Injuries to Avian Resources, Lower Fox River/Green Bay Natural Resource Damage Assessment

Final Report

Prepared for:

U.S. Fish and Wildlife Service
U.S. Department of the Interior
U.S. Department of Justice

Prepared by:

Stratus Consulting Inc.

Contacts:

Hector Galbraith
Mace Barron
Doug Beltman
Josh Lipton

May 7, 1999

CONTENTS

Chapter 6 Injuries to Waterfowl: Consumption Advisories

Chapter 7 Weight of Evidence Evaluation

Chapter 8 Injury Determination

Appendices

FIGURES

TABLES

ACRONYMS

AHH	aryl hydrocarbon hydroxylase
ALAS	aminolevulinic acid synthase
CDF	Confined Disposal Facility
DOI	Department of Interior
dioxin; TCDD	2,3,7,8-tetrachlorodibenzo-p-dioxin
EROD	ethoxyresorufin-O-deethylase
LD50	lethal dose resulting in 50% mortality of the exposed population
LOEC	lowest observed effect concentration
mRNA	messenger RNA
NOAA	National Oceanic and Atmospheric Administration
NOEC	no observed effect concentration
NRDA	natural resource damage assessment
PB	phenobarbital
PCBs	polychlorinated biphenyls
PCDD	polychlorinated dibenzodioxins
PCDF	polychlorinated dibenzo-furan
PWRC	Patuxent Wildlife Research Center
TCDD-eq	TCDD equivalent concentration
TEF	toxicity equivalency factor
U.S. EPA	United States Environmental Protection Agency
USFWS	United States Fish and Wildlife Service
USGS	United States Geological Survey
WHO	World Health Organization

CHAPTER 1
INTRODUCTION

This document presents an evaluation of injuries to avian resources (birds) resulting from releases of polychlorinated biphenyls (PCBs) from paper company facilities along the Lower Fox River, Wisconsin. This evaluation has been performed as part of the natural resource damage assessment (NRDA) being performed for the Lower Fox River/Green Bay site by the U.S. Fish and Wildlife Service (USFWS, or the Service) on behalf of the U.S. Department of the Interior (the Department), the National Oceanic and Atmospheric Administration (NOAA), the Oneida Indian Tribe of Wisconsin, and the Menominee Indian Tribe. This report was prepared by Stratus Consulting Inc. under contract to the Service. The purpose of this injury evaluation is to assess:

- whether birds that use the Lower Fox River, Green Bay, and parts of Lake Michigan (the assessment area) have been exposed to PCBs
- whether birds in the assessment area have been injured as a result of exposure to PCBs.

In this chapter we introduce and define relevant terms from the Department NRDA regulations at 43 CFR Part 11, provide background information on PCB contamination of the area, and describe the overall organization of this report.

1.1 TERMS AND DEFINITIONS

The Department has promulgated regulations for the performance of NRDAs [43 CFR Part 11]. The term "injury" is defined in the Departmental regulations as "a measurable adverse change, either long or short term, in the chemical or physical quality or the viability of a natural resource resulting either directly or indirectly from exposure to a release of a hazardous substance" [43 CFR § 11.14]. The Departmental regulations also identify specific adverse changes that are defined as injuries. The relevant definitions of injury to avian resources assessed by the Trustees in this report are:

- concentrations of PCBs "sufficient to cause the biological resource or its offspring to have undergone at least one of the following adverse changes in viability: death, disease, behavioral abnormalities, cancer, genetic mutations, physiological malfunctions (including malfunctions in reproduction), or physical deformations" [43 CFR § 11.62 (f)(1)(i)]

- concentrations of PCBs sufficient to "exceed action or tolerance levels established under section 402 of the Food, Drug and Cosmetic Act, 21 U.S.C. 342, in edible portions of organisms" [43 CFR § 11.62 (f)(1)(ii)]

- concentrations of PCBs sufficient to "exceed levels for which an appropriate State health agency has issued directives to limit or ban consumption of such organism" [43 CFR § 11.62 (f)(1)(iii)].

