested cormorant	14.2	nd	38	Dale and Stromborg1993
Spider Island	1989	Double-crested cormorant	15.5	8.04	27	Williams et al. 1995
Spider Island	1989 - 1990	Double-crested cormorant	7.8	3.3	26	Larson et al. 1996
Spider Island	1994 - 1995	Double-crested cormorant	10.4*	4.6	10	Custer et al. in press

mg/kg, ww = milligrams per kilogram, wet weight

* wet weight vs. dry weight not specified

nd = no data available

Table 9. Hazard Quotient Calculations for Fish
Upper Green Bay
Green Bay, WI
February 2000

EGGS:

	Estimated PCB Conc. (mg/kg, ww)	Fish Egg NOAEC (µg/kg, ww)	Fish Egg LOAEC (µg/kg, ww)	Fish Egg HQ using the NOAEC	Fish Egg HQ using the LOAEC
NRDA Database Mean	0.68	0.16	1.6	4.2	0.4
NRDA Database Max.	1.52	0.16	1.6	9.5	0.9
Overall Mean	0.64	0.16	1.6	4.0	0.4
Overall Max.	1.52	0.16	1.6	9.5	0.9

WHOLE BODY:

	PCB Conc. (mg/kg, ww)	Whole Body NOAEC (mg/kg, ww)	Whole Body LOAEC (mg/kg, ww)	Whole Body HQ using the NOAEC	Whole Body HQ using the LOAEC
NRDA Database Mean	3.23	0.77	7.7	4.2	0.4
NRDA Database Max.	7.26	0.77	7.7	9.4	0.9
Overall Mean	3.04	0.77	7.7	3.9	0.4
Overall Max.	7.26	0.77	7.7	9.4	0.9

mg/kg, ww = milligrams per kilogram, wet weight

µg/kg, ww = micrograms per kilogram, wet weight

NOAEC = No observable adverse effect concentration

LOAEC = Lowest observed adverse effect concentration

HQ = Hazard quotient

Table 10. Hazard Quotient Calculations for Bird Eggs
Upper Green Bay
Green Bay, WI
February 2000

	Bird Egg PCB Conc. (mg/kg, ww)	Bird Egg NOAEC [a] (mg/kg, ww)	Bird Egg LOAEC [a] (mg/kg, ww)	Bird Egg HQ using the NOAEC	Bird Egg HQ using the LOAEC
Caspian tern	15.8 (mean)	4.7	7.6	3.4	2.1
Double-crested cormorant	10.4 (mean)	4.7	7.6	2.2	1.4
Double-crested cormorant	20.1(max)	4.7	7.6	4.3	2.6

	Bird Egg PCB Conc. (mg/kg, ww)	Bird Egg NOAEC [b] (mg/kg, ww)	Bird Egg LOAEC [b] (mg/kg, ww)	Bird Egg HQ using the NOAEC	Bird Egg HQ using the LOAEC
Caspian tern	15.8 (mean)	0.8	8	19.8	1.9
Double-crested cormorant	10.4 (mean)	0.8	8	13.0	1.3
Double-crested cormorant	20.1(max)	0.8	8	25.1	2.5

[a] TRV from Hoffman et al. 1993. Effect observed was decreased hatching success.

[b] TRV from Ludwig et al. 1996. Effect observed was increased deformity rate.

mg/kg, ww = milligrams per kilogram, wet weight

NOAEC = No observable adverse effect concentration

LOAEC = Lowest observed adverse effect concentration

HQ = Hazard quotient

Table 11. Food Chain Model and Chronic Hazard Quotient Calculations for the Caspian Tern
Upper Green Bay
Green Bay, WI
February 2000

Using the NOAEL and the Maximum PCB Concentrations:

Maximum Water Conc. (mg/L)	Source of Fish Data	Maximum Fish Conc. (mg/kg, ww)	Water Ing. Rate (L/day)	Food Ing. Rate (kg/day)	AUF	Body Weight (kg)	Dose (mg/kg BW/day)	NOAEL (mg/kg BW/day)	HQ Without Water Ingestion	HQ With Water Ingestion
0.00000311	NRDA Database	7.26	0.04	0.0405	1	0.574	0.51	0.112	4.6	4.6
0.00000311	Mass Balance Model*	4.20	0.04	0.0405	1	0.574	0.30	0.112	2.6	2.6

Using the NOAEL and the Mean PCB Concentrations:

Mean Water Conc. (mg/L)	Source of Fish Data	Mean Fish Conc. (mg/kg, ww)	Water Ing. Rate (L/day)	Food Ing. Rate (kg/day)	AUF	Body Weight (kg)	Dose (mg/kg BW/day)	NOAEL (mg/kg BW/day)	HQ Without Water Ingestion	HQ With Water Ingestion
0.00000100	NRDA Database	3.23	0.04	0.0405	1	0.574	0.23	0.112	2.0	2.0
0.00000100	Mass Balance Model*	1.28	0.04	0.0405	1	0.574	0.09	0.112	0.8	0.8

Using the LOAEL and the Maximum PCB Concentrations:

Maximum Water Conc. (mg/L)	Source of Fish Data	Maximum Fish Conc. (mg/kg, ww)	Water Ing. Rate (L/day)	Food Ing. Rate (kg/day)	AUF	Body Weight (kg)	Dose (mg/kg BW/day)	LOAEL (mg/kg BW/day)	HQ Without Water Ingestion	HQ With Water Ingestion
0.00000311	NRDA Database	7.26	0.04	0.0405	1	0.574	0.51	1.12	0.5	0.5
0.00000311	Mass Balance Model*	4.20	0.04	0.0405	1	0.574	0.30	1.12	0.3	0.3

Using the LOAEL and the Mean PCB Concentrations:

Mean Water Conc. (mg/L)	Source of Fish Data	Mean Fish Conc. (mg/kg, ww)	Water Ing. Rate (L/day)	Food Ing. Rate (kg/day)	AUF	Body Weight (kg)	Dose (mg/kg BW/day)	LOAEL (mg/kg BW/day)	HQ Without Water Ingestion	HQ With Water Ingestion
0.00000100	NRDA Database	3.23	0.04	0.0405	1	0.574	0.23	1.12	0.2	0.2
0.00000100	Mass Balance Model*	1.28	0.04	0.0405	1	0.574	0.09	1.12	0.1	0.1

NRDA = Natural Resource Damage Assessment Data Set
* Data from the Mass Balance Model is for forage fish only.
mg/kg, ww = milligrams per kilogram, wet weight
mg/L = milligrams per liter
L/day = liters per day
kg/day = kilograms per day
AUF = area use factor
mg/kgBW/day = milligrams per kilogram body weight per day
NOAEL = No observable adverse effect level
LOAEL = Lowest observed adverse effect level
HQ = Hazard quotient

Table 12. Food Chain Model and Chronic Hazard Quotient Calculations for the Double-Crested Cormorant
Upper Green Bay
Green Bay, WI
February 2000

Using the NOAEL and the Maximum PCB Concentrations:

Maximum Water Conc. (mg/L)	Source of Fish Data	Maximum Fish Conc. (mg/kg, ww)	Water Ing. Rate (L/day)	Food Ing. Rate (kg/day)	AUF	Body Weight (kg)	Dose (mg/kg BW/day)	NOAEL (mg/kg BW/day)	HQ Without Water Ingestion	HQ With Water Ingestion
0.00000311	NRDA Database	7.26	0.079	0.475	1	1.9	1.81	0.112	16.2	16.2
0.00000311	Mass Balance Model*	4.20	0.079	0.475	1	1.9	1.05	0.112	9.4	9.4

Using the NOAEL and the Mean PCB Concentrations:

Mean Water Conc. (mg/L)	Source of Fish Data	Mean Fish Conc. (mg/kg, ww)	Water Ing. Rate (L/day)	Food Ing. Rate (kg/day)	AUF	Body Weight (kg)	Dose (mg/kg BW/day)	NOAEL (mg/kg BW/day)	HQ Without Water Ingestion	HQ With Water Ingestion
0.00000100	NRDA Database	3.23	0.079	0.475	1	1.9	0.81	0.112	7.2	7.2
0.00000100	Mass Balance Model*	1.28	0.079	0.475	1	1.9	0.32	0.112	2.9	2.9

Using the LOAEL and the Maximum PCB Concentrations:

Maximum Water Conc. (mg/L)	Source of Fish Data	Maximum Fish Conc. (mg/kg, ww)	Water Ing. Rate (L/day)	Food Ing. Rate (kg/day)	AUF	Body Weight (kg)	Dose (mg/kg BW/day)	LOAEL (mg/kg BW/day)	HQ Without Water Ingestion	HQ With Water Ingestion
0.00000311	NRDA Database	7.26	0.079	0.475	1	1.9	1.81	1.12	1.6	1.6
0.00000311	Mass Balance Model*	4.20	0.079	0.475	1	1.9	1.05	1.12	0.9	0.9

Using the LOAEL and the Mean PCB Concentrations:

Mean Water Conc. (mg/L)	Source of Fish Data	Mean Fish Conc. (mg/kg, ww)	Water Ing. Rate (L/day)	Food Ing. Rate (kg/day)	AUF	Body Weight (kg)	Dose (mg/kg BW/day)	LOAEL (mg/kg BW/day)	HQ Without Water Ingestion	HQ With Water Ingestion
0.00000100	NRDA Database	3.23	0.079	0.475	1	1.9	0.81	1.12	0.7	0.7
0.00000100	Mass Balance Model*	1.28	0.079	0.475	1	1.9	0.32	1.12	0.3	0.3

NRDA = Natural Resource Damage Assessment Data Set
* Data from the Mass Balance Model is for forage fish only.
mg/kg, ww = milligrams per kilogram, wet weight
mg/L = milligrams per liter
L/day = liters per day
kg/day = kilograms per day
AUF = area use factor
mg/kgBW/day = milligrams per kilogram body weight per day
NOAEL = No observable adverse effect level
LOAEL = Lowest observed adverse effect level
HQ = Hazard quotient

Table 13. Food Chain Model and Chronic Hazard Quotient Calculations for the Mink
Upper Green Bay
Green Bay, WI
February 2000

Using the NOAEL and the Maximum PCB Concentrations:

Maximum Water Conc. (mg/L)	Maximum Sediment Conc. (mg/kg, ww)	Source of Fish Data	Maximum Fish Conc. (mg/kg, ww)	Water Ing. Rate (L/day)	Sediment Ing. Rate (kg/day)	Food Ing. Rate (kg/day)	AUF	Body Weight (kg)	Dose (mg/kg BW/day)	NOAEL (mg/kg BW/day)	HQ Without Sed. or Water Ingestion	HQ with Sed and Water Ingestion
0.00000311	0.027	NRDA Database	7.26	0.0572	0.0103	0.114	1	0.52	1.59	0.004	397.6	397.8
0.00000311	0.027	Mass Balance Model*	4.20	0.0572	0.0103	0.114	1	0.52	0.92	0.004	230.2	230.3

Using the NOAEL and the Mean PCB Concentrations:

Mean Water Conc. (mg/L)	Mean Sediment Conc. (mg/kg, ww)	Source of Fish Data	Mean Fish Conc. (mg/kg, ww)	Water Ing. Rate (L/day)	Sediment Ing. Rate (kg/day)	Food Ing. Rate (kg/day)	AUF	Body Weight (kg)	Dose (mg/kg BW/day)	NOAEL (mg/kg BW/day)	HQ Without Sed. or Water Ingestion	HQ with Sed and Water Ingestion
0.00000100	0.011	NRDA Database	3.23	0.0572	0.0103	0.114	1	0.52	0.71	0.004	177.2	177.3
0.00000100	0.011	Mass Balance Model*	1.28	0.0572	0.0103	0.114	1	0.52	0.28	0.004	70.2	70.2

Using the LOAEL and the Maximum PCB Concentrations:

Maximum Water Conc. (mg/L)	Maximum Sediment Conc. (mg/kg, ww)	Source of Fish Data	Maximum Fish Conc. (mg/kg, ww)	Water Ing. Rate (L/day)	Sediment Ing. Rate (kg/day)	Food Ing. Rate (kg/day)	AUF	Body Weight (kg)	Dose (mg/kg BW/day)	LOAEL (mg/kg BW/day)	HQ Without Sed. or Water Ingestion	HQ with Sed and Water Ingestion
0.00000311	0.027	NRDA Database	7.26	0.0572	0.0103	0.114	1	0.52	1.59	0.134	11.9	11.9
0.00000311	0.027	Mass Balance Model*	4.20	0.0572	0.0103	0.114	1	0.52	0.92	0.134	6.9	6.9

Using the LOAEL and the Mean PCB Concentrations:

Mean Water Conc. (mg/L)	Mean Sediment Conc. (mg/kg, ww)	Source of Fish Data	Mean Fish Conc. (mg/kg, ww)	Water Ing. Rate (L/day)	Sediment Ing. Rate (kg/day)	Food Ing. Rate (kg/day)	AUF	Body Weight (kg)	Dose (mg/kg BW/day)	LOAEL (mg/kg BW/day)	HQ Without Sed. or Water Ingestion	HQ with Sed and Water Ingestion
0.00000100	0.011	NRDA Database	3.23	0.0572	0.0103	0.114	1	0.52	0.71	0.134	5.3	5.3
0.00000100	0.011	Mass Balance Model*	1.28	0.0572	0.0103	0.114	1	0.52	0.28	0.134	2.1	2.1

NRDA = Natural Resource Damage Assessment Data Set
* Data from the Mass Balance Model is for forage fish only.
mg/kg, ww = milligrams per kilogram, wet weight
mg/L = milligrams per liter
L/day = liters per day
kg/day = kilograms per day
AUF = area use factor
mg/kgBW/day = milligrams per kilogram body weight per day
NOAEL = No observable adverse effect level
LOAEL = Lowest observed adverse effect level
HQ = Hazard quotient

Table 14. Summary of Hazard Quotient Calculation Results
Upper Green Bay Portion of the Fox River Site
Green Bay, WI
February 2000

ASSESSMENT ENDPOINT	LINES OF EVIDENCE	NOAEL HQ	LOAEL HQ	PREDICTED RISK	SECTION NUMBER
Pelagic Fish Reproduction and Survival	Egg Concentration				
	NRDA data mean	4.2	0.4	Potential	6.1.1
	NRDA data maximum	9.5	0.9	Potential	6.1.1
	Overall mean	3.9	0.4	Potential	6.1.1
	Overall maximum	9.5	0.9	Potential	6.1.1
	Adult Tissue Concentration				
	NRDA data mean	4.2	0.4	Potential	6.1.2
	NRDA data maximum	9.4	0.9	Potential	6.1.2
	Overall mean	4	0.4	Potential	6.1.2
	Overall maximum	9.4	0.9	Potential	6.1.2
Piscivorous Bird Reproduction and Survival	Egg Concentration (TRV = 4.7, 7.6)				
	Caspian tern	3.4	2.1	Yes	6.2.1
	Double-crested cormorant				
	Mean	2.2	1.4	Yes	6.2.1
	Maximum	4.3	2.6	Yes	6.2.1
	Egg Concentration (TRV = 0.8, 8.0)				
	Caspian tern	19.8	1.9	Yes	6.2.1
	Double-crested cormorant				
	Mean	13	1.3	Yes	6.2.1
	Maximum	25.1	2.5	Yes	6.2.1
	Food Chain Model				
	Caspian tern				
	NRDA data mean	2	0.2	Potential	6.2.2.1
	NRDA data maximum	4.6	0.5	Potential	6.2.2.1
	Overall mean	0.8	0.1	No	6.2.2.1
	Overall maximum	2.6	0.3	Potential	6.2.2.1
	Double-crested cormorant				
	NRDA data mean	7.2	0.7	Potential	6.2.2.2
	NRDA data maximum	16.2	1.6	Yes	6.2.2.2
	Overall mean	2.9	0.3	Potential	6.2.2.2
	Overall maximum	9.4	0.9	Potential	6.2.2.2
	Field Studies				
	Caspian tern	NA	NA	Not Conclusive	6.2.3.1
	Double-crested cormorant	NA	NA	Yes	6.2.3.2
Piscivorous Mammal Reproduction and Survival	Food Chain Model				
	Mink				
	NRDA data mean	177.3	5.3	Yes	6.3.1
	NRDA data maximum	397.8	11.9	Yes	6.3.1
	Overall mean	70.2	2.1	Yes	6.3.1
	Overall maximum	230.3	6.9	Yes	6.3.1

HQ = Hazard quotient
NRDA = Natural Resource Damage Assessment Data Set
NA = Data not applicable to hazard quotient method

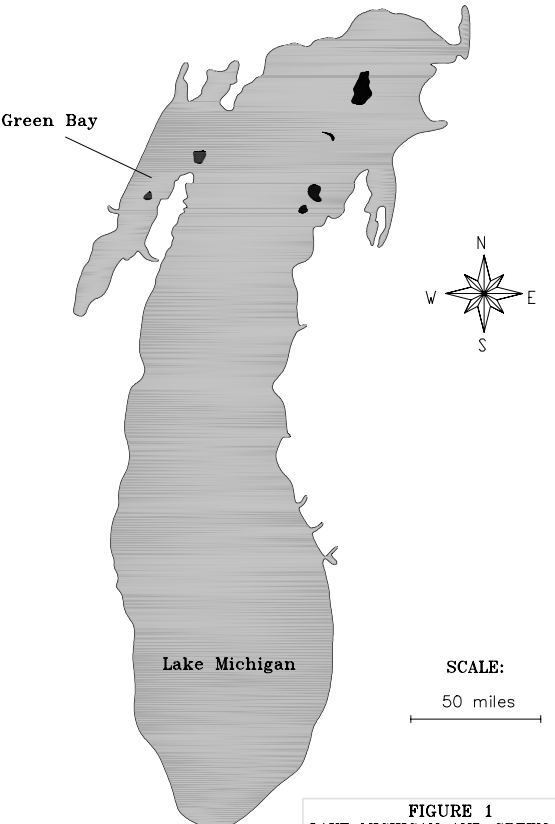


FIGURE 1
LAKE MICHIGAN AND GREEN BAY
UPPER GREEN BAY SITE
GREEN BAY, WI
FEBRUARY 2000

U.S. EPA ENVIRONMENTAL RESPONSE TEAM CENTER
RESPONSE ENGINEERING AND ANALYTICAL CONTRACT
68-C99-223
W.O.# R1A00019

019/FIGURE_1.DWG 10/21/99

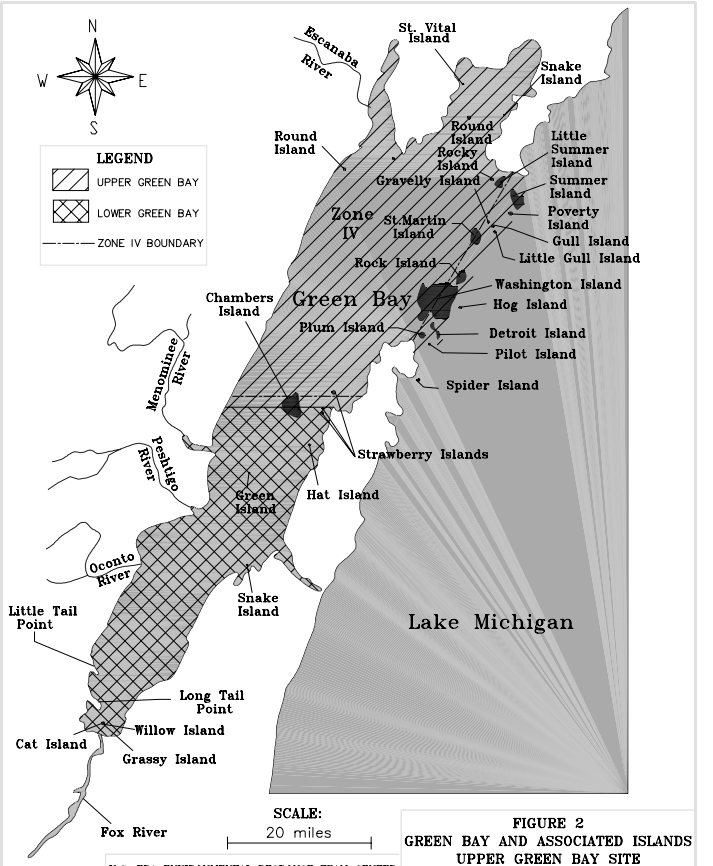


FIGURE 2
GREEN BAY AND ASSOCIATED ISLANDS
UPPER GREEN BAY SITE
GREEN BAY, WI
FEBRUARY 2000

019/figure-2.dwg 10/21/99

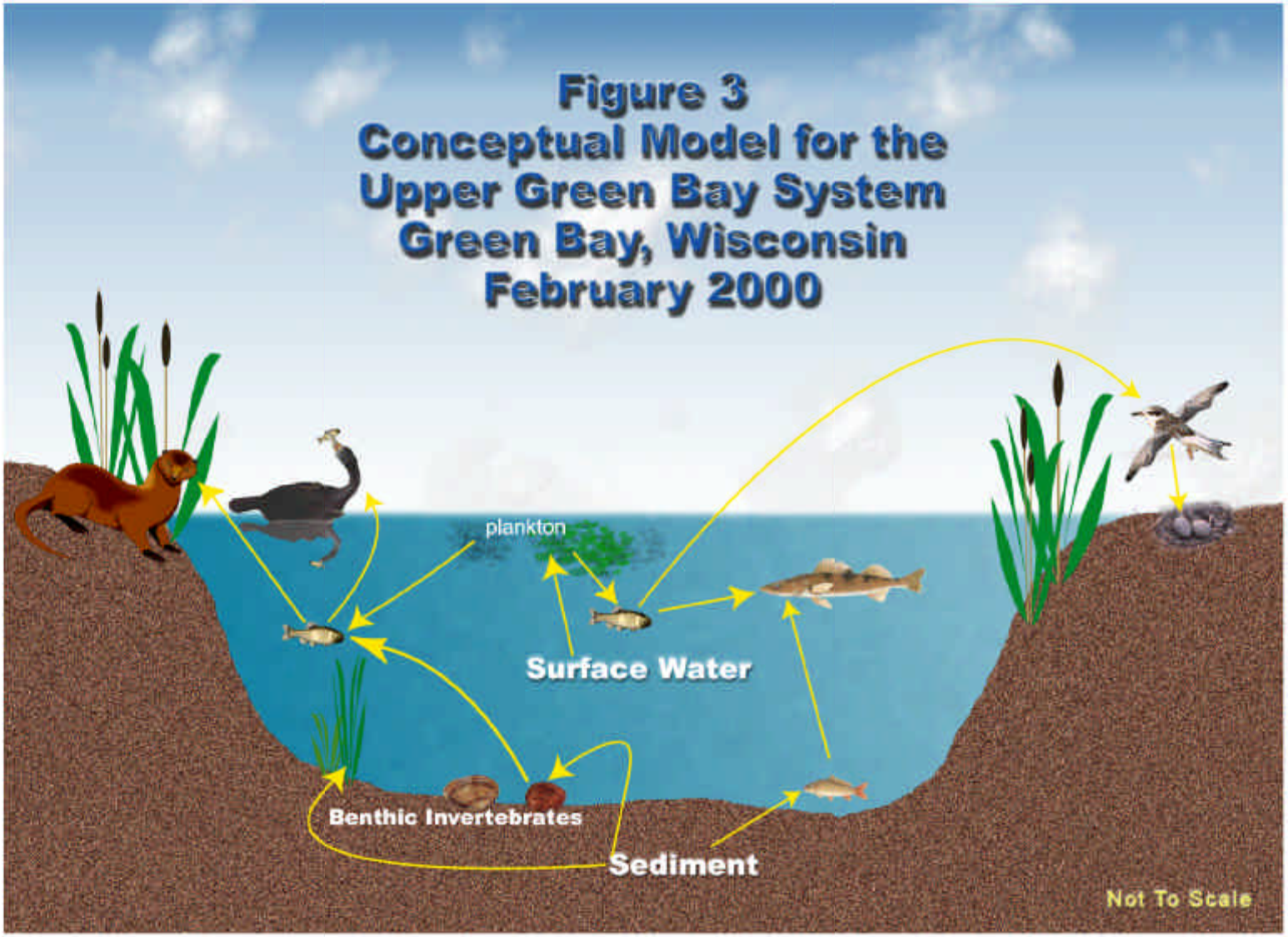


Figure 3
Conceptual Model for the
Upper Green Bay System
Green Bay, Wisconsin
February 2000
plankton
Surface Water
Benthic Invertebrates
Sediment
Not To Scale

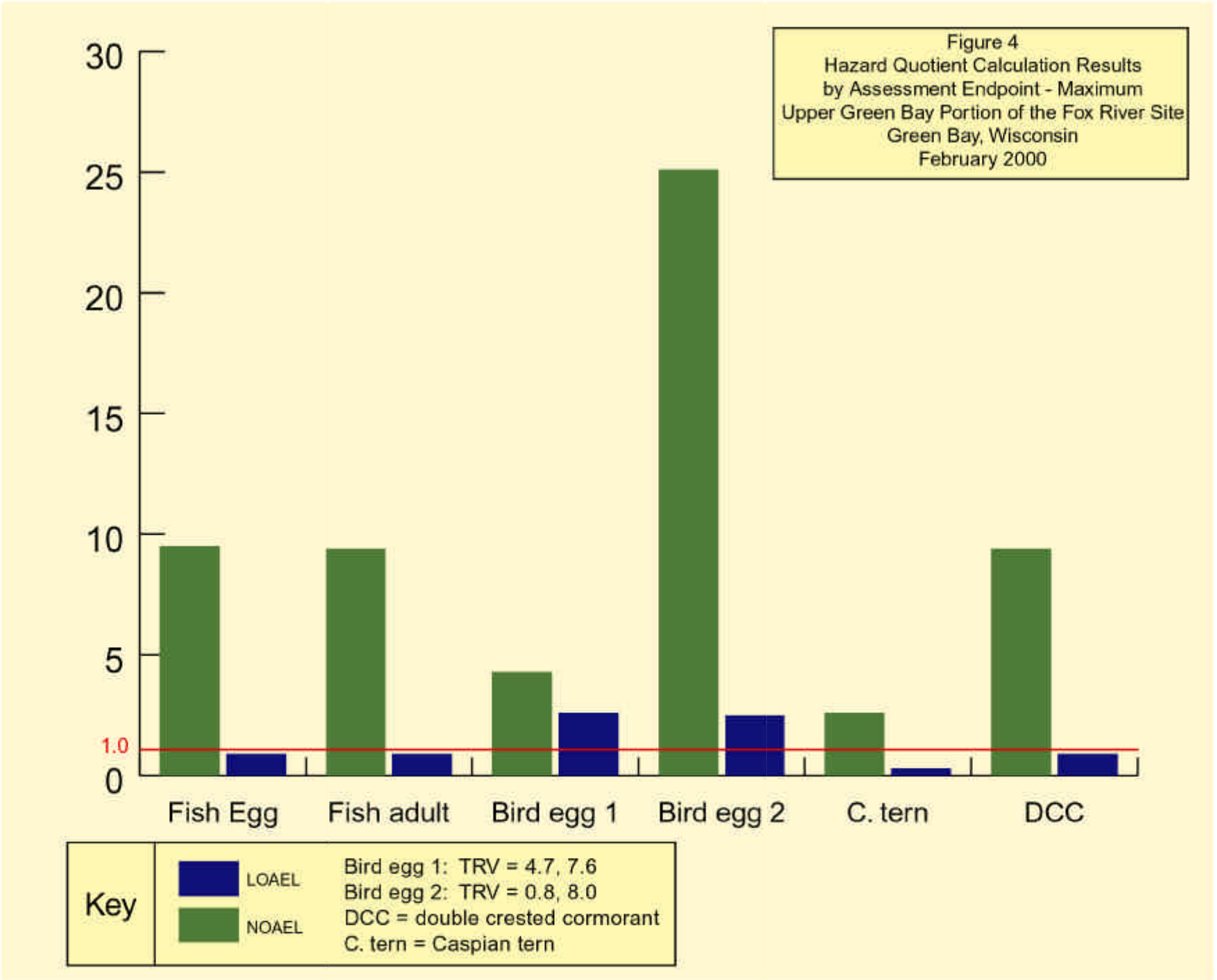
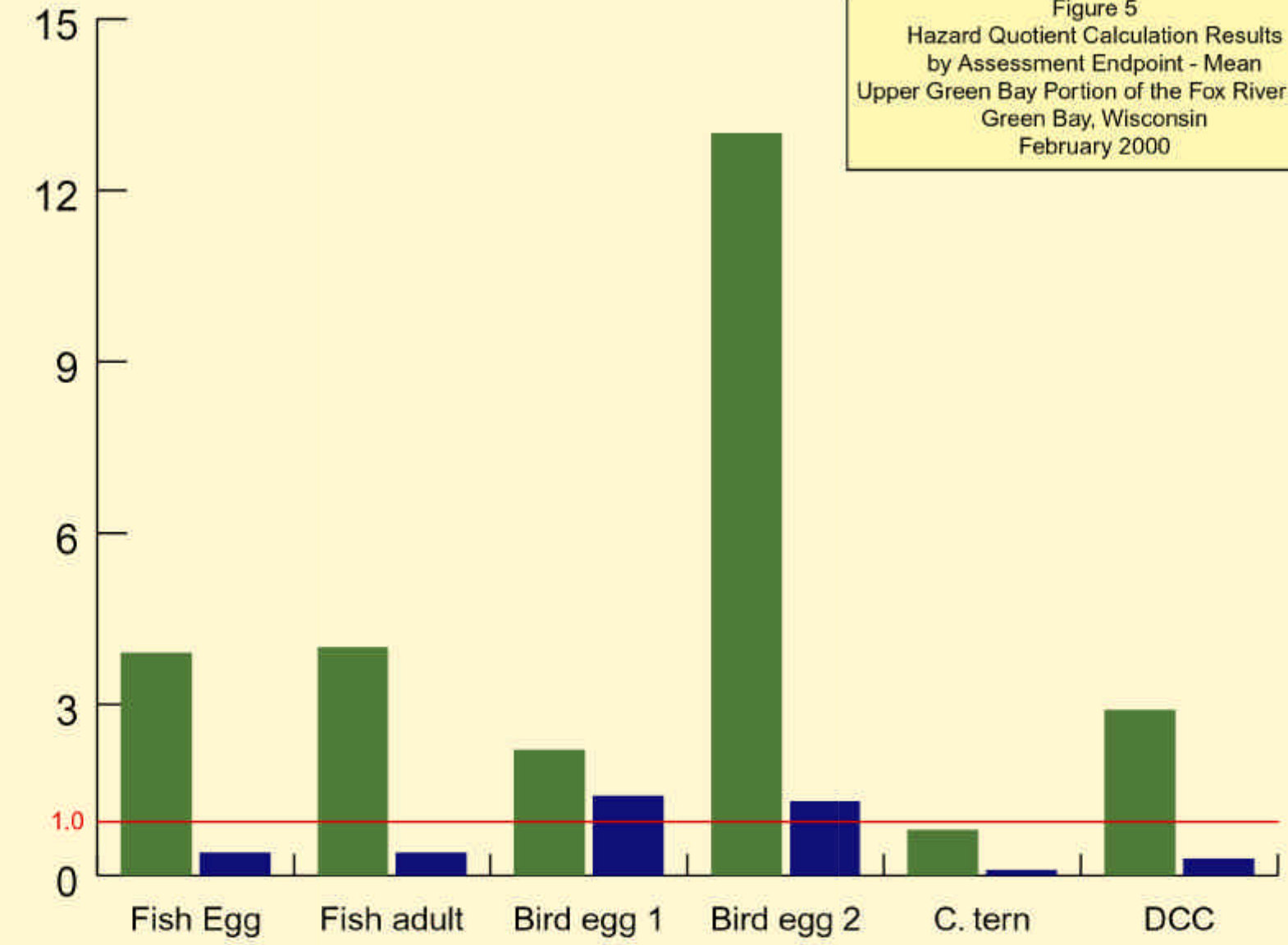


Figure 4
Hazard Quotient Calculation Results
by Assessment Endpoint - Maximum
Upper Green Bay Portion of the Fox River Site
Green Bay, Wisconsin
February 2000
30
25
20
15
10
5
1.0
0
Fish Egg
Fish adult
Bird egg 1
Bird egg 2
C. tern
DCC
Key
LOAEL
NOAEL
Bird egg 1: TRV = 4.7, 7.6
Bird egg 2: TRV = 0.8, 8.0
DCC = double crested cormorant
C. tern = Caspian tern

Figure 5
Hazard Quotient Calculation Results
by Assessment Endpoint - Mean
Upper Green Bay Portion of the Fox River Site
Green Bay, Wisconsin
February 2000



Key

LOAEL

NOAEL

Bird egg 1: TRV = 4.7, 7.6
Bird egg 2: TRV = 0.8, 8.0
DCC = double crested cormorant
C. tern = Caspian tern

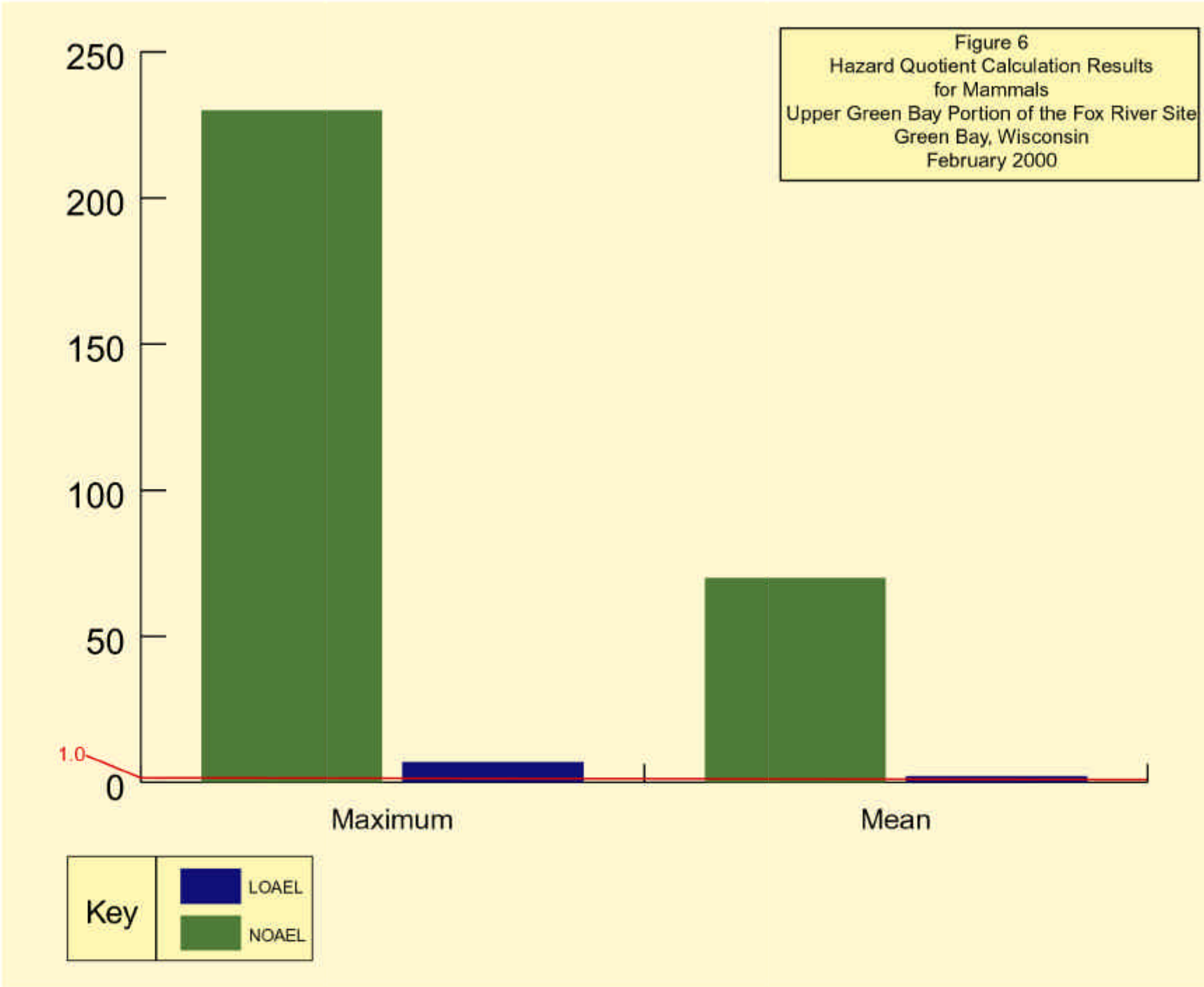


Figure 6
Hazard Quotient Calculation Results
for Mammals
Upper Green Bay Portion of the Fox River Site
Green Bay, Wisconsin
February 2000
250
200
150
100
50
1.0
0
Maximum
Mean
Key
LOAEL
NOAEL

APPENDIX A

Toxicity Reference Values
Upper Green Bay Portion of the Fox River Site
Green Bay, WI
February 2000

APPENDIX A

TOXICITY REFERENCE VALUES

A.1 Derivation of Toxicity Reference Values (TRVs)

A toxicity reference value (TRV) is a contaminant dose level that is compared with a predicted exposure dose level, calculated based on site-specific data, in order to assess the presence and degree of risk to a receptor or group of receptors from that contaminant. A TRV is based on data from laboratory toxicological evaluations. Usually, two TRVs are used in order to predict ecological risk, a no observable adverse effect level (NOAEL) and a lowest observable adverse effect level (LOAEL). The NOAEL is the highest dose at which adverse effects are not expected to occur, and the LOAEL is the lowest dose at which adverse effects are expected to occur.

In order to derive TRVs, a comprehensive literature search was performed in which studies on the toxicity of PCBs to ecological receptors were located. A variety of databases were available to be searched for literature references containing toxicological information. Some of these literature sources included Biological Abstracts, Applied Ecology Abstracts, Chemical Abstract Services, Medline, Toxline, BIOSIS, ENVIROLINE, Current Contents, Hazardous Substances Data Bank (HSDB), Registry of Toxic Effects of Chemical Substances (RTECS), Integrated Risk Information System (IRIS), and the Aquatic Information Retrieval Database (ACQUIRE).

In addition, a number of secondary literature sources provided summaries or reviews of the toxicological literature related to a variety of contaminants. These documents were not used directly to derive TRVs because they do not capture the details of the toxicological methods which are imperative to the selection of technically defensible TRVs. However, these summary documents provided an excellent source of original studies that may have been overlooked in the database searches. Examples of such summary documents include Agency for Toxic Substances Disease Registry (ATSDR) documents, U.S. Fish and Wildlife Service Contaminant Hazard Reviews, U.S. EPA Great Lakes Water Quality Initiative documents, and U.S. EPA Ambient Water Quality Criteria documents.

Studies that were obviously not useful or appropriate for deriving a TRV were eliminated. A number of criteria were considered when evaluating the appropriateness of using a particular study for deriving a TRV. The most important consideration was the suitability of the test result for evaluating the assessment endpoint. A number of additional criteria were also considered. For example, studies were selected in which the test organism was in as similar a taxonomic grouping as possible to the measurement endpoint species. Doses had to be quantified and effects measured and reported. The exposure duration was preferably either chronic, sub-chronic, or involved a sensitive life stage, and multigenerational studies were also deemed appropriate. For laboratory studies, the likelihood that a similar result would be obtained if the test were repeated was an additional consideration. Sample sizes had to be adequate and the treatment groups must have been compared to appropriate control groups. At the very least, a negative control should have been included in the study design. In addition, the measured endpoints of the study had to be ecologically relevant. For the purposes of deriving a TRV for an ecological risk assessment, an ecologically relevant endpoint is one which is closely tied to the survival of a population in the field. Usually, the endpoints that are measured for this purpose are survival, growth, and reproduction. In addition, appropriate statistical analyses must have been performed and the statistical significance reported. Finally, the study design preferably included at least three treatments in addition to any controls which may have been selected.

The selected TRVs were based preferably on high-quality studies which satisfy many or all of the requirements above. From these high quality studies, the lowest concentration that was associated with

adverse ecological effects on the test organism was selected as the LOAEL. Studies which reported both a LOAEL and NOAEL were selected over studies which reported only one effect level, due to the uncertainty associated with an unbounded effect level. If a LOAEL could not be located for a receptor, the highest concentration that was associated with no adverse effects was selected as the NOAEL. If only a LOAEL or a NOAEL could be identified from the studies, an uncertainty factor of 10 was used to convert from one to the other (U.S. EPA 1989; Sample et al. 1996; Amdur et al. 1996). Professional judgement was used in some cases to select the most appropriate TRV.

The studies which were used to derive toxicity reference values for this risk assessment are described below. In addition, these studies are also summarized in Table A1.

A.2 Toxicity of PCBs to Fish

A.2.1 Toxicity of PCBs in Fish Eggs

A number of studies indicate that the early life stages of fish are the most sensitive to PCB toxicity and that PCBs are transferred from maternal tissue to eggs (Ankley et al. 1991; Newsted et al. 1995; Larsson et al. 1993). Lake trout eggs have been shown to be particularly sensitive to PCB toxicity (Mac et al. 1985; Mac 1988; Zabel et al. 1995). Therefore, a literature review was conducted to determine toxicity reference values for PCBs in fish eggs. Ankley et al. (1991) collected 10 female Lake Michigan chinook salmon, sampled their eggs, and measured hatching success and fry survival to swim-up. Total PCBs in the eggs were negatively correlated with hatching success. Concentrations of approximately 3.7 and 4.2 mg/kg ww in the egg were identified as the NOAEC and LOAEC, respectively. Mac and Schwartz (1992) found a decrease in hatching of eggs from lake trout collected from the Great Lakes at a PCB concentration of approximately 3 mg/kg,ww, and observed no effects at an egg concentration of approximately 2.8 mg/kg, ww. When 2-year old female rainbow trout were exposed to Aroclor 1254 in the diet for two months and then spawned, fry growth was decreased at a corresponding egg concentration of 1.6 mg/kg, ww (Hendricks et al. 1981). In another study, rainbow trout eggs containing 2.7 mg/kg PCBs, ww, exhibited 75% mortality, and 60 to 70% had deformities after 30 days posthatch (Hogan and Brauhn 1975). Mac and Edsall (1991) collected lake trout from southeastern Lake Michigan, reared them in the laboratory, and measured egg hatchability and fry survival. They found a significant decrease in hatchability and fry survival in eggs with a concentration of 0.314 mg/kg total PCBs. No adverse effects on hatchability and fry survival were noted in eggs with a concentration of total PCBs of 0.173 mg/kg, ww.

Studies conducted using fish collected from the Great Lakes were not utilized to derive TRVs for PCBs in this risk assessment due to the presence of measurable concentrations of other contaminants due to their exposure in Lake Michigan. The study by Hendricks et al. (1981) was used to derive the fish egg toxicity reference values for this risk assessment. This is because of the low LOAEC observed in this study, and the fact that the test species used in this study (rainbow trout) is taxonomically similar to the measurement endpoint species (lake trout). Therefore, a fish egg concentration of 1.6 mg/kg, wet weight, was used as a LOAEC to evaluate the toxicity of PCBs to fish in the upper Green Bay. This value was converted to a NOAEC of 0.16 mg/kg, wet weight, using an accepted conversion factor of 10.

A.2.2 Toxicity of PCBs in Fish Whole Body Tissues

A variety of additional studies have been performed on fish in which reproductive endpoints have been adversely affected and whole body concentrations of PCBs were measured. Lethal body burden concentrations have been estimated at greater than 100 mg/kg for young fish and greater than 250 mg/kg for older fish (Niimi 1996). When fathead minnows were exposed to Aroclor 1254 at 1.8 ug/L, spawning was reduced. Corresponding male and female mean tissue

concentrations were 196 and 429 mg/kg PCBs, respectively (Nebeker et al. 1974). Freeman and Idler (1975) exposed brook trout to 0.2 mg/L Aroclor 1254 in water, and exposed the resulting eggs to either control water or water containing 0.2 mg/L Aroclor 1254. They found that egg hatch was only 78% (compared to 100% in the control) when the eggs were exposed to control water. When the eggs were exposed to water containing Aroclor 1254, none of the eggs hatched. The corresponding adult muscle tissue contained 32.8 mg/kg PCBs. In another study, when fingerling channel catfish were exposed to four Aroclors in the diet for 193 days, no effects on growth were observed, and PCB tissue concentrations were 14 to 32 mg/kg. In the same study, growth in salmon was not affected after exposure to Aroclor 1254 in the diet for 260 days. The salmon tissue concentrations were from 0.4 to 645 mg/kg (Mayer et al. 1977). In another study, adult fathead minnows were exposed for 16 weeks in aquaria containing a 2 to 4 cm layer of sediment contaminated with three different concentrations of PCBs, and tissue PCB concentrations were measured at 7 and 16 weeks. Reproduction was significantly less than the controls in fish exposed to the two highest concentrations. Corresponding mean tissue PCB concentrations ranged from 13.7 to 47.2 mg/kg, wet weight (wet weight). No significant adverse effects were noted in fish exposed to the lowest concentration, corresponding to tissue concentrations ranging from 5.25 mg/kg, wet weight, at 7 weeks to 11.6 mg/kg, wet weight, at 16 weeks (U.S. ACOE 1988). When Mayer et al. (1985) exposed rainbow trout to 2.9 ug/L of an Aroclor mixture (1:2 ratio of 1254:1260) for 90 days, growth was reduced by ten percent. The corresponding PCB tissue concentration was 120 mg/kg. In fish exposed to 0.2 to 5 ug/L of the Aroclor mixture, growth was not affected, and fish tissue concentrations were 6 to 70 mg/kg. Hansen et al. (1976) exposed catfish to 20 mg/kg Aroclor in the diet for 140 days, after which PCB administration was suspended for 56 days, followed by another 56 days with 20 mg/kg PCBs in the diet again. By day 130, growth rates in the PCB-fed fish were significantly lower than those in the control. However, from day 140 to day 252, during which PCBs were fed only during the last 56 days, the growth rate of the PCB-fed fish was greater than that in the controls. By the end of the study, the mean whole body fish concentration in the treated group was 10.86 mg/kg PCBs. When Aroclor 1254 was fed to trout at 15 mg/kg in the diet for 224 days, growth and liver histology were not affected at corresponding tissue concentrations of 8 mg/kg PCBs (Lieb et al. 1974). When brook trout were exposed to 3.1 to 13 ug/L Aroclor 1248 for 118 days, 21-100% mortality was observed, and concentrations of PCBs in dead fry were greater than 125 mg/kg (Mauck et al. 1978). When cyprinid minnows were exposed to Clophen A50 in the diet, premature hatching and death of fry were observed, with corresponding whole body concentrations of 15 and 170 mg/kg, wet weight. No significant adverse effects were noted in fish with corresponding whole body concentrations of 1.6 mg/kg, wet weight (Bengtsson 1980). Mac et al. (1993) found a correlation between embryo mortality and PCB concentrations in lake trout whole body tissues at concentrations ranging from approximately 3 to 14 mg/kg, wet weight. However, since the lake trout in this study were not compared to appropriate controls, a NOAEC and a LOAEC could not be determined from this study.

Another method to determine whole body concentrations at which adverse effects would be expected is to estimate a whole body concentration based on an egg concentration that is associated with adverse effects. This method was derived based on the fact that whole body concentrations are often available, while fish concentrations are not. Early life stages are most sensitive to adverse effects of PCBs, therefore it is important to identify maternal whole-body concentrations that result in critical egg/fry PCB concentrations. In a study by Mac et al. (1993), lake trout whole body and egg concentrations of PCBs were measured in seven lake trout collected from various Great Lakes. When the egg PCB concentrations (wet weight) were divided by the whole body PCB concentrations (wet weight), a mean ratio of 0.209 was calculated. Using this ratio, one can calculate an expected lake trout whole body concentration based on a lake trout egg concentration. Therefore, a whole body concentration that would be expected to elicit adverse effects can be calculated from an egg concentration that has been shown to elicit adverse effects. When the egg LOAEC concentration of 1.6 mg/kg, wet weight, derived above (Section A.2.1), is

divided by 0.209, the resulting whole body concentration is 7.7 mg/kg, wet weight.