NRDA injury determination assessment comprises two phases:

1. ***Pathway determination.*** In the pathway determination phase, pathways by which natural resources come into contact with hazardous substances are identified [43 CFR § 11.63]. The pathway is the "route or medium through which . . . a hazardous substance is or was transported from the source of the discharge or release to the injured resource" [43 CFR § 11.14 (dd)]. Thus, pathway analysis is an important component of demonstrating the linkage between the release of a hazardous substance and the injured natural resource. Other NRDA reports will specifically evaluate the pathways by which PCBs released from paper company facilities have come to be located in the Fox River, Green Bay, and parts of Lake Michigan. This report, however, presents data that describe those pathways by which birds have been exposed to PCBs. For all of the birds addressed in this report, the primary pathway is through dietary exposure.

2. ***Injury determination.*** In this phase, the trustees determine whether adverse effects that meet the definitions of injury set forth at 43 CFR § 11.62 have occurred as a result of exposure to hazardous substances.

1.2 BACKGROUND

PCBs were released into the Fox River/Green Bay system from Fox River paper company facilities that produced or processed PCB-containing carbonless copy paper waste (Wisconsin DNR, 1998). Estimates of the amount of PCBs discharged into the Fox River from paper company facilities range from 420,000 to 825,000 pounds from 1954 to the present (Wisconsin DNR, 1998). An extensive study of PCB fate and transport in the Fox River/Green Bay system demonstrated that PCBs move from the river into the bay, where they enter the food chain (DePinto et al., 1994). A mass balance study estimated that over 90% of the PCBs entering Green Bay in 1989 were from the Fox River (DePinto et al., 1994).

1.3 INJURY EVALUATION METHODS

Injuries to avian resources are determined in this report primarily through the use and interpretation of historical studies on birds in the assessment area. PCB contamination in Green Bay birds was first detected in the early 1970s (Bishop et al., 1992). Since then, multiple studies have been conducted on the exposure to and accumulation of PCBs in Green Bay birds and on adverse effects resulting from this exposure. Most of these studies have been published in the peer-reviewed literature; the evaluation presented in this report is based primarily on peer-reviewed scientific papers.

For this injury determination, the previously available information was supplemented by the collection and chemical analysis of a limited number of tern eggs (12) from the Green Bay assessment area in 1996. This data collection effort, which was outlined in the NRDA Assessment Plan published by the Service (61 Fed. Reg. 43,558), is described in detail in Appendix B.

When available, we relied on and report the statistical analyses conducted by the study authors. In cases where the study authors did not conduct their own statistical analyses, we conducted the analyses using raw data either reported in the study paper or obtained directly from the study authors. Cases where we conducted our own statistical analyses are clearly identified as such, and the statistical methods used to conduct the analyses are also identified. We used a statistical significance level of $\propto = 0.05$.

Finally, the available information was used in a weight-of-evidence approach to determine injuries to avian resources. The methods of the weight-of-evidence approach are described in Chapter 7.

As described in more detail in Chapter 8, the methods described here that were used to determine injuries to avian resources are consistent with those contained in the Departmental regulations for NRDA [43 CFR §11.64].

1.4 DOCUMENT ORGANIZATION

This document is organized as follows. Chapter 2 provides an overview of the avian resources of the Lower Fox River and Green Bay. Chapter 3 contains a review of the known toxicological effects of PCBs on birds, develops a “taxonomy” of PCB-induced injuries to birds, and develops avian toxicity thresholds. Chapter 4 summarizes information regarding exposure to PCB contamination in birds in the assessment area and evaluates the likelihood of PCB-induced injuries to these birds by comparing the exposure concentrations to the toxicity thresholds developed in Chapter 3. In Chapter 5, available site-specific biological data are reviewed by individual species to evaluate whether field data indicate that birds in the assessment area have been injured by PCBs according to injury definitions related to adverse effects on viability. Because of the availability of scientific information and studies, the bird species discussed in Chapter 5 are the double-crested cormorant, Forster's tern, common tern, Caspian tern, tree swallow, red-breasted merganser, black-crowned night heron, and bald eagle. Chapter 6 determines injuries to ducks according to the injury definitions of exceedences of state or federal tolerance limits for PCBs in tissue, and exceedences of state or federal threshold concentrations for establishing consumption advisories. Chapter 7 then presents a weight-of-evidence evaluation of the role that PCBs have played in causing injuries to birds in the assessment area. Chapter 8 presents a determination of injuries pursuant to the Departmental NRDA regulations. Chapter 9 lists references cited.