Since the latter method provided the lowest LOAEC for whole body fish PCB concentrations, a LOAEC of 7.7 mg/kg, wet weight was selected to evaluate the effects of PCBs on fish survival and reproduction in the upper Green Bay using whole body concentrations. This LOAEC was converted to a NOAEC of 0.77 mg/kg, wet weight, in whole body fish tissue using an accepted conversion factor of 10.

A.3 Toxicity of PCBs to Birds

There is a great degree of variability among different bird species in response to PCBs. In sensitive species, normal patterns of growth, behavior, reproduction, and metabolism may be altered. Liver concentrations of PCBs are generally highest in piscivorous birds, followed by birds that feed on other small birds and mammals, birds that feed on worms and insects, and herbivorous or seed eating birds, respectively (NAS 1979).

A.3.1 Dietary Toxicity of PCBs to Birds

No studies were found in which the toxicity of PCBs to either of the two measurement endpoint species (Forster's tern and double-crested cormorant) was examined. Therefore, literature pertaining to the toxicity of PCBs to other bird species was reviewed and is summarized below. It should be noted that due to the fact that the test species used in the studies summarized below are different from the measurement endpoint species, the dosages calculated in these studies had to be normalized to account for differences in food ingestion rates and body weights between the test species and the measurement endpoint species. To do this, the concentrations of PCBs in food reported in the literature were multiplied by the food ingestion rate and divided by the body weight of the test species. If the food ingestion rate and/or the body weight of the test organisms were not reported in the study, then a food ingestion and/or body weight reported elsewhere in the literature was used. If this information was not available elsewhere in the literature, then body weights were obtained from Dunning (1993) and converted into food ingestion rates using an allometric equation developed by Nagy (1987).

A dietary concentration of 1500 mg/kg (dry weight) was administered to red-winged blackbirds for six days, by which time 50 percent of the birds had died (Stickel et al. 1984). Due to the acute nature of this study (short duration and high mortality), it was not used to assess the chronic effects of PCBs to birds in this risk assessment. In another study, robins, *Erithacus rubecula*, fed a diet containing 5 mg Clophen A50 per day for a period of 11 to 13 days displayed abnormal nocturnal behavior and activity patterns compared to control birds (Ulfstrand and Sondergrund 1971). The average body weight of this robin is reported to be 18.2 grams (Dunning 1993). Subsequently, the daily dose would equal 275 mg Clophen A50/kg/day.

Mallard ducklings, over 9 weeks of age, were fed a PCB-treated diet for 5 days, followed by 3 days of an untreated diet. The 8-day LC50s ranged from 1,975 mg/kg for Aroclor 1260 to 3,182 mg/kg for Aroclor 1242 (Heath et al. 1972). The lowest LC50 value was converted to a LOAEL of 197.5 mg/kg using an accepted conversion factor of 10. In order to express this value in units of mg/kg BW/day, 197.5 mg/kg was multiplied by a food ingestion rate of 0.15 kg/day and the inverse of the lowest reported body weight of 1 kg, both reported for juvenile mallard ducks (Szaro et al. 1981). This yielded an exposure concentration of 29.63 mg/kg BW/day. In another study, a dietary concentration of 150 mg/kg Aroclor 1242 resulted in egg shell thinning of 8.9% in mallard ducks (Haseltine and Prouty 1980). To convert this dosage to units of mg/kg BW/day, the dose was first multiplied by the food ingestion rate for the mallard duck of 0.25 kg/day (Newell et al. 1987), and then divided by the lowest reported adult body weight of 1.043 kg (U.S. EPA 1993) to yield a dose of approximately 36 mg/kg BW/day.

When Aroclor 1254 was fed to 9 month-old mallard hens at a concentration of 25 mg/kg, dry weight, in the diet for at least one month prior to egg laying, no detrimental effects on reproduction or nest attentiveness were observed (Custer and Heinz 1980). Assuming that the diet was one-third solids, this equates to a wet weight concentration of approximately 8.3 mg/kg. To convert this dosage to units of mg/kg BW/day, the dose was first multiplied by the food ingestion rate for the mallard duck of 0.25 kg/day (Newell et al. 1987), and then divided by the lowest reported adult body weight of 1.043 kg (U.S. EPA 1993) to yield a dose of approximately 2.0 mg/kg BW/day.

When screech owls were fed Arclor 1248 in their diet at a concentration of 3 mg/kg for two breeding seasons, the number of eggs per clutch, hatchability, chick malformations, survival, and eggshell thickness were not affected (McLane and Hughes 1980). To convert to units of mg/kg BW/day, this value was divided by the reported mean body weight of 0.185 kg for screech owls (Dunning 1993) and multiplied by a food ingestion rate of 0.019 kg/day that was calculated using an allometric equation (Nagy 1987). This resulted in a dietary dosage of 0.3 mg/kg BW/day.

Nestling white pelicans captured from the wild received 100 mg of Aroclor 1254 as daily oral doses for 10 weeks in addition to a controlled diet. Following the 10 week exposure period, the birds were stressed for an additional 2 weeks by reducing their food consumption in half. The initial mean body weight of the birds prior to the treatment was 6.2 kg. The mean body weight at the end of the 12 week experimental period was 4.8 kg. Micrograph examination of the livers from the birds in the treatment group indicated a 22 percent increase in hepatocyte size, a significant 25 percent increase in the number of mitochondria, a significant 20 percent fewer cristae per mitochondria, and a 22 percent increase in the number of lysosomes, microbodies, and other membrane-bounded vacuoles (Stotz and Greichus 1978). For this risk assessment, the dose (100 mg/day) was multiplied by the inverse of the lower mean body weight (from the end of the experimental period) to yield an exposure concentration of 20.8 mg/kg BW/day.

Peakall and Peakall (1973) maintained ring doves on a diet that contained 10 mg/kg Aroclor 1254. They found that reproductive success was dependent on exposure of the female to the PCB compound. Females fed PCB-spiked food were less attentive to their nest and had erratic nesting behaviors which interfered with egg development. Artificial incubation greatly increased the breeding success for these birds. The food concentration of 10 mg/kg was converted to 1.12 mg Aroclor 1254 /kg/day in chicken feed using 11.2 gm/day as the ingestion rate, and 100 grams as a body mass estimate (data based on mourning dove; Kenaga 1973). Similar values were obtained by Peakall et al. (1972) for the ringed turtle dove, in which a dietary Aroclor 1254 concentration of 10 mg/kg adversely affected hatching success due to heavy embryonic mortality .
Another study investigated the behavioral component of reproduction in mourning doves given dietary supplements of 0, 10, or 40 mg/kg Aroclor 1254 (Tori and Peterle 1983). Using the ingestion rate and body weight specified previously (Kenaga 1973), these doses correspond to 0, 1.12 mg/kg BW/day, and 4.48 mg/kg BW/day. Control doves displayed normal courtship behaviors and patterns. Doves that were fed at the 10 ppm (1.12 mg/kg BW/day) level spent twice as much time in the courtship phase as the control birds, with only 50% completing courtship and nesting. Of the 50% that did nest and incubate eggs, nest initiation was significantly delayed, resulting in a delay in egg laying as well. None of the doves on the 40 ppm dietary supplement completed the nesting process (Tori and Peterle 1983). It was hypothesized that the decline of reproductive activity was induced by the degradation of estrogen and androgen present in the birds which is presumably a result of increased hepatic microsomal enzyme activity due to the presence of PCBs (Tori and Peterle 1983).

Hatchability of chicken eggs was reduced in hens fed a diet which was supplemented with 20 mg/kg of total PCBs; reproductive impairment was observed at supplemental dietary levels as low as 5 mg/kg (Heinz et al. 1984). The lower dose was converted to 0.9 mg/kg BW/day using a reported body weight of 0.8 kg and an ingestion rate of 0.14 kg/day for adult chickens (RTECS

1986). When Lillie et al. (1975) exposed chickens to diets containing either Aroclor 1016, 1232, 1242, 1248, or 1254 for 8 weeks, hatching success was significantly reduced at a concentration as low as 10 mg/kg (for Aroclor 1232 and Aroclor 1242), while no effects were noted at a concentration of 5 mg/kg. These values were converted to 1.75 and 0.875 mg/kg BW/day, respectively, using the reported body weight and ingestion rate for chickens indicated above. Similar results were described in Britton and Huston (1973), in which eggs from chickens fed diets containing 10 mg/kg Aroclor 1242 also exhibited reduced hatching success. Again, no effects were observed at a dietary concentration of 5 mg/kg. Similar results were also obtained by Scott (1977), in which hatching success was also decreased in chickens fed a diet containing 10 mg/kg Aroclor 1248. In this study, no effects were observed at 1 mg/kg. The value of 1 mg/kg was converted to 0.175 mg/kg BW/day using a reported body weight of 0.8 kg and an ingestion rate of 0.14 kg/day for adult chickens (RTECS 1986). When Platanow and Reinhart (1973) exposed chickens to Aroclor 1254 in the diet, a concentration of 5 mg/kg resulted in a decrease in both egg production and female fertility. This concentration was converted to a dietary dosage of 0.875 mg/kg BW/day using the reported body weight and ingestion rate indicated above. Finally, when Lillie et al. (1974) exposed chickens to diets containing either Aroclor 1221, 1232, 1242, 1248, 1254, or 1268, chick growth was significantly reduced at a concentration as low as 2 mg/kg (for Aroclors 1248 and 1254). To convert this concentration to units of mg/kg BW/day, the body weight and ingestion rate indicated above were used, yielding a dietary dosage of 0.35 mg/kg BW/day.

Yearling male American kestrels were fed prey items (day-old cockerels) containing approximately 33 mg/kg, wet weight, of Aroclor 1254 for 62 to 69 days. This dose was converted by the investigators to a daily exposure concentration of 9 to 10 mg/kg BW/day. Kestrels receiving the treated diet exhibited a significant 22 to 27 percent reduction in sperm concentrations. This response was associated with a muscle PCB concentration of 107 mg/kg, lipid normalized, and a testes concentration of 128 mg/kg, lipid normalized (Bird et al. 1983).

Male and female pairs of American kestrels were fed diets containing 3 mg/kg, wet weight, of Aroclor 1248 incorporated into a commercial diet for approximately 20 weeks. Eggs were collected from the pairs 2 to 4 days after egg-laying was complete. The eggs collected from the treated pairs of birds exhibited a significant 5 percent reduction in eggshell thickness. This response was associated with a parent muscle tissue PCB concentration of 18.5$\pm$5.1 mg/kg, wet weight (Lowe and Stendell 1991). Neither the body weights nor the food ingestion rates were reported in this study; therefore, values from a different study were used to convert the 3 mg/kg dose into an exposure concentration to be used in this risk assessment. The 3 mg/kg dose was multiplied by the inverse of an adult American kestrel body weight of 0.200 kg and a food ingestion rate of 0.0154 kg/day (Nice 1938) to yield an exposure concentration of 0.231 mg/kg BW/day. However, a more recent summary paper by Peakall and Lincer (1996) indicates that PCBs do not cause eggshell thinning except at very high doses that are likely to cause other reproductive toxicological effects as well. Therefore, the LOAEL based on the Lowe and Stendall (1991) study was not used in this risk assessment to evaluate the dietary toxicity of PCBs in birds.

Summer et al. (1996a) exposed white Leghorn hens for eight weeks with commercial diets mixed with contaminated carp from Saginaw Bay, Lake Huron. The concentrations of PCBs in the resulting diets, measured as the sum of Aroclors 1242, 1248, 1254, and 1260, were 0.3 mg/kg (control), 0.8 mg/kg, and 6.6 mg/kg, wet weight. Hens were artificially inseminated weekly, and food consumption, body weights, and egg production were monitored daily. Food consumption initially declined in all the treatment groups but was greatest in the high dose group by the end of the study. Body weights were greater in the control and the low dose groups by the end of the study. Finally, egg production initially decreased during the acclimation period prior to the study, but egg production in the high dose group returned to pre-trial levels by the end of the study while egg production in the control and the low dose group remained significantly lower. The decreased

egg production, as well as the increased body weights, in the control and the low dose group were explained by the authors as effects of fatty liver hemorrhagic syndrome (FLHS), with which the necropsy results were consistent. It was hypothesized that the PCBs in the high dose group provided a protective mechanism against FLHS, thus resulting in the higher egg production, since this protective mechanism had been observed in other studies. In a second phase of this experiment (Summer et al. 1996b), eggs were allowed to develop through day 25 of incubation, and hatching and deformity rates were observed and noted. Rates of deformities correlated with concentrations of PCBs in food, and both treatments (0.8 and 6.6 mg/kg, wet weight, in the diet) produced significantly higher rates of deformities (24% and 40%, respectively) compared to the control (17%). To convert the lower PCB treatment concentration (0.8 mg/kg, wet weight) to units of mg/kg BW/day, the average daily PCB consumption of hens in this treatment group reported by the authors (Summer et al. 1996a) for the 8-week duration of the study (67.1 ug/day) was divided by the corresponding average body weight (1620 g) to obtain a dietary dosage of 0.0414 mg/kg BW/day. To convert the control PCB concentration (0.3 mg/kg, wet weight) to units of mg/kg BW/day, the average daily PCB consumption of hens in this treatment group reported by the authors (Summer et al. 1996a) for the 8-week duration of the study (26.75 ug/day) was divided by the corresponding average body weight (1690 g) to obtain a dietary dosage of 0.0158 mg/kg BW/day. Although this study provided the lowest LOAEL and NOAEL of the studies presented here, these values were not selected for use in this risk assessment because the food source for the study came from an area that is known to contain a variety of pollutants in addition to PCBs, and the contribution of these other contaminants to the effects observed in this study are unknown.

The results of the Tori and Peterle (1983), Peakall and Peakall (1973), and Peakall et al. (1972) studies were selected for use in this risk assessment due to the significance of the endpoints (reproductive success and behavior) and the specificity of the test chemical (PCBs only). Therefore, a LOAEL of 1.12 mg/kg BW/day will be used in this risk assessment to evaluate the risk from PCBs to the Forster's tern and the double-crested cormorant. A NOAEL of 0.112 mg/kg BW/day was calculated from this LOAEL using an accepted conversion factor of 10.

A.3.2 Toxicity of PCBs in Bird Eggs

A variety of field and laboratory studies have been performed in which concentrations of PCBs in bird eggs have been correlated with adverse effects on survival, growth, or reproduction. No apparent adverse reproductive effects were observed in nine colonies of great blue herons, of which the highest mean egg PCB concentration was 7.8 mg/kg, wet weight (Boily et al. 1994). Similarly, no adverse reproductive effects were observed in a field population of black-crowned night herons with mean egg PCB concentrations of up to 10.9 mg/kg, wet weight (Tremblay and Ellison 1980). Mallard ducks fed Aroclor 1254 did not exhibit any adverse effects on reproductive success or nest attentiveness at corresponding egg PCB concentrations of 23.3 mg/kg, wet weight (Custer and Heinz 1980). Haseltine and Prouty (1980) observed 8.9% egg shell thinning at a corresponding mean egg concentration of 105 mg/kg, wet weight, in mallard ducks fed 150 ppm Aroclor 1242. No effects on the number of eggs laid, eggs hatched, number of young fledged, and eggshell thickness were observed in screech owls fed 3 ppm Aroclor 1248, resulting in a mean egg PCB concentration of 7.1 mg/kg, wet weight (McLane and Hughes 1980). In bald eagles, the mean egg PCB concentration in successful nests (defined as having one or more young produced in the year of sample egg collection) was 7.2 mg/kg, wet weight, and in unsuccessful nests, the mean egg PCB concentration was 13 mg/kg, wet weight (Wiemeyer et al. 1984). Similar results were obtained for bald eagles by Wiemeyer et al. (1993), in which a significant reduction in the number of young raised were noted at a corresponding mean egg PCB concentration of 13 mg/kg, although the authors indicate that DDE may have contributed more to the decreased production than PCBs . Wiemeyer (1990) later reports that eagle egg PCB concentrations of 4.0 mg/kg should be adequate to ensure normal reproduction. These studies, however, are confounded by the presence of DDE in the eggs, and controversy exists over the contribution of DDE versus PCBs causing the

observed effects (Bosveld and Van den Berg 1994). Bosveld and Van den Berg (1994) also report adverse effects on hatching success in the Forster's tern and common tern at egg PCB concentrations of 19 mg/kg and 8 mg/kg, respectively, with a corresponding NOAEL for both bird species of 7 mg/kg (Bosveld and Van den Berg 1994).

Struger and Weseloh (1985) did not observe any adverse effects on eggshell thickness or reproductive success in caspian terns from the Great Lakes with egg PCB concentrations as high as approximately 39 mg/kg PCBs, wet weight. Based on data presented in Kubiak et al. (1989), a NOAEC and a LOAEC of 4.5 mg/kg, wet weight, and 22.2 mg/kg, wet weight, respectively, can be derived for hatching success in the Forster's tern. Hoffman et al. (1993) did not observe any apparent adverse effects in a field population of common terns with corresponding egg PCB concentrations of 4.7 mg/kg, wet weight, but a decrease in hatching success and increase in embryo deformities was observed at corresponding egg PCB concentrations of 7.6 mg/kg, wet weight. Peakall et al. (1972) observed a decrease in hatching success due to heavy embryonic mortality at a corresponding mean egg concentration of 50 mg/kg, dry weight, in turtle doves fed 10 ppm Aroclor 1254. Assuming a percent solids composition of 33% for chicken eggs, this corresponds to a wet weight concentration of approximately 16 mg/kg.

Ludwig et al. (1996) reviewed available data on concentrations of contaminants in eggs and observed deformities in embryos and chicks of Double-crested cormorants and Caspian terns. Between 1986 and 1991, hatched chicks and live and dead eggs from 37 colonies in the upper Great Lakes were evaluated annually for gross anatomical deformities. Deformity rates were higher in all Great Lakes areas evaluated (including Green Bay) than at a reference colony. Hatching and deformity rates were correlated with concentrations of planar PCBs and TCDD-EQs. PC concentrations ranged from 3.6 mg/kg in eggs collected from Lake Superior to 7.3 mg/kg in eggs collected from Green Bay; PCB concentration in eggs from the reference colony was 0.8 mg/kg. The authors concluded that the weight of evidence was sufficient to conclude there is a causal relationship between the incidence of deformities in cormorants and terns and exposure to planar halogenated compounds measured as TCDD-EQs or total PCBs in the Great Lakes.

Tillitt et al. (1992) monitored 11 double-crested cormorant colonies around the Great Lakes as well as a reference site outside of the Great Lakes for hatching success in 1986, 1987, and 1988. A significant correlation was found between total egg PCB concentrations and egg mortality. A NOAEC and LOAEC could not be derived from this study because 21% egg mortality was observed in a colony whose mean egg PCB concentration was 0.1 mg/kg, wet weight, whereas the reference area exhibited 8% egg mortality with a corresponding mean egg PCB concentration of 0.8 mg/kg, wet weight. The next highest mean egg PCB concentration was 4.4 mg/kg, wet weight, for another colony, where 26% egg mortality was observed.

When Britton and Huston (1973) exposed laying hens to a dietary concentration 10 ppm Aroclor 1242 in the lab, no effects on hatching success were noted at a corresponding mean egg yolk PCB concentration of 0.95 mg/kg, wet weight, but hatching success was significantly reduced at a corresponding mean egg yolk PCB concentration of 1.5 mg/kg, wet weight. In another study, a drastic reduction in the hatchability of chicks was observed at a corresponding mean egg PCB concentration of 2.5 mg/kg, but no adverse effects on eggshell quality, egg production, or hatchability were noted at a mean egg PCB concentration of 0.36 mg/kg in chickens (Scott 1977). In another study, chickens fed 5 ppm Aroclor 1254 exhibited a significant reduction in egg production and female fertility, with a corresponding egg PCB concentration of 5 mg/kg (Platanow and Reinhart 1973). The same study states that no adverse effects were noted at egg PCB concentrations less than 5 mg/kg, wet weight.

These studies indicate that the chicken is the most sensitive species to PCB toxicity. Indeed, numerous studies have documented the greater sensitivity of chickens to TCDD-like toxicity as

compared to bird species in the wild (Eisler and Belisle 1996). Therefore, studies in which chickens were used as the test subject were not selected for derivation of the NOAEC and LOAEC in this risk assessment, since doing so would overestimate the risk posed to the bird species inhabiting the upper Green Bay.

Based on the Hoffman et al. (1993) study, a LOAEC of 7.6 mg/kg, wet weight, and a NOAEC of 4.7 mg/kg, wet weight, for PCBs in bird eggs were selected for use in the ecological risk assessment. Ludwig et al. (1996) reported a NOAEC of 0.8 mg/kg. This concentration will also be evaluated in this risk assessment for comparative purposes, however it should be noted that this is an unbounded NOAEL and it was not selected as the sole TRV for this reason.

A.4 PCB Toxicity to Mammals

A variety of PCB-induced toxic effects have been observed in mammals. Mink are particularly sensitive to dietary PCB levels (Aulerich et al. 1985; Giesy et al. 1994). Anorexia, weight loss, lethargy, enlarged livers, and intestinal discharge of blood have been noted in exposed mink (Eisler 1986). Placental and mammary transfer of PCBs have been shown to be direct routes of transfer of PCBs between mother and young. PCB exposure can lead to behavioral disorders, specifically in sleep/wake cycles, and in animals that hibernate or aestivate (Montz et al. 1982; Sanders and Kirkpatrick 1977). Negative effects of PCBs on metabolism, thyroid control, ATPase activity, oxidative phosphorylation, steroid hormone activity, immunity, and vitamin A pathways have been noted (Safe 1984; U.S.EPA 1980).

PCB toxicity in mammals is highly variable. While some PCBs are extremely toxic, and can produce death and cause reproductive failure in very low levels, others appear to produce few, if any, toxic responses (Eisler 1986). Toxic responses to PCBs are also highly species specific. Mink are highly susceptible to PCB toxicity, while closely related mammals, such as the European ferret, are more resistant (Eisler 1986). Younger mammals appear to be more susceptible to PCB poisoning than adults (Eisler 1986). Mutagenic, carcinogenic, and teratogenic effects of PCB exposure have been observed, with mutagenic activity appearing to increase with increasing chlorination of the PCB molecule (Eisler 1986).

Several studies were found pertaining to the dietary toxicity of PCBs to mink, most of which examined effects on reproduction, growth and survival. Since the mink is the measurement endpoint receptor to be evaluated in this risk assessment, these mink studies were the only studies that were reviewed to derive a TRV for piscivorous mammals.

In a preliminary study to determine the cause of reproductive complications in mink fed Great Lakes fish, adult breeder mink were fed a basal diet supplemented with 30 mg/kg of PCBs for six months (181 days). However, all of the mink died, emaciated, by the end of the experimental period (Aulerich and Ringer 1977). As a result of the preliminary study, a long-term study was conducted to ascertain the effects of long-term, low-level consumption of PCBs on growth. Mink were fed a basal diet supplemented with 5 and 10 mg/kg of PCBs for a period of approximately 8.5 months. The basal diet plus 10 mg/kg of PCBs resulted in a significant 56 percent decrease in body weight gain after a period of 4 months. Body weight gain was reduced by 39 percent in the 5 mg/kg treatment group, but this reduction was not significant. Both the 5 and 10 mg/kg treatment groups failed to produce offspring; the control group produced 17 live and 8 dead kits. Various degrees of embryotoxicity were observed during necropsy of the treated animals (Aulerich and Ringer 1977). The 5 and 10 mg/kg doses were converted to daily exposure concentrations by multiplying them by the food ingestion rate of 0.114 kg/day [calculated by multiplying the highest reported food ingestion rate for mink of 0.22 g/g BW/day (U.S. EPA 1993) by the lowest reported body weight of 520 g (Merritt 1987), and dividing by 1000] and dividing by the lowest body weight (0.923 kg) reported by the investigators for this treatment group. This yielded exposure concentrations of 1.1 and 2.2 mg/kg BW/day for the 5 and 10 mg/kg treatment groups, respectively.

Based on the results of this experiment, another experiment was conducted to determine the effects of long-

term consumption of low-level PCBs on reproduction. Fifteen mg/kg of Aroclor 1254 in the diet resulted in a complete inhibition of reproduction and 31 percent adult mortality, compared to 6 percent mortality in the controls. Five mg/kg of Aroclor 1254 resulted in a 95 percent reduction in the number of kits born live; the ratio of live kits to female adults was reduced by 87 percent. However, in an effort to determine the persistence of the impaired reproductive condition, 11 adult females that received 5 mg/kg of Aroclor 1254 for a period of six months were placed on a control diet for one year. The results indicate that the impaired reproductive performance of these females was not a permanent condition (Aulerich and Ringer 1977). The 5 and 15 mg/kg doses were converted to daily exposure dosages by multiplying them by the food ingestion rate (0.114 kg/day) for the mink and dividing them by the lowest reported body weight for the mink (0.52 kg) to yield exposure dosages of 1.1 and 3.3 mg/kg BW/day, respectively.

Eight month old mink fed a basal diet containing 1.0 mg/kg of Aroclor 1254 for a period of approximately six months exhibited no mortality or any significant changes in the thyroid, pituitary, adrenal glands, or serum T3 and T4 levels (Wren et al 1987a). Reproduction and kit development was evaluated under the same test conditions in a separate study (Wren et al. 1987b) by the same investigators. Male fertility and female offspring production were not affected by the 1.0 mg/kg Aroclor 1254 diet. However, growth rate of kits nursed by exposed mothers was significantly reduced. The investigators estimated the daily exposure concentrations to be 0.10 mg/kg BW/day for males and 0.18 mg/kg BW/day for females.

When Kubiak and Best (1991) fed mink a liver diet contaminated with PCBs , a concentration of 1.0 mg/kg PCBs resulted in reproductive impairment and a concentration of 5 mg/kg resulted in mortality. This dose was converted to a daily exposure concentration by multiplying it by the food ingestion rate of the mink (0.114 kg/day) and dividing by the lowest reported body weight of mink (0.52 kg). This yielded an exposure concentration of 0.22 mg/kg BW/day.

In another study, one-year-old mink were fed a diet of beef and cereal prepared from cows which had been given 10 consecutive daily oral doses of 1 and 10 mg/kg of Aroclor 1254 dissolved in an olive oil and dairy concentrate (Platanow and Karstad 1973). The cows did not exhibit any clinical, gross, or histopathological signs of PCB toxicity. The cows were killed 24 hours following the last dose, and the musculature, liver, and kidneys ground and mixed with commercial mink food cereal at a level of 24 percent cereal. The resulting rations containing 0.64 and 3.57 mg/kg of total PCB were fed to mink for a period of 160 days. The mink were fed this diet *ad libitum* 2 months prior to the breeding season and continued for 160 days. All 16 mink that were fed 3.57 mg/kg of PCBs died by day 105. Two of the 16 mink that were fed 0.64 mg/kg died by days 122 and 129. The mink exhibited poor appetites, lethargy, and weakness before dying. Some passed tarry feces, indicating gastrointestinal hemorrhaging. At both treatment levels, males survived longer than females. These doses were converted to daily exposure concentrations by multiplying them by the food ingestion rate of the mink (0.114 kg/day) and dividing by the lowest reported body weight of mink (0.52 kg). This yielded exposure concentrations of 0.14 and 0.78 mg/kg BW/day for the 0.64 and 3.57 mg/kg doses, respectively.

In another study, male and female ranch-bred mink were acclimated to a diet consisting of ocean fish scraps, commercial mink cereal, and meat by-products. Ocean fish scraps made up 40 percent of this diet. Dietary treatment levels were prepared by substituting 10, 20, and 40 percent of the ocean fish scraps with PCB-contaminated carp from Saginaw Bay, Lake Huron. The mean dietary PCB concentrations were 0.015 mg/kg (control), 0.72 mg/kg (10 percent carp), 1.53 mg/kg (20 percent carp), and 2.56 mg/kg (40 percent carp). Groups of 15 mink (3 males, 12 females) were assigned to one of the four treatment groups for a period of 12 weeks. Mink receiving the highest PCB-containing diet (40 percent carp or 0.32 mg/kg BW/day, as reported by the investigators) exhibited a 42 percent reduction in mean litter size, 86 percent fewer live kits at birth, and no kits surviving beyond 24-hours post-partum. Even mink receiving the 10 percent carp diet (or 0.13 mg/kg BW/day, as reported by the investigators) exhibited a 67 percent reduction in kits surviving three to six weeks relative to the control (Heaton et al. 1995).

In a related study on multigenerational effects in mink fed the same Saginaw Bay PCB-contaminated carp,

Restum et al.(1998) observed a significant reduction in kit body weights after parental exposure to 0.25 mg/kg, wet weight (0.05 mg/kg BW/day, as reported by the authors) of PCBs in fish. A significant reduction in kit survival was observed at a parental exposure concentration of 0.5 mg/kg wet weight. Of note in their study was that adverse effects on kit survival were observed even several months after the parents had been placed on the control diet. The inference was that long-term effects on mink can be observed even after short exposure periods to a PCB-contaminated diet. Some uncertainty is associated with using this study to derive the LOAEL because the mink in these studies were fed carp from Saginaw Bay, an area known to contain contaminants in addition to PCBs. However, the authors purport that the results of other studies on the effects of DDT, dieldrin, and heptachlor on mink indicate that at least these contaminants are not likely to have contributed to the toxicity observed in their study.

The LOAEL and NOAEL observed in the Heaton et al (1995) study (0.72 and 0.015 mg/kg diet) were selected as the TRVs for this risk assessment. The LOAELs cited by Heaton et al. (1995; 0.72 mg/kg) and Restum et al. (1998; 0.5 mg/kg) are effectively the same, and probably fall within the margin of error of the two studies. The daily exposure levels of 0.134 and 0.004 mg/kgBW/day reported by Heaton et al. (1995) were used in risk calculations.

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APPENDIX B

Life Histories and Exposure Profiles for the Food Chain Models
Upper Green Bay Portion of the Fox River Site
Green Bay, WI
February 2000

APPENDIX B

LIFE HISTORIES AND EXPOSURE PROFILES FOR THE FOOD CHAIN MODELS

B.1 Lake Trout (*Salvelinus namaycush*)

B.1.1 Life History

Lake trout are large, torpedo-shaped fish similar to the brook trout. Their body coloring consists of white spots on a silvery-gray background, shading to white on the belly. Lake trout range over much of the glaciated North America and are usually found near the bottom of well-oxygenated lakes. They usually occur in water about 50 degrees Fahrenheit (Smith 1985). The average lake trout in Lake Michigan weighs 7 pounds (with a range of 3 to 9 pounds) and adults range in length from 17 to 27 inches. Lake trout are longed lived and do not reach sexual maturity until 6 to 8 years old (University of Wisconsin 1998).

Lake trout spawn between September and December over rock and rubble. The eggs drop into crevices and are protected from predation in the crevices. Newly hatched lake trout feed on small zooplankton, but as they grow the diet shifts to insects and small fish. The diet of adult lake trout is 100 percent fish and may consist of chubs, sculpin, smelt, and alewife. Madenjian et al. (1998a) also found that the diet of lake trout greater than 600 millimeters total length in both the near shore and off shore waters of Lake Michigan was dominated by alewife.

Habitat deterioration causing lowered dissolved oxygen levels and reduced spawning grounds, over fishing, sea lamprey infestation, and pesticides caused a severe decline in the population of lake trout. Lake trout were once the most valuable commercial fish in the Upper Great Lakes. Lakes Erie and Ontario formerly supported a commercial fishery for this species but the native stock is now considered extinct (Smith 1985). There are many active programs to restock the population, improve habitat, reduce pesticide levels, and control sea lamprey which have increased the size of the population. Currently, there is still a commercial fishing ban for this species (Smith 1985).

B.2 Caspian Tern (*Sterna caspia*)

B.2.1 Life History

Caspian tern (*Sterna caspia*) is one of about fifty species of terns worldwide. It is 19-23" and is the largest tern in North America and the world. It is often mistaken for a gull due to its large size and gull-like characteristics. It is largely white, with a black cap, pale gray back and wings, and a heavy bright red bill and dusky underwing. Its legs and feet are black and has a slightly forked tail. In winter, the adult has white streaks on the crown. Young Caspian terns resemble their parents but they have a mottled plumage and an orange bill. A Caspian terns large sized, thicker bill, and low pitched harsh calls makes them easily distinguishable from other tern species. (WDNR 1998; Cassidy 1990; Bull and Farrand 1977; NRC 1998).

Caspian terns inhabit sandy or pebble shores of lakes and large rivers along seacoasts (Bull and Farrand 1977). The Caspian tern breeds on sea coasts, estuaries, or shores of inland lakes and seas and occasionally on rocky islands. These terns return to their breeding grounds in April, May and June (Richards 1990). They nest in colonies but may join Common tern or Ring-billed Gull colonies and they have been known to nest in isolated pairs (Environment Canada 1999). The eggs are either laid in a shallow depression in the ground or in nests lined with grasses, seaweeds, or mosses (WDNR 1998). Eggs are laid from May to July, two to three at a time and are buff

colored, blotched and spotted with dark brown. The shell lacks gloss and is rough to the touch (Richards 1990). They incubate 20-27 days, and chicks remain near the nest after hatching. The fledgling stage lasts 28-35 days and typically one young fledges from a successful nest (WDNR 1998). Caspian terns of the Great Lakes, disperse along the Atlantic Coast, in fall. They winter on the shores of the Gulf of Mexico and the Caribbean Islands (Environment Canada 1999).

Caspian terns typically plunge dive for small fish but also feed on the surface sometimes eating eggs or young of other birds. Alewives and rainbow smelt are the main prey for Caspian terns in and around the Great Lakes area but they have also been known to take Yellow Perch and Rock Bass (WDNR 1998; NRC 1998).

B.2.2 Exposure Profile for the Food Chain Model

The body weight of the Caspian tern ranges from 574 to 782 g (Dunning 1993). Data on food ingestion rates were not available, so the food ingestion rate for the Caspian tern was calculated using an allometric equation for food ingestion for birds (Nagy 1987). The food ingestion rate (FI) was calculated as 0.648*(weight in grams)$^{0.651}$ (U.S. EPA 1993). Using the lowest reported body weight of 574 g, a food ingestion rate of 40.5 g/day was calculated.

No data for water ingestion rates were available. Therefore, an allometric equation was used to calculate the water ingestion rate (WI) for Caspian terns as well. The rate was calculated in liters per day as 0.059*(weight in kilograms)$^{0.67}$ (U.S. EPA 1993). Using the lowest reported body weight of 574 g (0.574 kg), the water ingestion rate was determined to be 0.04 L/day (Nagy 1987).

An incidental sediment ingestion rate could not be located for Caspian tern. However, due to the open water feeding habits of this bird, it was assumed that the Caspian tern does not ingest sediment directly. Based on the probable feeding habits of Caspian tern prey items, it is also unlikely that the birds ingest sediment indirectly through their prey items.

A feeding radius for the Caspian tern could not be located in the literature. However, given the large size of the upper Green Bay, it was assumed that a Caspian tern could obtain 100 percent of its food from the upper Green Bay. Therefore, an area use factor of one will be assumed for this receptor.

Since Caspian terns consume fish, and given the habitat of the upper Green Bay, it will be assumed that 100 percent of the diet of the Caspian tern is comprised of fish for the purposes of the food chain model in this study.

B.3 Double-crested cormorant (*Phalacrocorax auritus*)

B.3.1 Life History

Cormorant is the common name for any of several web-footed water birds of the family Phalacrocoracidae, in the order Pelecaniformes. These fish-eating birds nest in colonies on the seacoasts of temperate and tropical regions of the world. A few species also live on large island lakes and rivers. They have slender, hooked beaks, long flexible necks, a patch of bare skin under the mouth, and a stiff tail. Their plumage is usually a glossy black, but some have white areas and many have brightly colored featherless rings around the eyes. They dive and swim deeply underwater in pursuit of fish (Environmental Advocates 1998).

The most widely distributed North American species is the double-crested cormorant, *P. auritus*, of both the Atlantic and Pacific coasts; it is the only species likely to be seen in the interior of the

continent (Environmental Advocates 1998). The double-crested cormorant is a black duck-like bird with an orange beak with a hook at the tip and orange at the jowls or cheeks. When paddling, the beak is held angled higher than parallel to water. It has an expandable throat pouch that is orange colored, and its wing span is four feet (Environmental Advocates 1998). The length of the body ranges from 74 to 89 cm. Tufts of narrow and curved black feathers found on its head during breeding season are referred to in the bird's name. Immature birds are more gray and brown (Nova Scotia Museum of Natural History 1998).

The double-crested cormorant breeds from southwestern Alaska and the interior of North America to the Gulf of St. Lawrence and southern Newfoundland, south to the southern United States and the Bahamas. Most of the birds in Atlantic Canada breed in the western Gulf of St. Lawrence and on the Atlantic coast of mainland Nova Scotia. The bird winters from the southern parts of its summer range south to Florida and the Gulf of Mexico (Nova Scotia Museum of Natural History 1998). In some of the mid-western United States cormorants are listed as endangered or threatened (SCCF 1995).

Double-crested cormorants nest in both salt and fresh water areas. In the south they nest in trees and in the north they nest in rocky ledge areas. They can be found nesting among the heron rookeries (SCCF 1995). Their nests are made from seaweed and other coarse vegetable matter placed on a rude foundation of small sticks. They usually nest in colonies, but sometimes in smaller groups, and the sites commonly chosen are of three types: on projecting shelves on the sides of steep cliffs; on level surfaces above the sea wall and preferably near its edge; and in trees 2-10 m or more in height. The trees chosen are usually on islands with low shores without cliffs and quickly die from exposure to the cormorants' excreta. The double-crested cormorant lays from three to six eggs (usually 4 to 5). The eggs are bluish white with an overlay of a chalk-like substance (Nova Scotia Museum of Natural History 1998). Both mother and father share in the child care. Babies are blind and helpless at the time of hatching. The young eat semi-digested foods from the parent’s beak. Fledging occurs at 8 weeks (SCCF 1995).

The double-crested cormorant eats almost entirely fish and for the most part species of fish not important to commercial fisheries. They chase fish underwater using both their powerful webbed feet and their wings in a sort of breast stroke to propel them through the water. Cormorants appear clumsy trying to get airborne after feeding. They generally always leave the water faced into the wind and use their feet to help them build speed for take off. Cormorants often sit on posts or wires to dry out with their wings outstretched (SCCF 1995).

B.3.2 Exposure Profile for the Food Chain Model

The body weight of an adult double-crested cormorant has been reported to be 1.9 kg (Environment Canada 1996). The double-crested cormorant has been estimated to consume approximately 25% of its body weight in fish per day (Environment Canada 1996), which equates to 0.475 kg/day. A water ingestion rate of 0.079 L/day was calculated using an allometric equation for water ingestion for birds (Nagy 1987).

An incidental sediment ingestion rate could not be located for the double-crested cormorant. However, due to the open water feeding habits of this bird, it was assumed that the double-crested cormorant does not incidentally ingest sediment.

A feeding radius for the double-crested cormorant could not be located in the literature. However, given the large size of the upper Green Bay, it was assumed that the double-crested cormorant could obtain 100 percent of its food from the upper Green Bay. Therefore, an area use factor of one will be assumed for this receptor.

Since double-crested cormorants are primarily piscivorus, it will be assumed that 100 percent of the diet of the double-crested cormorant is comprised of fish for the purposes of the food chain model in this study.

B.4 Mink (*Mustela vison*)

B.4.1 Life History

Mink are distributed over much of boreal North America, southward throughout the eastern United States and in the west to California, New Mexico, and Texas (Jones and Birney 1988). They are brown, weasel-like animals that can be found in virtually any habitat containing permanent water and are not commonly found in upland areas (Jones and Birney 1988). Although primarily nocturnal, their activity often extends into midday (Hoffmeister 1989).

Dens are always near water, and they are usually an old muskrat burrow or constructed by the mink itself (Jones and Birney 1988). Males tend to live in their own burrows which are less elaborate than ones occupied by females (Barbour and Davis 1974). Home ranges tend to be linear since mink often follow a shoreline (Jones and Birney 1988). Mink are solitary and mark their territories by spraying (Merritt 1987).

Seasonal food availability governs the dietary composition (Barbour and Davis 1974). Their diets may consist of crayfish, frogs, fish, snakes, rodents, rabbits, and plants among other items (Jones and Birney 1988; Schwartz and Schwartz 1981). Crayfish are a major portion of the summer diet in many regions of North America (Barbour and Davis 1974; Jones and Birney 1988; Merritt 1987).

Breeding occurs from January to early April with highly variable gestation periods ranging from 40 to 75 days (Merritt 1987; Schwartz and Schwartz 1981). A highly variable single litter of 1 to 17 young may be produced (Schwartz and Schwartz 1981). Average litter sizes vary among regions (Barbour and Davis 1974; Hoffmeister 1989; Jones and Birney 1988; Merritt 1987; Schwartz and Schwartz 1981). Young are weaned at about five to six weeks of age and are sexually mature by ten months (Merritt 1987; Schwartz and Schwartz 1981). Occasionally great horned owls, foxes, coyotes, bobcats, and dogs will prey on mink (Merritt 1987; Schwartz and Schwartz 1981). Although some individuals have lived up to six years, mink seldom exceed two years of age in the wild (Schwartz and Schwartz 1981).

B.4.2 Exposure Profile for the Food Chain Model

Adult mink weigh from 520 to 1,730 g (Merritt 1987; U.S. EPA 1993). Home ranges vary from 19 to 1,900 acres (U.S. EPA 1993).

A year-round food ingestion rate of 0.22 g/g BW/day has been estimated for both male and female mink (U.S. EPA 1993). To express this value in units of g/day, the food ingestion rate was multiplied by the lowest reported body weight (520 g) to yield a food ingestion rate of 114 g/day.

An estimated water ingestion rate of 0.11 g/g BW/day was reported for farm-raised females (U.S. EPA 1993). To express this value in units of g/day, this water ingestion rate was multiplied by the lowest reported body weight of 520 g to yield a water ingestion rate of 57.2 g/day (57.2 ml/day).

An incidental soil or sediment ingestion rate was not available from the literature for the mink. Therefore, an incidental soil or sediment ingestion rate for another mammalian species with similar feeding habits will be used to represent the incidental sediment ingestion rate for a mink. The raccoon was selected as a mammal with similar feeding habits as the mink because both species are

omnivorous, opportunistic feeders and will consume mammals, but also hunt aquatic prey such as fish, crayfish, and amphibians (U.S. EPA 1993). Beyer et al. (1994) reported a soil ingestion rate of 9 percent of the diet for raccoons. Therefore, it will be assumed for the purposes of this risk assessment that the sediment ingestion rate of the mink is also 9 percent of the diet. Using a food ingestion rate of 114 g/day, the incidental sediment ingestion rate is calculated to be 10.3 g/day for a mink.

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APPENDIX C

PCBs in Walleye from Green Bay and Tributaries
Wisconsin Department of Natural Resources Data, December 1999
Upper Green Bay Portion of the Fox River Site
Green Bay, WI
February 2000

FISH / SEDIMENT CONTAMINANTS SYSTEM JOB ID: 657 10:03 Monday, December 27, 1999
FISH RESULTS BY SITE NAME - COLUMN STYLE
PCBS IN WALLEYE FROM GREEN BAY AND TRIBUTARIES

SITE=FOX RIVER BELOW DEPERE LOCATION CODE=055008 COUNTY=BROWN

FIELD NUMBER	T/R/S	COLLECTION DATE	SAMPLE TYPE	SAMPLE FORM	NUMBER OF FISH	AVERAGE LENGTH (IN.)	AVERAGE WEIGHT (KG.)	PCB	
7705	23 20E 15	05/06/1977	WALLEYE	SKIN ON FILLET	1	15.00		4.5	- UG/G
7714	23 20E 15	05/06/1977	WALLEYE	SKIN ON FILLET	1	17.80		6.8	- UG/G
8304	23 20E 15	06/13/1983	WALLEYE	WHOLE FISH	2	19.70	1.30	16.	- UG/G
8405	23 20E 15	01/01/1984	WALLEYE	SKIN ON FILLET	3	15.80	0.40	8.1	- UG/G
8406	23 20E 15	01/01/1984	WALLEYE	SKIN ON FILLET	3	16.57	0.63	2.4	- UG/G
8601	23 20E 15	10/06/1986	WALLEYE	WHOLE FISH	3	17.00	0.78	12.	- UG/G
8603	23 20E 15	10/06/1986	WALLEYE	SKIN ON FILLET	1	19.00	1.06	2.1	- UG/G
8604	23 20E 15	10/06/1986	WALLEYE	SKIN ON FILLET	1	21.00	1.45	2.1	UG/G
8605	23 20E 15	10/06/1986	WALLEYE	SKIN ON FILLET	1	23.50	2.39	1.6	- UG/G
8727	23 20E 15	05/22/1987	WALLEYE	SKIN ON FILLET	1	15.50	0.65	0.88	- UG/G
8728	23 20E 15	05/22/1987	WALLEYE	SKIN ON FILLET	2	13.00	0.34	0.76	- UG/G
8729	23 20E 15	05/22/1987	WALLEYE	SKIN ON FILLET	3	8.60	0.08	0.51	- UG/G
8848	23 20E 15	04/28/1988	WALLEYE	SKIN ON FILLET	1	19.88	1.29	1.2	- UG/G
8849	23 20E 15	04/28/1988	WALLEYE	SKIN ON FILLET	1	21.88	1.57	1.9	- UG/G
8850	23 20E 15	04/28/1988	WALLEYE	SKIN ON FILLET	1	23.88	2.14	2.3	- UG/G
8906	23 20E 15	10/25/1989	WALLEYE	SKIN ON FILLET	5	16.00		2.0	- UG/G
8907	23 20E 15	10/25/1989	WALLEYE	SKIN ON FILLET	5	16.60		1.25	- UG/G
8908	23 20E 15	10/25/1989	WALLEYE	SKIN ON FILLET	5	16.70		1.47	- UG/G
8909	23 20E 15	10/25/1989	WALLEYE	SKIN ON FILLET	5	17.30		1.56	UG/G
8910	23 20E 15	10/25/1989	WALLEYE	SKIN ON FILLET	5	17.60		1.53	- UG/G
8911	23 20E 15	10/25/1989	WALLEYE	SKIN ON FILLET	5	17.80		0.8	- UG/G
8912	23 20E 15	10/25/1989	WALLEYE	SKIN ON FILLET	5	18.00		2.04	- UG/G
8913	23 20E 15	10/25/1989	WALLEYE	SKIN ON FILLET	5	19.00		1.5	- UG/G
8914	23 20E 15	10/25/1989	WALLEYE	SKIN ON FILLET	5	19.80		1.58	- UG/G
8915	23 20E 15	10/25/1989	WALLEYE	SKIN ON FILLET	5	24.10		1.7	- UG/G
8916	23 20E 15	10/25/1989	WALLEYE	SKIN ON FILLET	5	21.50		1.63	- UG/G
9232	23 20E 15	04/13/1992	WALLEYE	SKIN ON FILLET	1	10.30	0.16	0.26	- UG/G
9233	23 20E 15	04/18/1992	WALLEYE	SKIN ON FILLET	1	10.50	0.16	0.25	- UG/G
9234	23 20E 15	04/23/1992	WALLEYE	SKIN ON FILLET	1	13.20	0.33	0.75	- UG/G
9235	23 20E 15	04/23/1992	WALLEYE	SKIN ON FILLET	1	13.20	0.35	0.78	- UG/G
9236	23 20E 15	04/23/1992	WALLEYE	SKIN ON FILLET	1	14.50	0.41	1.1	UG/G
9237	23 20E 15	04/23/1992	WALLEYE	SKIN ON FILLET	1	14.50	0.45	0.71	- UG/G
9238	23 20E 15	04/23/1992	WALLEYE	SKIN ON FILLET	1	15.00	0.47	0.47	- UG/G
9239	23 20E 15	04/23/1992	WALLEYE	SKIN ON FILLET	1	15.80	0.57	0.88	- UG/G
9240	23 20E 15	04/23/1992	WALLEYE	SKIN ON FILLET	1	17.00	0.84	0.35	- UG/G
9241	23 20E 15	04/23/1992	WALLEYE	SKIN ON FILLET	1	17.50	0.82	0.2	- UG/G
9242	23 20E 15	03/29/1992	WALLEYE	SKIN ON FILLET	1	17.80	0.93	1.1	- UG/G
9243	23 20E 15	04/23/1992	WALLEYE	SKIN ON FILLET	1	18.20	1.01	< 0.2	- UG/G
9244	23 20E 15	03/29/1992	WALLEYE	SKIN ON FILLET	1	19.50	1.39	4.6	- UG/G
9245	23 20E 15	03/29/1992	WALLEYE	SKIN ON FILLET	1	19.80	1.74	1.	- UG/G
9246	23 20E 15	03/29/1992	WALLEYE	SKIN ON FILLET	1	20.20	1.48	2.3	- UG/G
9247	23 20E 15	03/29/1992	WALLEYE	SKIN ON FILLET	1	21.50	1.85	2.6	- UG/G
9248	23 20E 15	03/29/1992	WALLEYE	SKIN ON FILLET	1	21.50	1.84	3.4	UG/G
9249	23 20E 15	03/29/1992	WALLEYE	SKIN ON FILLET	1	23.00	2.22	3.8	- UG/G
9250	23 20E 15	03/29/1992	WALLEYE	SKIN ON FILLET	1	23.50	2.30	1.7	- UG/G
9251	23 20E 15	03/29/1992	WALLEYE	SKIN ON FILLET	1	24.00	2.44	2.7	UG/G
9401	23 20E 15	10/06/1994	WALLEYE	SKIN ON FILLET	3	16.10	0.61	0.78	- UG/G
9645	23 20E 15	08/26/1996	WALLEYE	SKIN ON FILLET	1	15.70		0.45	MG/KG