Appendix A provides scientific names for all bird species mentioned in the text, and Appendix B provides documentation for 1996 field collection and chemical analysis of common and Forster's tern eggs. Appendix C provides the methods and results of waterfowl collection by the Service in 1987.

CHAPTER 2
ASSESSMENT AREA AVIAN RESOURCES

2.1 THE ASSESSMENT AREA

As described in the NRDA Assessment Plan published by the Service (61 Fed. Reg. 43,558), the assessment area for this NRDA includes the Lower Fox River, all of Green Bay, and parts of Lake Michigan. The assessment area is located on the northwest side of Lake Michigan (Figure 2-1) and lies within the Great Lakes ecoregion, an area that comprises a mixture of aquatic, agricultural, wetland, forested (deciduous and coniferous), and urban habitats. The main historical and current types of land use in the assessment area are agricultural, recreational, logging, and industrial/residential (largely confined to areas along the Fox River). Ecological habitats in the assessment area are primarily nonurban; while industrialization and residential development have taken place along parts of the Lower Fox River, much of the area is still dominated by low intensity agriculture, wetlands, and forests. This land use pattern has important implications for the use of the assessment area by birds, as described below.

The climate of the assessment area is highly seasonal and continental, with an average July air temperature of about 67°F and an average January air temperature of 20°F (Robbins, 1991). The average depth of soil frost in late February is about 20 inches. Annual precipitation is approximately 33 inches (Robbins, 1991). While the low winter temperatures ensure that many bird species that depend on freshwater habitats migrate out of the area, the high summer temperatures and precipitation ensure that vegetation growth is lush, with associated diverse bird habitats and communities.

2.2 AVIAN DIVERSITY IN THE ASSESSMENT AREA

Green Bay and the Lower Fox River is an important site within the Great Lakes ecoregion for breeding and migratory birds (Temple and Cary, 1987; Erdman and Jacobs, 1991; Robbins, 1991). During the five years of the Wisconsin Checklist Project, from 1982 until 1986, observers recorded over 250 bird species in the five Wisconsin counties (Door, Kewaunee, Brown, Oconto, and Marinette) immediately adjacent to Green Bay and the Lower Fox River (Table 2-1). During the Atlas of Breeding Birds of Michigan project (1983-1988), 91 bird species were found breeding in the townships adjacent to the Michigan Green Bay shore (Brewer et al., 1991). This high degree of species richness is largely due to four factors: the proximity of a major bird migration route, longitude, plant community diversity, and high quality habitat.