FISH / SEDIMENT CONTAMINANTS SYSTEM JOB ID: 657 10:03 Monday, December 27, 1999
FISH RESULTS BY SITE NAME - COLUMN STYLE
PCBS IN WALLEYE FROM GREEN BAY AND TIBUTARIES

SITE=FOX RIVER BELOW DEPERE LOCATION CODE=055008 COUNTY=BROWN
(continued)

FIELD NUMBER	T/R/S	COLLECTION DATE	SAMPLE TYPE	SAMPLE FORM	NUMBER OF FISH	AVERAGE LENGTH (IN.)	AVERAGE WEIGHT (KG.)	PCB
9646	23 20E 15	08/20/1956	WALLEYE	SKIN ON FILLET	1	16.60	0.12	1.6 - MG/KG
9647	23 20E 15	08/20/1956	WALLEYE	SKIN ON FILLET	1	15.90	0.14	1.4 - MG/KG
9648	23 20E 15	08/20/1956	WALLEYE	SKIN ON FILLET	1	16.50	0.15	0.85 MG/KG
9649	23 20E 15	08/20/1956	WALLEYE	SKIN ON FILLET	1	17.20	0.15	2.7 - MG/KG
9650	23 20E 15	08/20/1956	WALLEYE	SKIN ON FILLET	1	15.90		1.0 - MG/KG
9651	23 20E 15	08/20/1956	WALLEYE	SKIN ON FILLET	1	16.90	0.15	0.83 - MG/KG
9652	23 20E 15	08/20/1956	WALLEYE	SKIN ON FILLET	1	15.70		0.93 - MG/KG
9653	23 20E 15	08/20/1956	WALLEYE	SKIN ON FILLET	1	18.00	0.16	2.9 - MG/KG
9654	23 20E 15	08/20/1956	WALLEYE	SKIN ON FILLET	1	15.30		0.85 - MG/KG
9655	23 20E 15	08/20/1956	WALLEYE	SKIN ON FILLET	1	16.00	0.12	0.95 - MG/KG
9656	23 20E 15	08/20/1956	WALLEYE	SKIN ON FILLET	1	16.50	0.69	0.95 - MG/KG
9657	23 20E 15	08/20/1956	WALLEYE	SKIN ON FILLET	1	14.80	0.50	0.75 - MG/KG
9658	23 20E 15	08/20/1956	WALLEYE	SKIN ON FILLET	1	15.50	0.56	0.79 - MG/KG
9659	23 20E 15	08/20/1956	WALLEYE	SKIN ON FILLET	1	15.50		0.67 - MG/KG
9660	23 20E 15	08/20/1956	WALLEYE	SKIN ON FILLET	1	15.00		0.75 - MG/KG
9661	23 20E 15	08/20/1956	WALLEYE	SKIN ON FILLET	1	21.00		1.5 - MG/KG
9662	23 20E 15	08/20/1956	WALLEYE	WHOLE FISH	1	16.00		8. - MG/KG
9663	23 20E 15	08/20/1956	WALLEYE	WHOLE FISH	1	16.60	0.53	8.1 - MG/KG
9664	23 20E 15	08/20/1956	WALLEYE	WHOLE FISH	1	18.00	0.86	18. MG/KG
9665	23 20E 15	08/20/1956	WALLEYE	WHOLE FISH	1	16.90	0.73	7.5 - MG/KG
9666	23 20E 15	08/20/1956	WALLEYE	WHOLE FISH	1	15.70	0.65	7.1 - MG/KG
9667	23 20E 15	08/20/1956	WALLEYE	WHOLE FISH	1	15.90	0.65	14. MG/KG
9668	23 20E 15	08/20/1956	WALLEYE	WHOLE FISH	1	16.50	0.71	7. - MG/KG
9669	23 20E 15	08/20/1956	WALLEYE	WHOLE FISH	1	17.20	0.75	11. - MG/KG
9670	23 20E 15	04/15/1956	WALLEYE	SKIN ON FILLET	1	20.00		0.96 - MG/KG
9671	23 20E 15	04/15/1956	WALLEYE	SKIN ON FILLET	1	21.00		1.5 - MG/KG
9672	23 20E 15	05/03/1956	WALLEYE	SKIN ON FILLET	1	21.00		1.6 - MG/KG
9673	23 20E 15	04/29/1956	WALLEYE	SKIN ON FILLET	1	22.40		1.4 - MG/KG
9674	23 20E 15	04/29/1956	WALLEYE	SKIN ON FILLET	1	23.90		4. - MG/KG
9675	23 20E 15	04/15/1956	WALLEYE	SKIN ON FILLET	1	20.50		2. - MG/KG
9676	23 20E 15	04/15/1956	WALLEYE	SKIN ON FILLET	1	20.50		1.3 - MG/KG
9837	23 20E 15	04/20/1958	WALLEYE	SKIN ON FILLET	1	11.50	0.20	0.19 UG/G
9838	23 20E 15	04/20/1958	WALLEYE	SKIN ON FILLET	1	11.75	0.22	0.15 - UG/G
9839	23 20E 15	04/22/1958	WALLEYE	SKIN ON FILLET	1	12.75	0.27	0.15 - UG/G
9840	23 20E 15	04/22/1958	WALLEYE	SKIN ON FILLET	1	12.83	0.29	0.21 - UG/G
9841	23 20E 15	04/17/1958	WALLEYE	SKIN ON FILLET	1	13.38	0.31	0.31 - UG/G
9842	23 20E 15	04/22/1958	WALLEYE	SKIN ON FILLET	1	15.63	0.54	0.36 - UG/G
9843	23 20E 15	04/23/1958	WALLEYE	SKIN ON FILLET	1	17.75	0.76	1.1 - UG/G
9844	23 20E 15	04/23/1958	WALLEYE	SKIN ON FILLET	1	17.83	0.76	1.4 - UG/G
9845	23 20E 15	07/08/1958	WALLEYE	SKIN ON FILLET	1	18.25	1.11	0.54 - UG/G
9850	23 20E 15	04/17/1958	WALLEYE	SKIN ON FILLET	1	22.00	1.69	1.2 - UG/G
9851	23 20E 15	04/17/1958	WALLEYE	SKIN ON FILLET	1	22.50	1.63	1.8 - UG/G
9852	23 20E 15	04/17/1958	WALLEYE	SKIN ON FILLET	1	24.25	2.50	0.79 - UG/G

FISH / SEDIMENT CONTAMINANTS SYSTEM JOB ID: 657 10:03 Monday, December 27, 1999
FISH RESULTS BY SITE CODE - COLUMN STYLE
PCBS IN WALLEYE FROM GREEN BAY AND TRIBUTARIES

SITE=FOX RIVER DEPERE FAIRGROUNDS LOCATION CODE=055016 COUNTY=BROWN

FIELD NUMBER	T/R/S	COLLECTION DATE	SAMPLE TYPE	SAMPLE FORM	NUMBER OF FISH	AVERAGE LENGTH (IN.)	AVERAGE WEIGHT (KG.)	PCB
7805	23 20E 22	08/11/1978	WALLEYE	WHOLE FISH	5	18.00		25. - UG/G
7901	23 20E 22	04/04/1979	WALLEYE	WHOLE FISH	2	15.80		16. - UG/G
7904	23 20E 22	04/04/1979	WALLEYE	SKIN ON FILLET	1	15.20		3.7 - UG/G
7905	23 20E 22	04/04/1979	WALLEYE	SKIN ON FILLET	1	17.10		3.3 - UG/G
7906	23 20E 22	04/04/1979	WALLEYE	SKIN ON FILLET	3	11.40		3.3 - UG/G
7907	23 20E 22	04/04/1979	WALLEYE	SKIN ON FILLET	6	9.40		1.5 - UG/G
7908	23 20E 22	04/04/1979	WALLEYE	SKIN ON FILLET	2	12.60		3.2 - UG/G
8001	23 20E 22	10/02/1980	WALLEYE	WHOLE FISH	5	15.10	0.40	9.1 - UG/G
8101	23 20E 22	03/13/1981	WALLEYE	WHOLE FISH	5	19.20	1.10	10.7 - UG/G
8102	23 20E 22	03/13/1981	WALLEYE	WHOLE FISH	4	20.40	1.60	13.7 - UG/G
8104	23 20E 22	09/28/1981	WALLEYE	WHOLE FISH	5	17.00	0.70	22. - UG/G
8104	23 20E 22	09/28/1981	WALLEYE	WHOLE FISH	5	17.30	0.70	13. - UG/G
8201	23 20E 22	08/03/1982	WALLEYE	WHOLE FISH	5	16.20	0.67	16.3 - UG/G
8501	23 20E 22	08/01/1985	WALLEYE	SKIN ON FILLET	1	14.50	0.50	1.6 - UG/G
8502	23 20E 22	08/01/1985	WALLEYE	SKIN ON FILLET	1	15.00	0.46	2. - UG/G
8503	23 20E 22	08/01/1985	WALLEYE	SKIN ON FILLET	1	15.75	0.58	1.6 - UG/G
8504	23 20E 22	08/01/1985	WALLEYE	SKIN ON FILLET	1	17.25	0.91	2.9 - UG/G
8505	23 20E 22	08/01/1985	WALLEYE	SKIN ON FILLET	1	18.00	0.96	1.2 - UG/G
8506	23 20E 22	08/01/1985	WALLEYE	SKIN ON FILLET	1	19.50	1.17	1.2 - UG/G
8508	23 20E 22	08/01/1985	WALLEYE	LIVER	4	18.00		12. - UG/G
8701	23 20E 22	05/11/1987	WALLEYE	SKIN ON FILLET	2	9.90	0.13	0.47 - UG/G

SITE=FOX RIVER HIGHWAY 172 BRIDGE LOCATION CODE=055012 COUNTY=BROWN

FIELD NUMBER	T/R/S	COLLECTION DATE	SAMPLE TYPE	SAMPLE FORM	NUMBER OF FISH	AVERAGE LENGTH (IN.)	AVERAGE WEIGHT (KG.)	PCB
8703	23 20E 14	05/11/1987	WALLEYE	SKIN ON FILLET	2	12.23	0.26	0.59 - UG/G
8704	23 20E 14	05/11/1987	WALLEYE	SKIN ON FILLET	1	13.50	0.36	0.4 - UG/G
8705	23 20E 14	05/11/1987	WALLEYE	SKIN ON FILLET	1	15.30	0.54	1.7 - UG/G

SITE=FOX RIVER HIGHWAY 29 LOCATION CODE=055006 COUNTY=BROWN

FIELD NUMBER	T/R/S	COLLECTION DATE	SAMPLE TYPE	SAMPLE FORM	NUMBER OF FISH	AVERAGE LENGTH (IN.)	AVERAGE WEIGHT (KG.)	PCB
8002	23 20E 01	09/30/1980	WALLEYE	WHOLE FISH	5	13.5	0.45	15. - UG/G

SITE=FOX RIVER HIGHWAY 141 BRIDGE LOCATION CODE=055014 COUNTY=BROWN

FIELD NUMBER	T/R/S	COLLECTION DATE	SAMPLE TYPE	SAMPLE FORM	NUMBER OF FISH	AVERAGE LENGTH (IN.)	AVERAGE WEIGHT (KG.)	PCB
873B	24 20E 01	05/11/1987	WALLEYE	SKIN ON FILLET	1	19.3	1.14	2.3 - UG/G
873C	24 20E 01	05/11/1987	WALLEYE	SKIN ON FILLET	1	20.5	1.41	3.1 - UG/G

FISH / SEDIMENT CONTAMINANTS SYSTEM JOB ID: 657 10:03 Monday, December 27, 1999
FISH RESULTS BY SITE NAME - COLUMN STYLE
PCBS IN WALLEYE FROM GREEN BAY AND TRIBUTARIES

SITE=FOX RIVER JONES POINT LOCATION CODE=055007 COUNTY=BROWN

FIELD NUMBER	T/R/S	COLLECTION DATE	SAMPLE TYPE	SAMPLE FORM	NUMBER OF FISH	AVERAGE LENGTH (IN.)	AVERAGE WEIGHT (KG.)	PCB	
8002	23 20E 14	10/02/1980	WALLEYE	WHOLE FISH	4	9.3	0.11	6.	UG/G

SITE=FOX RIVER MOUTH LOCATION CODE=055004 COUNTY=BROWN

FIELD NUMBER	T/R/S	COLLECTION DATE	SAMPLE TYPE	SAMPLE FORM	NUMBER OF FISH	AVERAGE LENGTH (IN.)	AVERAGE WEIGHT (KG.)	PCB	
7906	24 20E 24	10/17/1979	WALLEYE	WHOLE FISH	5	15.0	0.70	.6.	- UG/G
8202	24 20E 24	08/23/1982	WALLEYE	WHOLE FISH	6	17.1	0.71	19.8	- UG/G

SITE=FOX RIVER RAILROAD AT PORT HOWARD LOCATION CODE=055015 COUNTY=BROWN

FIELD NUMBER	T/R/S	COLLECTION DATE	SAMPLE TYPE	SAMPLE FORM	NUMBER OF FISH	AVERAGE LENGTH (IN.)	AVERAGE WEIGHT (KG.)	PCB	
8702	24 21E 31	05/11/1987	WALLEYE	SKIN ON FILLET	1	21.3	1.25	2.3	- UG/G
8706	24 21E 31	05/11/1987	WALLEYE	SKIN ON FILLET	1	22.3	2.16	1.9	- UG/G

SITE=GREEN BAY GRID 1001 LOCATION CODE=055011 COUNTY=BROWN

FIELD NUMBER	T/R/S	COLLECTION DATE	SAMPLE TYPE	SAMPLE FORM	NUMBER OF FISH	AVERAGE LENGTH (IN.)	AVERAGE WEIGHT (KG.)	PCB	
8904	24 21E 11	04/21/1989	WALLEYE	SKIN ON FILLET	5	19.2	1.17	2.383	- UG/G
8905	24 21E 11	04/21/1989	WALLEYE	SKIN ON FILLET	5	17.2	0.85	1.96	UG/G
8906	24 21E 11	04/24/1989	WALLEYE	SKIN ON FILLET	5	17.6	0.88	1.32	- UG/G
8907	24 21E 11	08/21/1989	WALLEYE	SKIN ON FILLET	5	18.0	1.01	1.48	- UG/G
8908	24 21E 11	09/13/1989	WALLEYE	SKIN ON FILLET	5	21.1	1.66	1.56	- UG/G
8909	24 21E 11	09/13/1989	WALLEYE	SKIN ON FILLET	5	21.1	1.76	2.059	UG/G
8910	24 21E 11	08/24/1989	WALLEYE	SKIN ON FILLET	5	18.4	1.06	1.3	- UG/G
8911	24 21E 11	08/21/1989	WALLEYE	SKIN ON FILLET	5	20.0	1.35	1.44	- UG/G
8912	24 21E 11	08/24/1989	WALLEYE	SKIN ON FILLET	5	18.4	1.07	1.285	UG/G
8913	24 21E 11	08/24/1989	WALLEYE	SKIN ON FILLET	5	19.9	1.37	2.064	- UG/G
8914	24 21E 11	08/24/1989	WALLEYE	SKIN ON FILLET	5	18.0	0.95	1.499	- UG/G
8915	24 21E 11	11/13/1989	WALLEYE	SKIN ON FILLET	5	20.8	1.74	0.971	- UG/G
8916	24 21E 11	09/28/1989	WALLEYE	SKIN ON FILLET	5	18.0	1.22	0.055	- UG/G
8931	24 21E 11	04/24/1989	WALLEYE	SKIN ON FILLET	5	16.8	0.78	1.457	- UG/G
8932	24 21E 11	05/01/1989	WALLEYE	SKIN ON FILLET	5	18.1	0.97	1.492	- UG/G
8933	24 21E 11	05/01/1989	WALLEYE	SKIN ON FILLET	5	18.6	1.05	1.14	- UG/G
8934	24 21E 11	11/01/1989	WALLEYE	SKIN ON FILLET	5	21.3	1.88	1.664	- UG/G

FISH / SEDIMENT CONTAMINANTS SYSTE JOB ID: 657 10:03 Monday, December 27, 1999

FISH RESULTS BY SITE NAME - C... STYLE

PCBS IN WALLEYE FROM GREEN BAY AND TRIBUTARIES

SITE-GREEN BAY GRID 605 LOCATION CODE-155014 COUNTY-DOOR

FIELD NUMBER	T/R/S	COLLECTION DATE	SAMPLE TYPE	SAMPLE FORM	NUMBER OF FISH	AVERAGE LENGTH (IN.)	AVERAGE WEIGHT (KG.)	PCB
3906		06/13/1989	WALLEYE	SKIN ON FILLET	5	21.1	1.89	1.32 - UG/G
3907		06/13/1989	WALLEYE	SKIN ON FILLET	5	20.0	1.61	1.25 - UG/G
3908		06/13/1989	WALLEYE	SKIN ON FILLET	5	21.4	1.94	1.456 - UG/G
3909		07/21/1989	WALLEYE	SKIN ON FILLET	4	20.3	1.76	1.37 - UG/G
3910		09/23/1989	WALLEYE	SKIN ON FILLET	5	21.1	1.75	9.884 - UG/G
3911		07/19/1989	WALLEYE	SKIN ON FILLET	5	21.7	1.90	1.51 - UG/G

SITE-GREEN BAY GRID 703 LOCATION CODE-155030 COUNTY-DOOR

FIELD NUMBER	T/R/S	COLLECTION DATE	SAMPLE TYPE	SAMPLE FORM	NUMBER OF FISH	AVERAGE LENGTH (IN.)	AVERAGE WEIGHT (KG.)	PCB
8919	30 21E 09	11/10/1989	WALLEYE	SKIN ON FILLET	5	21.3	1.82	1.072 - UG/G
8920	30 21E 09	09/11/1989	WALLEYE	SKIN ON FILLET	4	20.4	1.57	0.56 - UG/G
8921	30 21E 09	11/09/1989	WALLEYE	SKIN ON FILLET	5	21.2	1.61	1.437 - UG/G
8922	30 21E 09	08/31/1989	WALLEYE	SKIN ON FILLET	5	20.7	1.66	0.789 - UG/G

SITE-GREEN BAY GRID 802 LOCATION CODE-435011 COUNTY-OCONTO

FIELD NUMBER	T/R/S	COLLECTION DATE	SAMPLE TYPE	SAMPLE FORM	NUMBER OF FISH	AVERAGE LENGTH (IN.)	AVERAGE WEIGHT (KG.)	PCB
8965	28 21E 07	06/19/1989	WALLEYE	SKIN ON FILLET	3	19.9	1.31	1.02 UG/G

SITE-GREEN BAY GRID 803 LOCATION CODE-155006 COUNTY-DOOR

FIELD NUMBER	T/R/S	COLLECTION DATE	SAMPLE TYPE	SAMPLE FORM	NUMBER OF FISH	AVERAGE LENGTH (IN.)	AVERAGE WEIGHT (KG.)	PCB
7601	28 25E 30	06/23/1976	WALLEYE	SKIN ON FILLET	5	12.8		0.7 - UG/G
7602	28 25E 30	06/23/1976	WALLEYE	SKIN ON FILLET	5	14.8		0.6 - UG/G
7603	28 25E 30	06/23/1976	WALLEYE	SKIN ON FILLET	5	16.5		0.5 - UG/G
8005	28 25E 30	07/27/1980	WALLEYE	SKIN ON FILLET	7	15.9	0.95	8.1 - UG/G
8101	28 25E 30	05/20/1981	WALLEYE	SKIN ON FILLET	1	18.3	1.03	3.4 - UG/G
8102	28 25E 30	05/20/1981	WALLEYE	SKIN ON FILLET	1	19.5	1.35	2.3 - UG/G
8103	28 25E 30	05/20/1981	WALLEYE	SKIN ON FILLET	1	21.3	1.71	4.2 - UG/G
8104	28 25E 30	05/20/1981	WALLEYE	SKIN ON FILLET	1	21.5	1.93	2.5 - UG/G
8105	28 25E 30	05/20/1981	WALLEYE	SKIN ON FILLET	1	21.8	1.80	2.3 - UG/G
8106	28 25E 30	05/20/1981	WALLEYE	SKIN ON FILLET	1	23.4	2.20	4. - UG/G
8107	28 25E 30	05/20/1981	WALLEYE	ABDOMINAL FAT	1	25.6	2.90	94. - UG/G
8108	28 25E 30	05/20/1981	WALLEYE	SKIN ON FILLET	1	25.8	2.90	5.1 - UG/G
8601	28 25E 30	05/28/1986	WALLEYE	SKIN ON FILLET	1	17.5	0.88	0.92 - UG/G
8602	28 25E 30	05/28/1986	WALLEYE	SKIN ON FILLET	1	22.3	1.85	2.3 - UG/G
8603	28 25E 30	05/28/1986	WALLEYE	SKIN ON FILLET	1	24.5	3.04	4.7 - UG/G
8901	28 25E 30	10/21/1989	WALLEYE	SKIN ON FILLET	5	17.7	1.02	0.856 - UG/G

FISH / SEDIMENT CONTAMINANTS SYST JOB ID: 657 10:03 Monday, December 27, 1999
FISH RESULTS BY SITE NAME - COLUMN STYLE
PCBS IN WALLEYE FROM GREEN BAY AND TRIBUTARIES

SITE=GREEN BAY GRID #03 LOCATION CODE=155038 COUNTY=DOOR
(continued)

FIELD NUMBER	T/R/S	COLLECTION DATE	SAMPLE TYPE	SAMPLE FORM	NUMBER OF FISH	AVERAGE LENGTH (IN.)	AVERAGE WEIGHT (KG.)	PCB
8902	28 25E 30	10/22/1969	WALLEYE	SKIN ON FILLET	5	19.2	1.43	0.822 - UG/G
8903	28 25E 30	10/21/1969	WALLEYE	SKIN ON FILLET	5	19.7	1.48	1.37 - UG/G
8906	28 25E 30	10/21/1969	WALLEYE	SKIN ON FILLET	5	19.4	1.35	1.304 UG/G