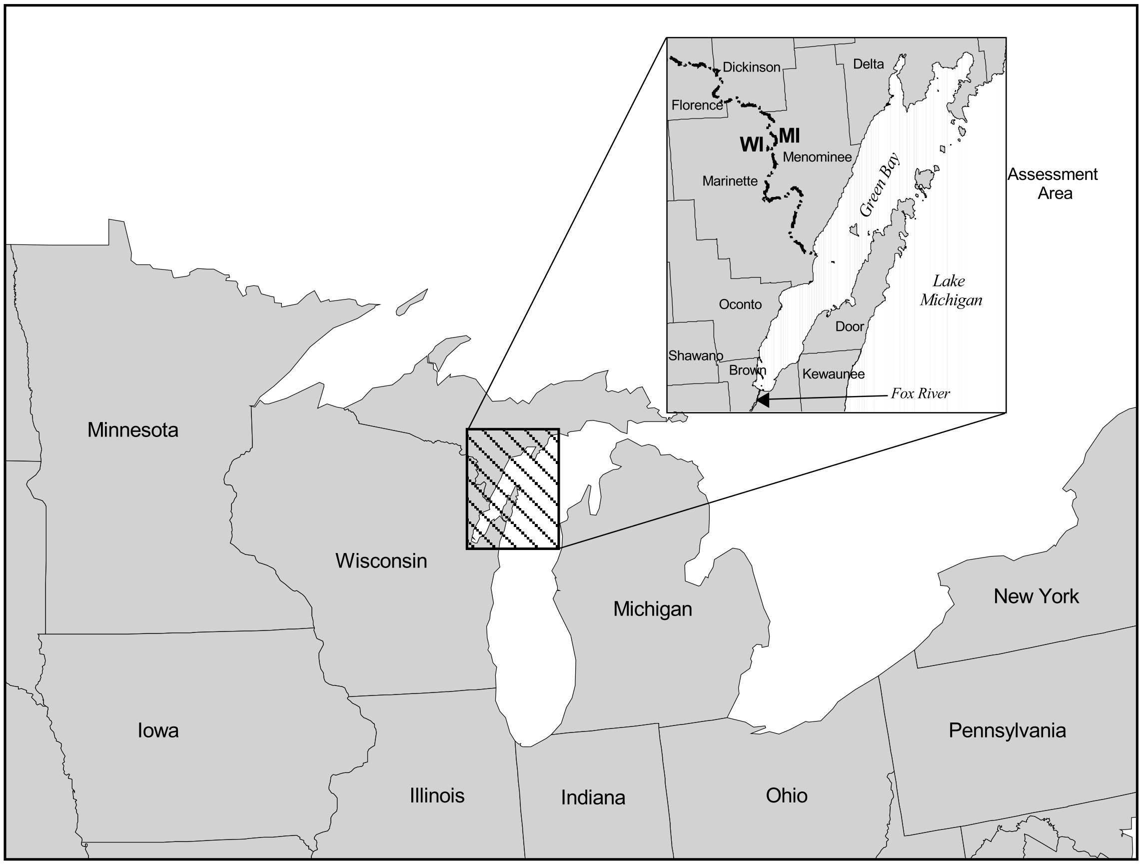


Figure 2-1. The location of the assessment area.

Green Bay and the Lower Fox River are situated on one of the major bird migration routes in North America — the Mississippi Flyway (Figure 2-2). Birds flying south during the fall from their breeding areas in Canada and flying north in the spring funnel through the Lake Michigan and Green Bay area. This results in the regular occurrence of many species that neither breed nor winter in the area (e.g., tundra swans, oldsquaw, and a large number of shorebird species).

The most spectacular of these migratory movements involves the fall influx of waterfowl species into Green Bay. Hundreds of thousands of ducks and geese traveling south from northerly breeding areas use the wetlands surrounding the bay as roosting and feeding areas. These species, which include Canada goose, mallard, teal species, scaup, goldeneye, and many others, are the basis for the intense and economically important duck hunting that takes place in the bay each fall.

Because of its longitude, the assessment area supports birds that are typical of both more western and eastern habitats. For example, both the western and the eastern meadowlark were recorded in the Green Bay area during the Wisconsin Checklist Project, as were the western marbled godwit

Table 2-1
Bird Species Recorded during the Wisconsin Checklist Project in Assessment Area Counties from 1982 until 1986

Species[a]	Breeding/ Summer Visitor	Migrant	Winter Visitor	Year Round Resident	Seasonal Status Uncertain	Federal Status[b]	State Status[b]
Red-throated loon		✔					
Common loon	✔						
Pied-billed grebe	✔						
Horned grebe		✔					
Red-necked grebe		✔					E
Double-crested cormorant	✔	✔					
American bittern	✔						
Least bittern	✔						
Great blue heron	✔						
Green-backed heron	✔						
Great egret	✔						T
Cattle egret	✔						
Black-crowned night-heron	✔						
Tundra swan		✔					
Mute swan				✔			
Snow goose		✔					
Wood duck	✔	✔					
Canada goose		✔		✔			
Green-winged teal	✔	✔					
American black duck		✔		✔			
Mallard		✔		✔			
Northern pintail	✔	✔					
Blue-winged teal	✔	✔					
Northern shoveller		✔					
Gadwall	✔	✔					
American wigeon		✔					
Canvasback		✔					
Redhead	✔	✔					
Ring-necked duck	✔	✔					
Greater scaup	✔	✔					

a. Bird species in italics obtain their food from aquatic habitats and therefore are at increased risk of exposure to PCBs.
b. T = threatened, E = endangered.