SITE=GREEN BAY GRID #04 LOCATION CODE=155005 COUNTY=DOOR

FIELD NUMBER	T/R/S	COLLECTION DATE	SAMPLE TYPE	SAMPLE FORM	NUMBER OF FISH	AVERAGE LENGTH (IN.)	AVERAGE WEIGHT (KG.)	PCB
7901	28 25E 25	04/30/1979	WALLEYE	WHOLE FISH	5	17.1		6.3 - UG/G
7903	28 25E 25	05/28/1979	WALLEYE	WHOLE FISH	5	17.9	1.10	7.5 - UG/G
8901	28 25E 25	05/25/1989	WALLEYE	SKIN ON FILLET	4	15.9	0.70	0.802 - UG/G
8903	28 25E 25	05/22/1989	WALLEYE	SKIN ON FILLET	5	18.7	1.11	1.11 - UG/G
8904	28 25E 25	05/22/1989	WALLEYE	SKIN ON FILLET	5	17.6	0.92	0.924 - UG/G
8905	28 25E 25	07/12/1989	WALLEYE	SKIN ON FILLET	5	17.2	0.89	0.593 - UG/G
8909	28 25E 25	08/22/1989	WALLEYE	SKIN ON FILLET	5	18.3	1.09	1.227 - UG/G

SITE=GREEN BAY LITTLE STURGEON BAY LOCATION CODE=155004 COUNTY=DOOR

FIELD NUMBER	T/R/S	COLLECTION DATE	SAMPLE TYPE	SAMPLE FORM	NUMBER OF FISH	AVERAGE LENGTH (IN.)	AVERAGE WEIGHT (KG.)	PCB
8301	27 24E 11	05/19/1983	WALLEYE	SKIN ON FILLET	5	15.3	0.58	0.92 - UG/G
8302	27 24E 11	05/19/1983	WALLEYE	SKIN ON FILLET	5	22.6	2.10	4. - UG/G
8405	27 24E 11	05/30/1984	WALLEYE	SKIN ON FILLET	5	15.5	0.70	0.84 - UG/G
8406	27 24E 11	05/30/1984	WALLEYE	SKIN ON FILLET	5	19.0	1.20	1.3 - UG/G
8407	27 24E 11	05/30/1984	WALLEYE	SKIN ON FILLET	3	23.0	2.60	4.7 - UG/G

SITE=GREEN BAY RED ARROW POINT LOCATION CODE=385011 COUNTY=MARINETTE

FIELD NUMBER	T/R/S	COLLECTION DATE	SAMPLE TYPE	SAMPLE FORM	NUMBER OF FISH	AVERAGE LENGTH (IN.)	AVERAGE WEIGHT (KG.)	PCB
7601	30 24E 10	08/03/1976	WALLEYE	SKIN ON FILLET	3	14.8		0.52 UG/G

SITE=MENOMINEE RIVER ANSUL CHEMICAL LOCATION CODE=385025 COUNTY=MARINETTE

FIELD NUMBER	T/R/S	COLLECTION DATE	SAMPLE TYPE	SAMPLE FORM	NUMBER OF FISH	AVERAGE LENGTH (IN.)	AVERAGE WEIGHT (KG.)	PCB
7604	30 24E 08	08/18/1976	WALLEYE	SKIN ON FILLET	3	16.7		3.3 - UG/G
8303	30 24E 08	05/31/1983	WALLEYE	SKIN ON FILLET	6	15.2	0.32	< 0.2 - UG/G
8304	30 24E 08	05/31/1983	WALLEYE	SKIN ON FILLET	1	17.6	1.15	0.31 UG/G

FISH / SEDIMENT CONTAMINANTS SYSTEM OR ID: 657 10:03 Monday, December 27, 1999
FISH RESULTS BY SITE NAME - COMBINED STYLE
PCBS IN WALLEYE FROM GREEN BAY AND TRIBUTARIES

SITE-MENOMINEE RIVER BELOW ANGUL CHEMICAL LOCATION CODE-305007 COUNTY-MARINETTE

FIELD NUMBER	T/R/S	COLLECTION DATE	SAMPLE TYPE	SAMPLE FORM	NUMBER OF FISH	AVERAGE LENGTH (IN.)	AVERAGE WEIGHT (KG.)	PCB
8701	30 24E 08	06/04/1987	WALLEYE	SKIN ON FILLET	1	14.3	0.44	0.34 - UG/G
8702	30 24E 08	06/04/1987	WALLEYE	SKIN ON FILLET	1	18.3	1.06	1.1 - UG/G
8703	30 24E 08	06/04/1987	WALLEYE	SKIN ON FILLET	1	19.4	1.20	3. - UG/G
8704	30 24E 08	06/04/1987	WALLEYE	SKIN ON FILLET	1	21.4	1.29	2.9 - UG/G
8705	30 24E 08	06/04/1987	WALLEYE	SKIN ON FILLET	1	24.4	2.26	1.3 - UG/G
8802	30 24E 08	11/11/1988	WALLEYE	SKIN ON FILLET	2	7.7	0.06	< 0.2 - UG/G
8803	30 24E 08	11/11/1988	WALLEYE	SKIN ON FILLET	1	13.6	0.39	< 0.2 - UG/G
8804	30 24E 08	11/11/1988	WALLEYE	SKIN ON FILLET	1	15.3	0.49	< 0.2 - UG/G
8805	30 24E 08	11/11/1988	WALLEYE	SKIN ON FILLET	1	16.6	0.70	< 0.2 - UG/G

SITE-MENOMINEE RIVER MATTIE STREET LOCATION CODE-305021 COUNTY-MARINETTE

FIELD NUMBER	T/R/S	COLLECTION DATE	SAMPLE TYPE	SAMPLE FORM	NUMBER OF FISH	AVERAGE LENGTH (IN.)	AVERAGE WEIGHT (KG.)	PCB
7702	30 24E 08	06/08/1977	WALLEYE	SKIN ON FILLET	4	19.9		0.2 - UG/G

SITE-MENOMINEE RIVER MARINETTE LOCATION CODE-305019 COUNTY-MARINETTE

FIELD NUMBER	T/R/S	COLLECTION DATE	SAMPLE TYPE	SAMPLE FORM	NUMBER OF FISH	AVERAGE LENGTH (IN.)	AVERAGE WEIGHT (KG.)	PCB
7701	30 24E 08	09/08/1977	WALLEYE	WHOLE FISH	5	17.5		13. - UG/G
9314	30 24E 08	04/06/1953	WALLEYE	SKIN ON FILLET	1	19.5	1.1	0.12 - UG/G
9316	30 24E 08	04/06/1953	WALLEYE	SKIN ON FILLET	1	22.0	1.8	1.9 - UG/G

SITE-MENOMINEE RIVER MOUTH LOCATION CODE-305014 COUNTY-MARINETTE

FIELD NUMBER	T/R/S	COLLECTION DATE	SAMPLE TYPE	SAMPLE FORM	NUMBER OF FISH	AVERAGE LENGTH (IN.)	AVERAGE WEIGHT (KG.)	PCB
7904	30 24E 05	07/12/1979	WALLEYE	WHOLE FISH	5	16		0.36 - UG/G

SITE-OCONTO RIVER BELOW STILES DAM LOCATION CODE-435014 COUNTY-OCONTO

FIELD NUMBER	T/R/S	COLLECTION DATE	SAMPLE TYPE	SAMPLE FORM	NUMBER OF FISH	AVERAGE LENGTH (IN.)	AVERAGE WEIGHT (KG.)	PCB
9304	05 21E 18	04/07/1993	WALLEYE	SKIN ON FILLET	1	23.3	2.28	3.3 - UG/G
9305	05 21E 18	04/07/1993	WALLEYE	SKIN ON FILLET	1	23.5	2.40	2.2 - UG/G
9315	05 21E 18	06/08/1993	WALLEYE	SKIN ON FILLET	1	21.0	1.33	1.7 - UG/G

FISH / SEDIMENT CONTAMINANTS SYSTEM JOB ID: 657 10:03 Monday, December 27, 1999
FISH RESULTS BY SITE NAME - COLUMN STYLE
PCBS IN WALLEYE FROM GREEN BAY AND TRIBUTARIES

SITE=OCONTO RIVER OCONTO LOCATION CODE=435004 COUNTY=OCONTO

FIELD NUMBER	T/R/S	COLLECTION DATE	SAMPLE TYPE	SAMPLE FORM	NUMBER OF FISH	AVERAGE LENGTH (IN.)	AVERAGE WEIGHT (KG.)	PCB
8406	28 25E 19	09/19/1984	WALLEYE	SKIN ON FILLET	5	16.85	0.85	1.1 - UG/G

SITE=PESHTIGO RIVER BELOW BADGER MILL LOCATION CODE=385022 COUNTY=MARINETTE

FIELD NUMBER	T/R/S	COLLECTION DATE	SAMPLE TYPE	SAMPLE FORM	NUMBER OF FISH	AVERAGE LENGTH (IN.)	AVERAGE WEIGHT (KG.)	PCB
8401	30 24E 06	05/31/1984	WALLEYE	SKIN ON FILLET	5	11.9	0.24	0.27 - UG/G
8402	30 24E 06	05/31/1984	WALLEYE	SKIN ON FILLET	4	13.2	0.39	0.29 - UG/G
9302	30 24E 06	06/09/1993	WALLEYE	SKIN ON FILLET	1	20.9	1.56	1.1 - UG/G
9304	30 24E 06	06/09/1993	WALLEYE	SKIN ON FILLET	1	21.6	1.96	0.62 - UG/G
9332	30 24E 06	06/02/1993	WALLEYE	SKIN ON FILLET	1	27.0	3.15	1.6 - UG/G

SITE=PESHTIGO RIVER BELOW HIGHWAY 41 LOCATION CODE=385023 COUNTY=MARINETTE

FIELD NUMBER	T/R/S	COLLECTION DATE	SAMPLE TYPE	SAMPLE FORM	NUMBER OF FISH	AVERAGE LENGTH (IN.)	AVERAGE WEIGHT (KG.)	PCB
8301	30 23E 30	06/16/1983	WALLEYE	SKIN ON FILLET	2	22.9	2.42	3. - UG/G
8303	30 23E 30	06/21/1983	WALLEYE	SKIN ON FILLET	3	20.8	1.70	4.6 - UG/G

SITE=PESHTIGO RIVER PESHTIGO LOCATION CODE=385024 COUNTY=MARINETTE

FIELD NUMBER	T/R/S	COLLECTION DATE	SAMPLE TYPE	SAMPLE FORM	NUMBER OF FISH	AVERAGE LENGTH (IN.)	AVERAGE WEIGHT (KG.)	PCB
8001	30 23E 19	09/04/1980	WALLEYE	WHOLE FISH	2	16.6	0.80	7.3 UG/G
8102	30 23E 19	08/14/1981	WALLEYE	WHOLE FISH	2		0.32	3.5 UG/G
8105	30 23E 19	08/24/1981	WALLEYE	WHOLE FISH	4	16.1	0.75	3.25 - UG/G
8201	30 23E 19	08/19/1982	WALLEYE	WHOLE FISH	4	11.7	0.22	3.71 - UG/G

Final Baseline Human Health and Ecological Risk Assessment

Lower Fox River and Green Bay, Wisconsin Remedial Investigation and Feasibility Study

Prepared for:

Wisconsin Dept. of Natural Resources

Prepared by: The RETEC Group, Inc.

December 2002

Final Baseline Human Health and Ecological Risk Assessment

Lower Fox River and Green Bay, Wisconsin
Remedial Investigation and Feasibility Study

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December 2002

EXECUTIVE SUMMARY

A *Baseline Human Health and Ecological Risk Assessment for the Lower Fox River and Green Bay* (BLRA) has been prepared as a companion document to the Remedial Investigation (RI) and Feasibility Study (FS). This section summarizes the baseline risks to human health for the Lower Fox River and Green Bay, and the calculation of sediment quality thresholds (SQTs) that support the selection of a remedy which eliminates, reduces, and/or controls risks identified in the human health and ecological assessments. The SQTs themselves are not cleanup criteria, but are a good approximation of protective sediment values and can be considered to be "working values" from which to select a remedial action level.

This RI/FS report is consistent with the findings of the National Academy of Science's National Research Council Report entitled, *A Risk Management Strategy for PCB Contaminated Sediments* (NRC, 2001).

The overall goals of the BLRA for the Lower Fox River and Green Bay were to:

- Examine how the contaminants of potential concern (COPCs) carried forward from the Screening Level Risk Assessment (SLRA) (RETEC, 1998b) move from the sediment and water into human and ecological receptors within the Lower Fox River and Green Bay.
- Quantify the current (or baseline) human health and ecological risk associated with the COPCs.
- Distinguish those COPCs which pose the greatest potential for risk to human health and the environment and should be carried forward as contaminants of concern (COCs) in the FS.
- Determine which exposure pathways lead to the greatest risks.
- Support the selection of a remedy which eliminates, reduces, and/or controls identified risks by calculating sediment quality thresholds (SQTs).

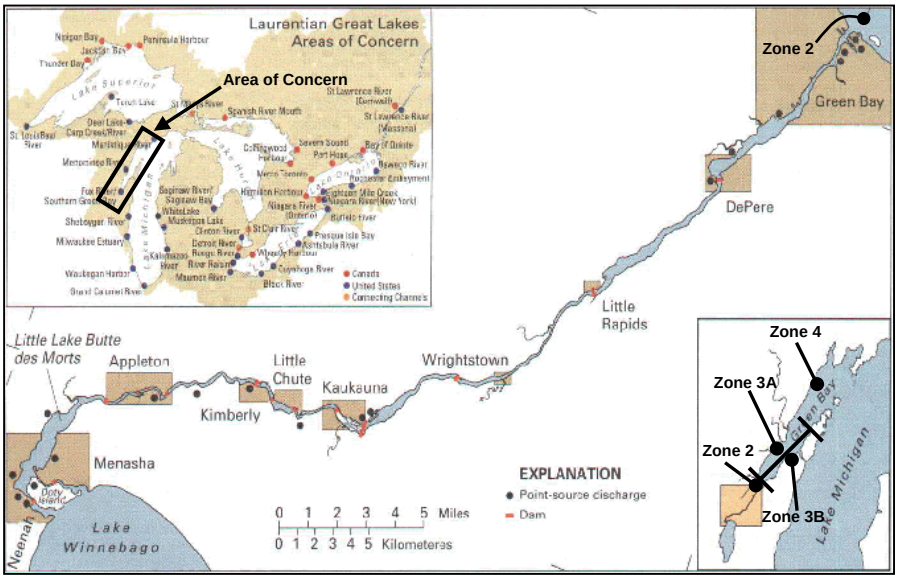


Figure 1 Risk Assessment Study Areas

Site Description

Between 1954 and 1971, paper mills in the Lower Fox River valley manufactured and recycled carbonless copy paper that contained PCBs, resulting in the release of an estimated 313,600 kg (691,370 pounds) of PCBs in the river. It is estimated that 70 percent of the total PCB mass in the river has been transported into Green Bay. Sediment from the Lower Fox River is primarily deposited on the southeastern edge of the bay.

The Fox River valley and Green Bay area is diverse in terms of land use, population density, and habitat types. Overall, the shoreline is much more developed and populated along the Lower Fox River as compared to Green Bay. Both the human health and ecological risk assessments focused on aquatic-dependent receptors and Green Bay has historically supported strong commercial and sport fishing.

For both the human health and ecological assessments, risk was characterized for the four reaches of the Lower Fox River: Little Lake Butte des Morts, Appleton to Little Rapids, Little Rapids to De Pere, and De Pere to Green Bay (Green Bay Zone 1); as well as the zones of the bay: Zone 2, Zone 3A, Zone 3B, and Zone 4 (Figure 1). Therefore, risks between each of these reaches and zones could be compared.

Data Evaluated

The COPCs carried forward from the SLRA included polychlorinated biphenyls (PCBs) (total and selected congeners), dioxins and furan congeners, dichlorodiphenyltrichloroethane (DDT) and its metabolites (4,4'-dichlorodiphenyltrichloroethane [DDE] and 4,4'-dichlorodiphenyldichloroethane [DDD]), dieldrin, and three metals (arsenic, lead, and mercury). In the SLRA, hazard quotients (HQs) calculated for PCBs were at least an order of magnitude greater than the HQs for any of the other COPCs. HQs are the ratios of measured COPC concentrations in media (water, sediment, tissue) as compared to safe COPC concentrations in these media.

All available electronic data collected from Lake Winnebago to northern Green Bay were compiled into a single database—the Fox River Database (FRDB). This database contains 474,218 records of sediment, water, and tissue data from the early 1970s through the late 1990s. For the assessment of baseline risk in the Lower Fox River and Green Bay, a subset of the data contained in the FRDB was evaluated. Data were included based on the specific receptors selected, the time during which the data were collected, and the COPCs of interest.

A time trend analysis of fish tissue data indicates that while PCB concentrations in fish tissue initially significantly decreased, since the mid 1980s changes in these concentrations have either slowed, remained constant, or have resulted in increased tissue concentrations. For this reason, only fish tissue concentrations from 1989 and after were considered for the ecological risk evaluation and the focused human health risk evaluation.

Similarly, for risk evaluation purposes, the concentration of total PCBs in the top 10 cm (4 inches) of sediment was interpolated, because this is the depth of sediment that is of primary biological activity. The degree of biological activity influences the potential for bioaccumulative compounds to be taken up in the food chain. PCB concentrations in sediment were interpolated both horizontally and vertically, but for comparative risk purposes non-interpolated sediment PCB concentrations were also evaluated for risk.

General Conclusions

General conclusions of both the human health and ecological assessments were that:

- Fish consumption is the exposure pathway that represents the greatest level of risk for receptors (other than direct risk to benthic invertebrates).

- The primary COC is PCBs, and other COCs carried forward for remedial evaluation and long-term monitoring are mercury and DDE.
- In general, areas evaluated with the greatest risk are Green Bay zones 1 and 2.

Human Health Risk Assessment

For the human health risk assessment, two evaluations were performed, a baseline risk assessment and a focused risk assessment, which are described shortly. For the baseline risk assessment, all data for a specific medium for each COPC were used to evaluate exposures and risks. For the focused risk assessment, which examined only exposure to PCBs in fish, only fish tissue data from 1989 and after were used.

Receptors evaluated in the human health risk assessment were:

- Recreational anglers,
- High-intake fish consumers,
- Hunters,
- Drinking water users,
- Local residents,
- Recreational water users (swimmers and waders), and
- Marine construction workers.

The principle findings of the human health risk assessment are:

- Consumption of fish from the Lower Fox River and Green Bay presented the highest cancer risks and noncancer hazard indices for the pathways evaluated which also included those associated with consumption of waterfowl, drinking water, breathing air near the river or bay, swimming, and construction in the river or bay.
- PCBs contribute more than 70 percent of the cancer risks found from the consumption of fish and waterfowl.
- Using fish data since 1989, lifetime cancer risks as great as one in 1,000 were found for recreational anglers and high-intake fish consumers exposed to PCBs. High-intake fish consumers are individuals in the recreational angler population who may eat significantly more fish than recreational anglers. Groups within the high-intake fish consumer category that were explicitly evaluated in this risk assessment were low-income minority anglers, Hmong/Laotian anglers, and Native American anglers.
- While high-intake fish consumers are individuals who may eat significantly more fish than typical recreational anglers, there were not large differences in risks between recreational anglers and high-intake fish consumers for the high fish consumption or reasonable maximum exposure scenarios.
- Cancer risks from fish consumption are 1,000 times greater than the one-in-a-million cancer risk, which is the point at which risk management decisions may be made under Superfund. The cancer risks are 100 times greater than the one-in-a-hundred-thousand lifetime cancer risk

used by Wisconsin for evaluating hazardous waste sites.

- Noncancer hazard indices from fish consumption were as much as 50 times greater than levels considered acceptable for exposures ranging from 7 years to a lifetime. The noncancer health effects associated with exposure to PCBs include developmental effects (e.g., neurological impairment in infants and children due to maternal exposure), reproductive effects (e.g., conceptive failure), and immune system suppression (e.g., increased incidence of infectious disease in infants).

- Noncancer hazard indices were also calculated for young children eating fish for the Little Lake Butte des Morts and De Pere to Green Bay reaches, the two reaches with the greatest population density. The hazard indices were approximately 2.4 times those found for adults or as much as 125 times greater than acceptable levels.

- Populations potentially exposed to PCBs via fish consumption are large. There are 136,000 fishing licenses issued to individuals living in counties adjacent to the Lower Fox River and Green Bay. About 10 percent of this angler population, or about 14,000 persons, would be considered high-intake anglers. These populations are potentially exposed to PCBs at levels associated with adverse health consequences.

- Cancer risks and noncancer hazard indices are more than 20 times greater than those from the consumption of fish from Lake Winnebago, which does not

Table 1 Summary of Human Health Risks

Location	Recreational Angler		High Intake Fish Consumer		Hunter		Drinking Water User		Local Resident		Swimmer		Wader		Marine Construction Worker	
	Cancer Risk	Noncancer Hazard Index	Cancer Risk	Noncancer Hazard Index	Cancer Risk	Noncancer Hazard Index	Cancer Risk	Noncancer Hazard Index	Cancer Risk	Noncancer Hazard Index	Cancer Risk	Noncancer Hazard Index	Cancer Risk	Noncancer Hazard Index	Cancer Risk	Noncancer Hazard Index
Little Lake Butte des Morts	$>10^{-4}$	>25	$>10^{-4}$	>35	10^{-6}--10^{-4}	~1	$<10^{-6}$	<1	$<10^{-6}$	<1	$<10^{-6}$	<1	$<10^{-6}$	<1	$\sim10^{-6}$	<1
Appleton to Little Rapids	$>10^{-4}$	>20	$>10^{-4}$	>30	10^{-6}--10^{-4}	~1	$<10^{-6}$	<1	$<10^{-6}$	<1	$<10^{-6}$	<1	$<10^{-6}$	<1	$<10^{-6}$	<1
Little Rapids to De Pere	$>10^{-4}$	>15	$>10^{-4}$	>20	10^{-6}--10^{-4}	~1	$<10^{-6}$	<1	$<10^{-6}$	<1	$<10^{-6}$	<1	$<10^{-6}$	<1	$<10^{-6}$	<1
De Pere to GreenBay	$>10^{-4}$	>35	$>10^{-4}$	>50	10^{-6}--10^{-4}	~1	10^{-6}--10^{-4}	<1	$<10^{-6}$	<1	$<10^{-6}$	<1	$<10^{-6}$	<1	$<10^{-6}$	<1
Zone 3A	$>10^{-4}$	>25	$>10^{-4}$	>50	10^{-6}--10^{-4}	~1	$<10^{-6}$	<1	$<10^{-6}$	<1	$<10^{-6}$	<1	$<10^{-6}$	<1	$<10^{-6}$	<1
Zone 3B	$>10^{-4}$	>25	$>10^{-4}$	>35												
Zone 4	$>10^{-4}$	>25	$>10^{-4}$	>35												

Notes:

Risks and hazard indices are based on reasonable maximum exposures.

Interpretation of cancer risks:

$> 10^{-4}$ indicates significant risk

10^{-6}--10^{-4} indicates possibly significant risks

$<10^{-6}$ indicates risks are negligible

Interpretation of hazard indices:

>1 indicates significant noncancer health effects are possible

<1 indicates noncancer health effects are unlikely to be significant

For recreational anglers, high intake fish consumers and hunters, elevated risks and hazard indices are due primarily to PCBs.

For drinking water users in De Pere to Green Bay reach, arsenic is responsible for caculated cancer risk, but arsenic exposure point concentration was based on one detected value in four samples and reporting limits were high, so actual arsenic concentrat

have a known source of PCBs and serves as a background location.

- There were not large differences in risks between the Lower Fox River and Green Bay, or among the reaches within the Lower Fox River, or among the zones within Green Bay.

- While evidence exists for slow declines of PCBs in fish, such declines were not consistent among species or locations, and projections of future declines cannot be made with sufficient certainty for use in risk assessment. In addition, in some cases, PCBs were found to be increasing.

Other findings of the human health risk assessment are:

- Cancer risks to hunters consuming waterfowl approach a risk of one in 10,000. Noncancer hazard indices were 3.8 times acceptable levels.

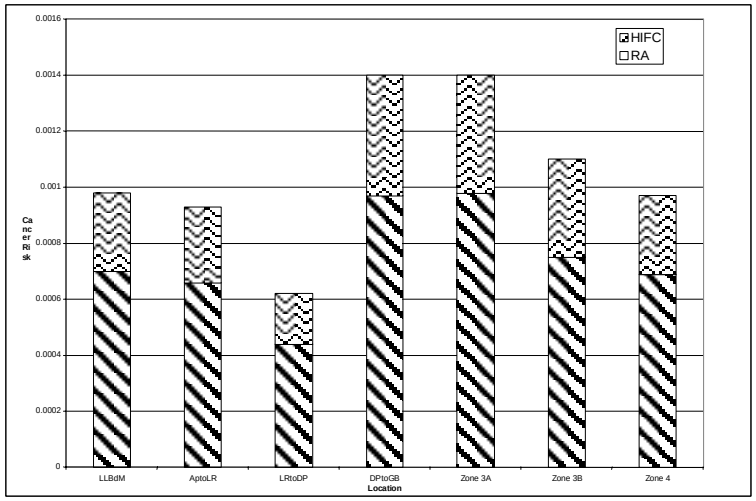
- Cancer risks to local residents exposed to chemicals only through inhalation of air, swimmers, and waders were less than one in a million.

- Cancer risks to drinking water users were less than one in a million in all reaches of the Lower Fox River and all of Green Bay with one exception. The cancer risk in the De Pere to Green Bay Reach was 3.8×10^{-5} due to exposure to arsenic. The arsenic and the exposure to arsenic were based on the detection of this chemical in one of four surface water samples. It is quite likely that this one detected value is anomalous and that the actual risk of exposure to arsenic is much lower. In addition, this reach of the Lower Fox River is not used as a source of drinking water.

- Marine construction workers had cancer risks slightly greater than one in a million. Noncancer hazard indices for drinking water users, local residents, swimmers, waders, and marine construction workers did not exceed acceptable levels.

These results are summarized in Table 1. Figure 2 presents the risks and Figure 3 presents the hazard indices for recreational anglers and high-intake fish consumers due to ingestion of PCBs in fish.

Figure 2 Maximum Cancer Risks for Recreational Anglers and High-intake Fish Consumers



Key:
RA - Recreational Angler
HIFC - High Intake Fish Consumer
LLBdm - Little Lake Butte des Morts
AptoLR - Appleton to Little Rapids
LRtoDP - Little Rapids to De Pere
DPtoGB - De Pere to Green Bay
Zone 3A - Zone 3A of Green Bay
Zone 3B - Zone 3B of Green Bay
Zone 4 - Zone 4 of Green Bay

Ecological Risk Assessment

Types of receptors evaluated for ecological risk included:

- **Aquatic Invertebrates:** Insects and other invertebrates that live in the water and are important prey items for fish and other insects.

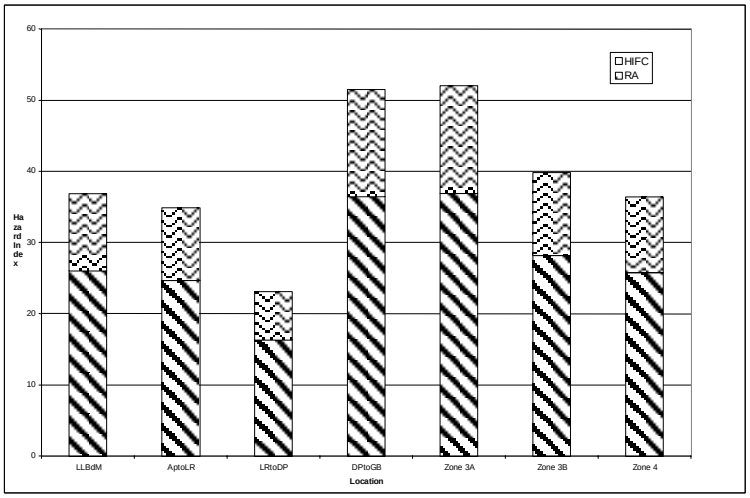
- **Benthic Invertebrates:** Insects and other invertebrates that live in or on the sediment that are important in recycling

nutrients and are a principal part of fish diets.

- **Benthic Fish:** Fish, such as carp and catfish, that live on and forage in the sediments and are in turn eaten by other fish, birds, mammals, and people.
- **Pelagic Fish:** Fish, such as walleye and yellow perch, that live in the water column, and eat other fish or insects that live in the water or on the sediments. These fish may be in turn eaten by other fish, birds, mammals, and people.
- **Insectivorous Birds:** Birds, such as swallows, that eat insects that hatch from the sediments.
- **Piscivorous Birds:** Birds, such as cormorants or terns, that principally eat fish from the Lower Fox River or Green Bay.
- **Carnivorous Birds:** Birds, such as eagles, that eat a variety of prey, including fish or small mammals.
- **Piscivorous Mammals:** Mammals, such as mink, that eat fish as an important part of their diet.

Risk was characterized for assessment endpoints based on the calculation of HQs. In the FRDB, data were generally lacking for piscivorous and carnivorous birds, and no data were available for piscivorous mammals, therefore, ecological modeling was used to estimate COPC exposure to these receptors. HQs that are greater than 1.0 imply that risk may be present. Where available, both the No Observed Adverse Effect Concentration (NOAEC) and Lowest Observed Adverse Effect Concentration (LOAEC) HQs were calculated. Effects evaluated were reproductive dysfunction, death at birth, or deformities in the surviving offspring. When NOAEC HQs exceeded 1.0, but LOAEC HQs were less than 1.0, then it was concluded that there was potential risk. When both the NOAEC and LOAEC HQs exceed 1.0, it was assumed that risk is present.

Figure 3 Maximum Hazard Indices for Recreational Anglers and High-intake Fish Consumers



In addition to the HQ, the assessment provides an evaluation of the uncertainties associated with the risk characterization, and evaluates the estimated risk relative to the habitat, field studies, and population data for the receptors species. Together with the HQs, the components of the evaluation provide resource managers with the information necessary to make risk decisions within the context of the Feasibility Study.

The principle findings of the ecological risk assessment are:

- Total PCBs cause, or potentially cause risk to all identified receptors. The exception is insectivorous birds where the weight of evidence suggests that these receptors are not at risk from PCB

concentrations. Not all receptors at risk or potentially at risk from PCBs are at risk in all river reaches or bay zones.

- Mercury poses a risk in all river reaches and zones, but not to all receptors. Mercury was not identified as a risk for benthic fish, insectivorous birds, or piscivorous mammals.

- DDT or its metabolites poses a risk to benthic invertebrates (Little Lake Butte des Morts Reach, Little Rapids to De Pere Reach, and Green Bay Zone 1), benthic fish (Green Bay zones 1 and 2), pelagic fish (Green Bay zones 1, 2, 3B, and 4), insectivorous birds (Green Bay Zone 2), piscivorous birds (Green Bay zones 1, 2, and 3B), and carnivorous birds (Green Bay zones 1, 2, 3B, and 4).

- Other COPCs identified as causing or potentially causing risk are arsenic (Zone 1 and Zone 3B benthic invertebrates only) lead (benthic invertebrates only in all areas except Green Bay Zone 2, Zone 3A, and Zone 4), 2,3,7,8-TCDD (benthic invertebrates only in Little Lake Butte des Morts Reach and Little Rapids to De Pere Reach), and dieldrin (piscivorous birds in zones 1, 2, and 3B, carnivorous birds in Green Bay Zone 3A, and piscivorous mammals in Green Bay zones 3A and 3B).

Table 2 summarizes ecological risks based on hazard quotients and other lines of evidence. Figures 4 (total PCBs), 5 (mercury), and 6 (DDT and metabolites) present HQs that were greater than 1.0 for selected receptors.

Table 2 Ecological Risk Summary Table

Location	Water Column Invertebrates	Benthic Invertebrates	Benthic Fish	Pelagial Fish	Insectivorous Bird	Piscivorous Bird	Carnivorous Bird	Piscivorous Mammal
LLBdM	● *mercury* ○ *PCBs*	● *lead; mercury; 2,3,7,8-TCDD; PCBs; DDD; DDT*	○ *PCBs*	○ *PCBs*	○ *PCBs*	○ *mercury; PCBs*	○ *PCBs*	● *PCBs*
Appleton to Little Rapids	○ *PCBs*	● *lead; mercury; PCBs*	○ *PCBs*	○ *PCBs*	NA	○ *mercury; PCBs*	● *PCBs* ○ *mercury*	● *PCBs*
Little Rapids to De Pere	● *mercury*	● *lead; mercury; 2,3,7,8-TCDD; PCBs; DDE; DDT*	○ *mercury; PCBs*	○ *mercury; PCBs*	NA	○ *mercury; PCBs*	○ *mercury; PCBs*	● *PCBs*
Zone 1	○ *PCBs*	● *arsenic; lead; mercury; PCBs; DDD; DDE*	○ *PCBs; DDE*	○ *mercury; PCBs; DDE*	○ *PCBs*	○ *mercury; PCBs; dieldrin; DDE*	○ *mercury; PCBs; DDE*	● *PCBs*
Zone 2	● *mercury*	● *mercury; PCBs*			○ *PCBs; DDE*			
Zone 3A		● *PCBs*	○ *PCBs*	○ *PCBs*	NA	○ *mercury; PCBs*	● *PCBs* ○ *dieldrin*	● *PCBs* ○ *dieldrin*
Zone 3B		● *arsenic; lead; mercury; PCBs*		● *PCBs* ○ *mercury; DDE*	NA	● *PCBs* ○ *mercury; dieldrin; DDE*	○ *mercury; PCBs; DDE*	● *PCBs* ○ *dieldrin*
Zone 4		● *PCBs*	NA	○ *PCBs; DDE*	NA	○ *mercury; PCBs*	○ *mercury; PCBs; DDE*	● *PCBs*

Notes:
NA - No data available.
Risk conclusions based on HQs:
- No risk
Risk - ●
Potential Risk - ○
Risk conclusions based on weight of evidence:
- Site-specific receptor data suggest that there is no risk.
- Because of the Federal listing of the bald eagle as threatened, it is concluded that potential risk is actual risk.

Sediment Quality Thresholds (SQTs)

SQTs are sediment concentrations that have been linked to a specific magnitude of risk. SQTs were estimated for PCBs with the assumption that a remedy that reduces PCB exposure would also address the other co-occurring COCs. Risk-based concentrations in fish for human and ecological receptors were determined based on:

- Human health cancer risk levels of 10^{-4}, 10^{-5}, and 10^{-6}, and a noncancer hazard index of 1.0 for risk in recreational anglers and high-intake fish consumers
- The NOAECs and LOAECs for species of benthic invertebrates, fish, birds, and riverine mammals found in the river and bay.

SQTs were developed for each pathway and receptor identified as important in the BLRA by the response agencies of the Lower Fox River and Green Bay (e.g., sport fishing consumption, bald eagles). The SQTs themselves are not cleanup criteria, but are used to evaluate levels of PCBs that will be addressed in the Feasibility Study. The final selection of the remedial action levels is a policy decision left to the response agencies. The development and validation of the mathematical model used to define SQTs is described in the BLRA.

To evaluate how PCBs in sediment result in risk to human or ecological receptors, a methodology is needed for translating concentrations of PCBs in sediment to concentrations in fish and higher order organisms. The Fox River Bioaccumulation Model (FRFood Model) was developed for this purpose. FRFood is a series of mathematical equations that describes a food web and the transfer of bioaccumulating contaminants within that food web. The model includes uptake routes from sediment and water to benthic infauna and ultimately fish, and the model was constructed so that it could be used to either predict fish tissue concentrations from a given sediment concentration, or to predict sediment concentrations from a given fish tissue concentration. The model was validated by running the model "forward;" that is, fish tissue concentrations were predicted from existing sediment concentrations and then compared to measured fish tissue concentrations. When the predicted concentrations were compared to the actual measured concentrations of total PCBs in fish collected in the Lower Fox River and Green Bay, the results were highly comparable.

Estimated SQTs for human health and ecological exposures are shown on Figure 7.

Human Health SQTs

To determine SQTs associated with the protection of human health, fish consumption limits were derived using several different assumptions and risk thresholds. Risk-based fish concentrations (RBFCs) were calculated for recreational anglers and high-intake fish consumers. For recreational anglers, RBFCs were calculated using the average fish intake assumptions from two studies on Michigan anglers (West *et al.*, 1989; West *et al.*, 1993). For high-intake fish consumers, RBFCs were calculated using the average fish intake assumptions for low-income minorities (West *et al.*, 1993) and Hmong (Hutchinson and Kraft, 1994). The RBFCs were generated for each of these exposure scenarios for three

different target risk levels (10^{-6}, 10^{-5}, and 10^{-4}) and for a target noncancer hazard index of 1.0. The RBFCs were used with the results of the FRFood Model to generate a range of SQTs.

Deriving SQTs for each of the consumption scenarios and each of the risks and hazard indices resulted in a total of 48 human health SQTs, a minimum SQT of 1.1 μg/kg (carp at a risk level of 10^{-6}, RME for a high-intake fish consumer) and a maximum SQT of 6,770 μg/kg (yellow perch at a risk level of 10^{-4}, CTE for a recreational angler).

Ecological SQTs

SQTs protective of ecological receptors were calculated for the Lower Fox River and Green Bay separately. Although the remedial methods may differ between reaches of the river evaluated, the SQTs derived for the De Pere to Green Bay Reach will be applied to the entire river. These SQTs are based upon levels of total PCBs in fish that either cause risk to the fish themselves, or to birds or mammals that are eating the fish. The SQTs for no observed adverse effects (NOAEC) to walleye is 176, and for carp is

Figure 4 Selected PCB HQs that Exceed 1.0

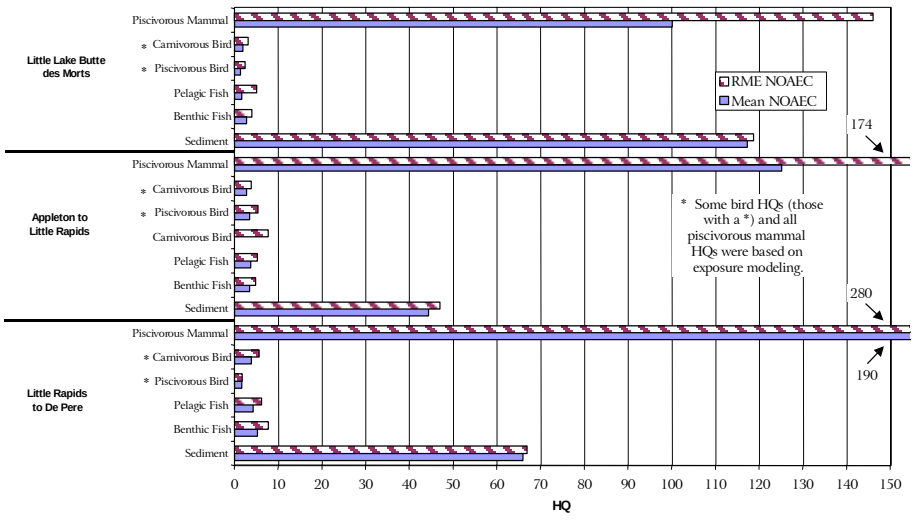
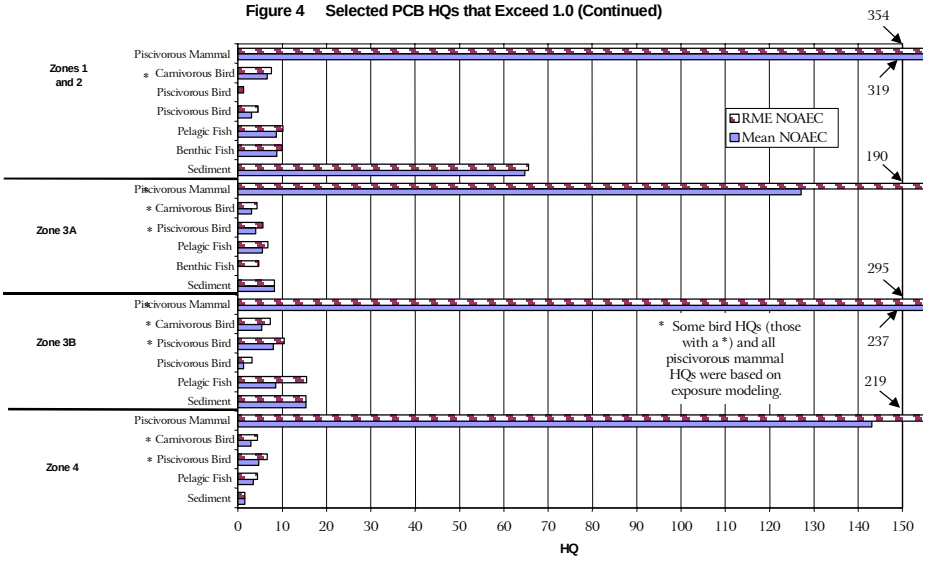


Figure 4 Selected PCB HQs that Exceed 1.0 (Continued)



363. The only calculated SQTs that were lower than these for any of the other receptors were the SQT for benthic invertebrates and the SQTs for piscivorous mammals (mink). The benthic invertebrates threshold effect level (TEL) is a sediment PCB concentration of 31.6 μg/kg and the NOAEC SQT for mink is 24 μg/kg.

Figure 5 Selected Mercury HQs that Exceed 1.0 *

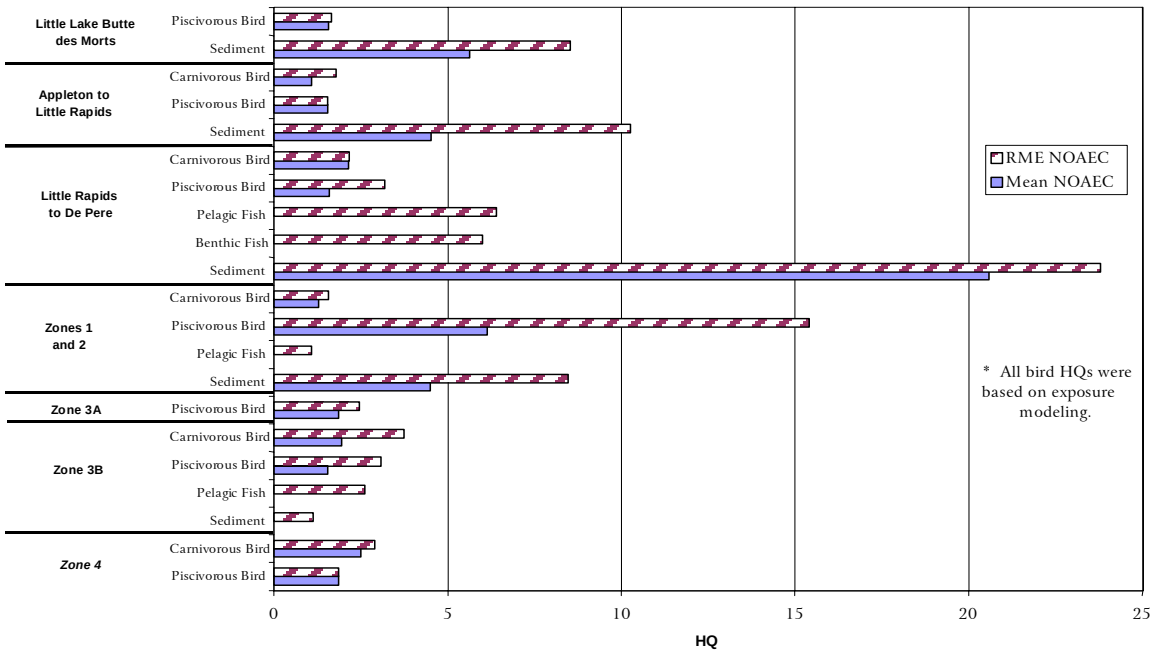


Figure 6 Selected DDT or Metabolite HQs that Exceed 1.0

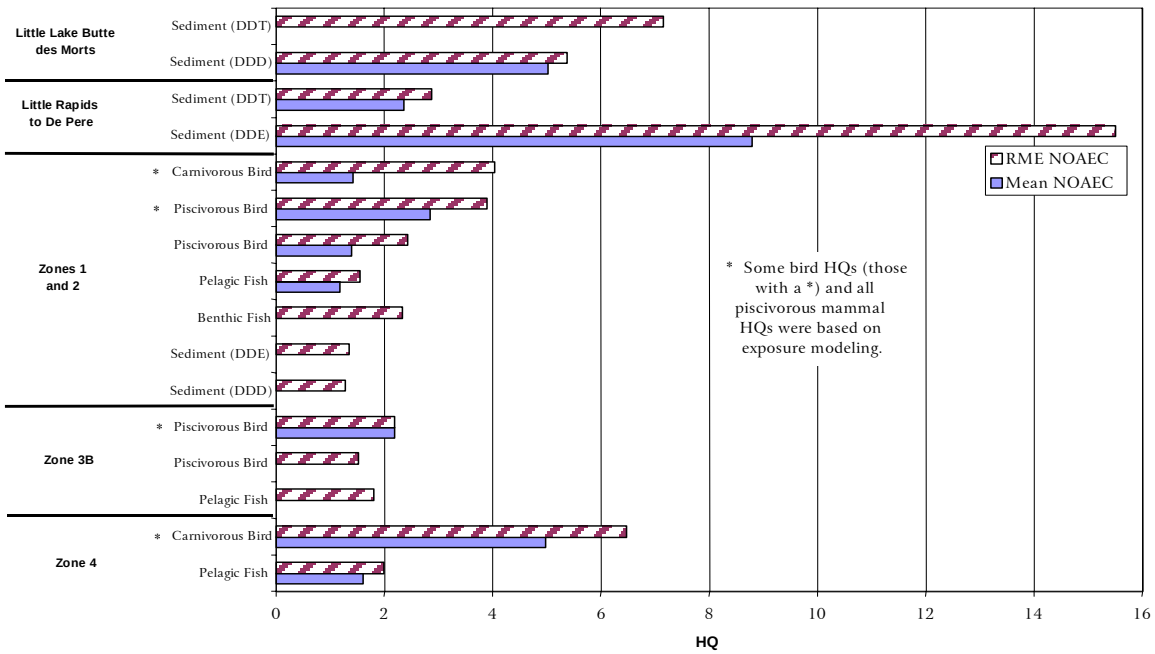


Figure 7. Summary of Sediment Quality Thresholds (SQTs - µg/kg)

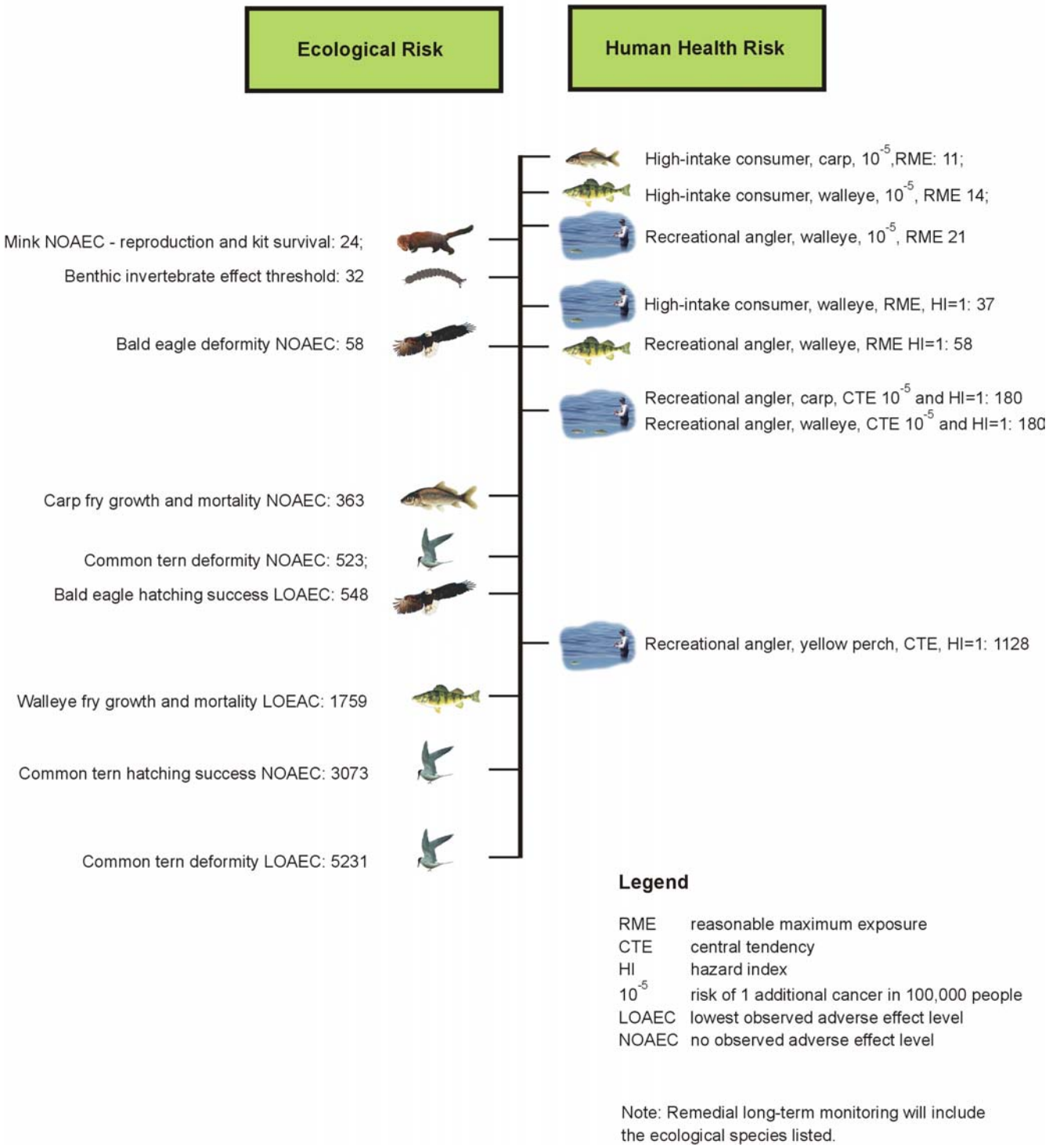


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Table of Contents

Table of Contents

Table of Contents

Table of Contents

Table of Contents

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List of Figures

List of Figures

List of Figures

List of Figures

List of Figures

List of Figures

List of Tables

List of Tables

List of Tables

List of Tables

List of Tables

List of Tables

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List of Tables

List of Tables

List of Tables

List of Tables

List of Tables

List of Tables

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List of Appendices

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List of Acronyms

2,3,7,8-TCDD	2,3,7,8-tetrachloro-*p*-dibenzodioxin
2,3,7,8-TCDF	2,3,7,8-tetrachloro-*p*-dibenzofuran
95% UCL	95 percent upper confidence limit
˚C	degrees centigrade
˚F	degrees Fahrenheit
μg	microgram
μg/dl	micrograms per deciliter
μg/dl-blood	micrograms per deciliter of blood
μg/kg	micrograms per kilogram
μg/kg-BW/day	micrograms per kilogram of body weight per day
μg/kg-day	micrograms per kilogram per day
μg/kg-fillet	micrograms per kilogram of fish fillet
μg/kg-whole body	micrograms per kilogram of whole-body fish
μg/kg-sediment	micrograms per kilogram of sediment
μg/L	micrograms per liter
$\mu g/m^3$	micrograms per cubic meter
μg-PCB/kg-BW/day	micrograms of polychlorinated biphenyl per kilogram of body weight per day
μg-TCDD/kg-lipid	micrograms of 2,3,7,8-tetrachloro-*p*-dibenzodioxin per kilogram of lipid
μm	micrometer
ABS	ingestion absorption factor (fraction absorbed) or inhalation absorption factor (fraction absorbed)
AChE	acetylcholinesterase
ADD	average daily dose
AE	assimilation efficiency (in %)
AEHS	Association for the Environmental Health of Soils
AF	sediment adherence factor (in mg/cm^2)
$a_{f\text{-}wb}$	ratio of concentrations in fish fillet to concentrations in whole body of fish (in kg-fish/kg-fillets)
AHH	aryl hydrocarbon hydroxylase
Ah-R	aryl hydrocarbon receptor
AQUIRE	Aquatic Information Retrieval Database
ARCS	Assessment and Remediation of Contaminated Sediments
As^{3+}	arsenite (trivalent arsenic compound)
As^{5+}	arsenate
AT	averaging time (in days)
ATc	averaging time (carcinogenic)

List of Acronyms

ATnc	averaging time (non-carcinogenic) or averaging time for chronic, noncancer effects (in days)
$ATnc_C$	non-carcinogenic averaging time for a child
ATSDR	Agency for Toxic Substances and Disease Registry (part of the United States Public Health Service)
BAF	bioaccumulation factor
BLRA	baseline risk assessment
BLRPC	Bay Lake Regional Planning Commission
BMF	biomagnification factor
BSAF	biota-to-sediment accumulation factor
BTAG	Biological Technical Assistance Group
BW or *bw*	body weight (in kg)
BW_C	body weight for a child
C	chemical concentration (in mg/kg-soil or mg/L-water)
CA	concentration of chemical in air (in mg/m^3)
C_{ab}	chemical concentration in indoor air during a bath
C_{as}	chemical concentration in indoor air during a shower
CDF	confined disposal facility
CERCLA	Comprehensive Environmental Response, Compensation and Liability Act of 1980 (the Superfund statute)
CF	conversion factor (in kg/g or kg/mg) or volumetric conversion factor (in L/1,000 cc)
Cfish	chemical concentration in fish (in mg/kg-fish)
$Cfish_{EPC}$	exposure point concentration in fish
$C_{fish\text{-}f}$	concentration of PCBs in fish fillet (in μg/kg-fillet)
$Cfish_{meas}$	measured fish chemical concentration
$Cfish_{measi}$	measured concentration of chemical *i* in fish (in mg/kg)
$C_{fish\text{-}wb}$	concentration of PCBs in whole body of fish (in μg/kg-whole body)
cfs	cubic feet per second
cm	centimeter
cm^2	square centimeter
cm^2/event	square centimeters per event
cm/hr	centimeters per hour
C_{oa}	chemical concentration in outdoor air

List of Acronyms

COC	chemical of concern
COPC	chemical of potential concern
C_{pw}	chemical concentration in sediment pore water
CR	contact rate or the amount of impacted medium contacted per event
CS	chemical concentration in sediment (in mg/kg-sediment)
C_{sed}	measured sediment chemical concentration
CSF	cancer slope factor
CSF_d	cancer slope factor for evaluating absorbed dermal doses (in $[mg/kg\text{-}day]^{-1}$)
CSF_i	inhalation cancer slope factor
CSF_o	cancer slope factor for evaluating administered ingestion doses (in $[mg/kg\text{-}day]^{-1}$)
CSFo	oral cancer slope factor (in $[mg/kg\text{-}day]^{-1}$)
C_{sw}	chemical concentration in surface water
$C_{sw\text{-}di}$	measured dissolved concentration for chemical *i* in water (in mg/L)
$C_{sw\text{-}ti}$	measured total concentration of chemical *i* in water (in mg/L)
CTE	central tendency exposure
CW	chemical concentration in water (in mg/L)
C_{wb}	chemical concentration in bath water
CWF	chemical concentration in waterfowl (in mg/kg-waterfowl)
CWF_{EPC}	exposure point concentration in waterfowl
CWF_{meas}	measured chemical concentration in waterfowl
CWF_{measi}	measured concentration of chemical *i* in waterfowl (in mg/kg)
C_{ws}	chemical concentration in shower water
C_x	concentration of the COPC in medium *x* (in mg/kg ww)
cy	cubic yard
days/yr	days per year
DDD	4,4'-dichlorodiphenyl dichloroethane (includes isomers o,p'-DDD and p,p'-DDD)
DDE	4,4'-dichlorodiphenyl dichloroethylene (includes isomers o,p'-DDE and p,p'-DDE)
DDOH-PA	metabolite of DDT conjugated to a fatty acid

List of Acronyms

DDT	4,4'-dichlorodiphenyl trichloroethane (includes isomers o,p'-DDT and p,p'-DDT)
DNA	deoxyribonucleic acid
D.O.	dissolved oxygen
dwt	dry weight
EC_{20}	20 percent effect concentration
EC_{30}	30 percent effect concentration
EC_{50}	50 percent effect concentration
ED	exposure duration (in years)
ED_C	exposure duration for a child
ED_T	estimated daily dose (in mg/kg-BW/day ww)
EEC	Exposure Effect Concentration or Extreme Effect Concentration
FRFood	Fox River Food Model
FRG	Fox River Group, which is composed of the following seven companies (listed alphabetically): Appleton Papers, Inc.,; Fort James Corporation; NCR Corporation; P. H. Glatfelter Company; Riverside Paper Corporation; U.S. Paper Mills Corporation; and Wisconsin Tissue Mills, Inc.
FS	feasibility study
g	gram
GAS	Graef, Anhalt, Schloemer and Associates, Inc.
GBFood	Green Bay Food Model
GBTOXe	Green Bay Toxics Model
g/day	grams per day
GE	gross energy (in kcal/g)
g-fish/day	grams of fish per day
GLEMEDS	Great Lakes Embryo Mortality, Edema, and Deformities Syndrome
GLWQI	Great Lakes Water Quality Initiative
g/meal	grams per meal
g/mole	grams per mole
g-waterfowl/day	grams of waterfowl per day
g/yr	grams per year
H^+	protons
HEAST	Health Effects Assessment Summary Table
Hg^0	elemental mercury
Hg^{2+}	mercuric ion

List of Acronyms

Hg_2^{2+}	mercurous ion
HgOH	inorganic mercury
HI	hazard index
Hi_i	hazard index for chemical *i*
HQ	hazard quotient
hrs/day	hours per day
I	chemical intake (in mg/kg-BW/day)
I_0	interpolated zeroed grid
I_d	interpolated deleted grid
$I_{der\text{-}s}$	absorbed dose from dermal contact with sediment (in mg/kg-BW/day)
$I_{der\text{-}w}$	absorbed intake from dermal contact with water (in mg/kg-BW/day) = $TBS \cdot FBE$
IEUBK	Integrated Exposure Biokinetic/Uptake Model
$I_{ing\text{-}f}$	intake from ingestion of fish (in mg/kg-BW/day)
$I_{ing\text{-}s}$	intake from incidental ingestion of sediment (in mg/kg-BW/day)
$I_{ing\text{-}w}$	intake from ingestion of water (in mg/kg-BW/day)
$I_{ing\text{-}wf}$	intake from ingestion of waterfowl (in mg/kg-BW/day)
I_{inhal}	intake from inhalation (in mg/kg-BW/day)
Inc	intake from ingestion of fish averaged over the exposure period (in mg/kg-day)
IntFacC	intake factor for cancer risk (in $[mg/kg]^{-1}$)
IntFacNC	intake factor for chronic, noncancer effects (in $[mg/kg]^{-1}$)
IPS	Integrated Paper Services
IR	ingestion rate (in g/day or L/day) or inhalation rate (in m^3/hour) or incidental sediment ingestion rate (in mg-sediment/day)
IR_A	ingestion rate for an adult
IR_C	ingestion rate for a child
IRIS	Integrated Risk Information System
IUPAC	International Union of Pure and Applied Chemistry
I_x	rate of ingestion of medium *x* (in mg/day or kg/day ww)
kcal/day	kilocalories per day
kcal/g	kilocalories per gram

List of Acronyms

kg	kilogram (1 kg is approximately equivalent to 2.2 pounds)
kg-fish/kg-fillets	kilograms of fish-to-kilograms of fillets
kg/g	kilograms per gram
kg/L	kilograms per liter
kg/mg	kilograms per milligram
km	kilometer (1 km is approximately equivalent to 0.6 mile)
km^2	square kilometer
km^3	cubic kilometer
K_{ow}	octanol-water partitioning coefficient
Kp	permeability coefficient
LADD	lifetime average daily dose
LC_{10}	10 percent lowest effect concentration
LC_{12}	12 percent lowest effect concentration
LC_{50}	50 percent lowest effect concentration
LC_{90}	90 percent lowest effect concentration
L/1,000 cc	liters per 1,000 cubic centimeters
LD_{10}	lethal dose to 10 percent of test population
LD_{20}	lethal dose to 20 percent of test population
LD_{30}	lethal dose to 30 percent of test population
LD_{50}	lethal dose to 50 percent of test population
L/day	liters per day
LLBdM	Little Lake Butte des Morts
L/m^3	liters per cubic meter
L/mg	liters per milligram
LOAEC	Lowest Observed Adverse Effect Concentration
LOAEL	Lowest Observed Adverse Effect Level
LOEL	Lowest Observed Effect Level
m^2	square meter
m^3	cubic meter
MDNR	Michigan Department of Natural Resources
ME	metabolizable energy (in kcal/g prey)
meals/yr	meals per year
MEC	Moderate Effect Concentration
MeHg	methylmercury (organic mercury)
MFO	mixed function oxidase
mg	milligram

List of Acronyms

mg-Aroclor 1254/kg-BW/day	milligrams of Aroclor 1254 per kilogram of body weight per day
mg/cm^2	milligrams per square centimeter
mg/day	milligrams per day
mg-Hg/kg-BW/day	milligrams of mercury per kilogram of body weight per day
mg/kg	milligrams per kilogram
mg/kg-BW	milligrams per kilogram of body weight
mg/kg-BW/day	milligrams per kilogram of body weight per day
mg/kg-day	milligrams per kilogram per day
mg/kg-DDE	milligrams per kilogram of 4,4'-dichlorodiphenyl dichloroethylene
mg/kg-DDT	milligrams per kilogram of 4,4'-dichlorodiphenyl trichloroethane
mg/kg-egg	milligrams per kilogram of egg
mg/kg-fish	milligrams per kilogram of fish
mg/kg-sediment	milligrams per kilogram of sediment
mg/kg-soil	milligrams per kilogram of soil
mg/kg-waterfowl	milligrams per kilogram of waterfowl
mg/L	milligrams per liter
mg/L-water	milligrams per liter of water
mg/m^3	milligrams per cubic meter
mg/mg	milligrams per milligram
mg-sediment/day	milligrams of sediment per day
mg/yr	milligrams per year
m^3/hr	cubic meters per hour
mi^2	square mile
mi^3	cubic mile
ml/day	milliliters per day
mm	millimeter
m^3/mg	cubic meters per milligram
MNFI	Michigan Natural Features Inventory
MRL	Minimal Risk Level
m/s	meters per second
m^3/s	cubic meters per second
MSA	Metropolitan Statistical Area
MT	metric ton
MW	molecular weight (in g/mole)
N	non-interpolated grid

List of Acronyms

NASS	National Agricultural Statistics Service
NAWQC	National Ambient Water Quality Criteria
NCP	National Contingency Plan
NCR	National Cash Register
"ND"	no data
NEC	No Effect Concentration
ng/kg	nanograms per kilogram
ng/kg-egg	nanograms per kilogram of egg
ng/kg-TCDD/egg	nanograms per kilogram of 2,3,7,8-tetrachloro-*p*-dibenzodioxin per egg
ng/kg-TEQ/egg	nanograms per kilogram of toxic equivalency per egg
ng/kg-ww/eagle	nanograms per kilogram of wet weight per eagle
ng/kg-ww/egg	nanograms per kilogram of wet weight per egg
ng/L	nanograms per liter
ng-TEQ/kg-ww/egg	nanograms of toxic equivalency per kilogram of wet weight per egg
NOAA	National Oceanic and Atmospheric Administration
NOAEC	No Observed Adverse Effect Concentration
NOAEL	No Observed Adverse Effect Level
NOEL	No Observed Effect Level
NRDA	Natural Resource Damage Assessment
N.W.R.	National Wildlife Refuge
OMOE	Ontario Ministry of the Environment
PAH	polynuclear aromatic hydrocarbon
PC	permeability constant (in cm/hr)
PCB	polychlorinated biphenyl
PCDD	polychlorinated dibenzodioxin
PCDF	polychlorinated dibenzofuran
PCH	planar chlorinated hydrocarbon
PCP	pentachlorophenol
pg	picogram
pg/g	picograms per gram
pg/kg-day	picograms per kilogram per day
PHH	planar halogenated hydrocarbons
ppb	parts per billion
ppm	parts per million
ppt	parts per trillion
PRP	potentially responsible party
QA	quality assurance

List of Acronyms

QA/QC	quality assurance/quality control
R	cancer risk
RA	risk assessment
$Ratio_{CAFI}$	child-to-adult fish ingestion ratio
RBFC	risk-based fish concentration
RBSC	risk-based screening concentration
$RBSC_{SA\text{-}fish}$	high-intake fish consumer risk-based screening concentration for carcinogenic or non-carcinogenic chemicals
RETEC	Remediation Technologies, Inc.
RF	reduction factor
RfC	EPA Reference Concentration
RfD	chronic oral reference dose (chemical-specific) or EPA Reference Dose
RfD_d	reference dose for evaluating absorbed dermal doses (in mg/kg-day)
RfD_o	reference dose for evaluating administered ingestion doses (in mg/kg-day)
$RfDo$	oral reference dose for chronic, noncancer effects (in mg/kg-day)
RF_{fish}	reduction factor for fish
RF_{fishi}	reduction factor for chemical i for fish (in mg/mg)
RF_{WF}	reduction factor for waterfowl
RF_{WFi}	reduction factor for chemical i for waterfowl (in mg/mg)
R_i	cancer risk for chemical i
RI	remedial investigation
RI/FS	remedial investigation and feasibility study
RME	reasonable maximum exposure
ROD	Record of Decision
SA	exposed skin surface area (in cm^2 or cm^2/event) = $TBS \cdot FBE$
SAIC	Science Applications International Corporation
SAV	submerged aquatic vegetation and/or floating vegetation
SCS	Soil Conservation Service
SEC	Sediment Effect Concentration
SF	oral cancer slope factor (chemical-specific)
SLRA	screening level risk assessment

List of Acronyms

SMDP	Scientific Management Decision Point
SMU	sediment management unit
SQC	Sediment Quality Criteria
SQT	sediment quality threshold
SVOC	semivolatile organic compound
SWAC	sediment-weighted average concentration
TBS	total body surface area (in cm^2)
TCDD	2,3,7,8-tetrachloro-*p*-dibenzodioxin
TCDD-Eq	2,3,7,8-tetrachloro-*p*-dibenzodioxin equivalent
TCDF	2,3,7,8-tetrachloro-*p*-dibenzofuran
TEC	Threshold Effect Concentration
TEF	toxic equivalency factor
TEL	Environmental Canada Threshold Effect Level
TEQ	toxic equivalency
TF_{bwa}	bath water-to-air transfer factor
Tf_{bwai}	transfer factor for chemical *i* for volatilization from bath water to air (in L/m^3)
TF_{sdpw}	sediment-to-pore water transfer factor
Tf_{sdpwi}	transfer factor for chemical *i* for sediment to pore water (in kg/L)
TF_{sh}	shower water-to-air transfer factor
Tf_{shi}	transfer factor for chemical *i* for volatilization from shower water to air (in L/m^3)
TF_{swoa}	surface water-to-air transfer factor
Tf_{swoai}	transfer factor for volatilization from surface water to outdoor air (in L/m^3)
THI	target hazard index
THQ	target hazard quotient
TIE	Toxicity Evaluation Identification
TOC	total organic carbon
TR	target risk
TRV	Toxicity Reference Value
TSS	total suspended solids
$UHIa1\text{-}inh\text{-}c_i$	unit hazard index for chemical *i* for inhalation of outdoor air by a young child (in m^3/mg)
$UHIa2\text{-}inh\text{-}c_i$	unit hazard index for chemical *i* for inhalation of outdoor air (in m^3/mg)
$UHIfd1\text{-}ing\text{-}c_i$	unit hazard index for chemical *i* for ingestion of waterfowl (in kg/mg)

List of Acronyms

*UHIfsh*1-*ing-c*$_i$	unit hazard index for chemical *i* for ingestion of fish (in kg/mg)
*UHIsd*1-*d-c*$_i$	unit hazard index for chemical *i* for dermal contact with sediment (in kg/mg)
*UHIsd*1-*ing-c*$_i$	unit hazard index factor for chemical *i* for ingestion of sediment (in kg/mg)
*UHIw*1*av-inh-c*$_i$	unit hazard index for chemical *i* for inhalation of indoor air by a young child (in m^3/mg)
*UHIw*1-*d-c*$_i$	unit hazard index for chemical *i* for dermal contact with surface water by a young child (in L/mg)
*UHIw*1-*ing-c*$_i$	unit hazard index for chemical *i* for incidental ingestion of surface water by a young child (in L/mg)
*UHIw*2*av-inh-c*$_i$	unit hazard index for chemical i for inhalation of indoor air by an adult (in m^3/mg)
*UHIw*2-*d-c*$_i$	unit hazard index for chemical *i* for dermal contact with surface water (in L/mg)
*UHIw*2-*ing-c*$_i$	unit hazard index for chemical *i* for incidental ingestion of surface water (in L/mg)
*UHIw*3-*d-c*$_i$	unit hazard index for chemical *i* for dermal contact with sediment pore water (in L/mg)
UP	Michigan's Upper Peninsula
URF	unit risk factor
*URFa*1-*inh-c*$_i$	unit risk factor for chemical *i* for inhalation of outdoor air by a young child (in m^3/mg)
*URFa*2-*inh-c*$_i$	unit risk factor for chemical *i* for inhalation of outdoor air (in m^3/mg)
*URFfd*1-*ing-c*$_i$	unit risk factor for chemical *i* for ingestion of waterfowl (in kg/mg)
*URFfsh*1-*ing-c*$_i$	unit risk factor for chemical *i* for ingestion of fish (in kg/mg)
URF$_i$	inhalation unit risk factor
*URFsd*1-*d-c*$_i$	unit risk factor for chemical *i* for dermal contact with sediment (in kg/mg)
*URFsd*1-*ing-c*$_i$	unit risk factor for chemical *i* for ingestion of sediment (in kg/mg)
*URFw*1*av-inh-c*$_i$	unit risk factor for chemical *i* for inhalation of indoor air by a young child (in m^3/mg)
*URFw*1-*d-c*$_i$	unit risk factor for chemical *i* for dermal contact with surface water by a young child (in L/mg)

List of Acronyms

*URFw*1-*ing*-c_i	unit risk factor for chemical *i* for incidental ingestion of surface water by a young child (in L/mg)
*URFw*2*av*-*inh*-c_i	unit risk factor for chemical *i* for inhalation of indoor air by an adult (in m^3/mg)
*URFw*2-*d*-c_i	unit risk factor for chemical *i* for dermal contact with surface water (in L/mg)
*URFw*2-*ing*-c_i	unit risk factor for chemical *i* for incidental ingestion of surface water (in L/mg)
*URFw*3-*d*-c_i	unit risk factor for chemical *i* for dermal contact with sediment pore water (in L/mg)
USACE	United States Army Corps of Engineers
USDA	United States Department of Agriculture
USFWS	United States Fish and Wildlife Service
USGS	United States Geological Survey
UWSGI	University of Wisconsin Sea Grant Institute
W.A.	Wildlife Area
WDH	Wisconsin Department of Health and Social Services
WDNR	Wisconsin Department of Natural Resources
WHO	World Health Organization
wLFRM	Whole Lower Fox River Model
WSEV	Window Subsampling Empirical Variance
ww	wet weight
YOY	young-of-the-year