Table 2-1 (cont.)
Bird Species Recorded during the Wisconsin Checklist Project in Assessment Area Counties from 1982 until 1986

Species[a]	Breeding/ Summer Visitor	Migrant	Winter Visitor	Year Round Resident	Seasonal Status Uncertain	Federal Status[b]	State Status[b]
Lesser scaup	✔	✔					
Oldsquaw			✔				
White-winged scoter		✔					
Common goldeneye		✔		✔			
Bufflehead		✔	✔				
Hooded merganser	✔	✔					
Common merganser		✔		✔			
Red-breasted merganser	✔	✔					
Ruddy duck		✔					
Turkey vulture	✔						
Osprey	✔						T
Bald eagle				✔		T	
Northern harrier	✔	✔					
Sharp-shinned hawk		✔		✔			
Cooper’s hawk		✔		✔			
Northern goshawk				✔			
Red-shouldered hawk	✔						T
Broad-winged hawk	✔	✔					
Red-tailed hawk		✔		✔			
Rough-legged hawk		✔		✔			
American kestrel		✔		✔			
Merlin	✔						
Peregrine falcon		✔				E	E
Gray partridge				✔			
Ring-necked pheasant				✔			
Ruffed grouse				✔			
Sharp-tailed grouse				✔			
Wild turkey				✔			
Northern bobwhite	✔						
Virginia rail	✔						

a. Bird species in italics obtain their food from aquatic habitats and therefore are at increased risk of exposure to PCBs.
b. T = threatened, E = endangered.

Table 2-1 (cont.)
Bird Species Recorded during the Wisconsin Checklist Project in Assessment Area Counties from 1982 until 1986

Species[a]	Breeding/ Summer Visitor	Migrant	Winter Visitor	Year Round Resident	Seasonal Status Uncertain	Federal Status[b]	State Status[b]
Sora	✔	✔					
Common moorhen	✔						
American coot	✔	✔					
Sandhill crane	✔	✔					
Black-bellied plover		✔					
Killdeer	✔						
Greater yellowlegs		✔					
Lesser yellowlegs		✔					
Solitary sandpiper	✔	✔					
Willet		✔					
Spotted sandpiper	✔	✔					
Upland sandpiper	✔	✔					
Hudsonian godwit		✔					
Marbled godwit		✔					
Ruddy turnstone		✔					
Red knot		✔					
Sanderling		✔					
Semipalmated sandpiper		✔					
Least sandpiper		✔					
White-rumped sandpiper		✔					
Baird's sandpiper		✔					
Pectoral sandpiper		✔					
Dunlin		✔					
Stilt sandpiper		✔					
Short-billed dowitcher		✔					
Long-billed dowitcher		✔					
Common snipe	✔	✔					
American woodcock	✔	✔					
Wilson's phalarope	✔	✔					
Red-necked phalarope		✔					

a. Bird species in italics obtain their food from aquatic habitats and therefore are at increased risk of exposure to PCBs.
b. T = threatened, E = endangered.

Table 2-1 (cont.)
Bird Species Recorded during the Wisconsin Checklist Project in Assessment Area Counties from 1982 until 1986

Species[a]	Breeding/ Summer Visitor	Migrant	Winter Visitor	Year Round Resident	Seasonal Status Uncertain	Federal Status[b]	State Status[b]
Franklin's gull		✔					
Bonaparte's gull	✔	✔					
Ring-billed gull				✔			
Herring gull				✔			
Glaucous gull			✔				
Caspian tern	✔	✔					E
Common tern	✔	✔					E
Forster's tern	✔	✔					E
Black tern	✔						
Rock dove				✔			
Mourning dove				✔			
Black-billed cuckoo	✔						
Yellow-billed cuckoo	✔						
Eastern screech owl					✔		
Great horned owl				✔			
Snowy owl			✔				
Barred owl				✔			
Long-eared owl	✔						
Short-eared owl				✔			
Northern saw-whet owl					✔		
Common nighthawk	✔	✔					
Whip-poor-will	✔	✔					
Chimney swift	✔	✔					
Ruby-throated hummingbird	✔	✔					
Belted kingfisher	✔	✔					
Red-headed woodpecker				✔			
Red-bellied woodpecker				✔			
Yellow-bellied sapsucker	✔	✔					
Downy woodpecker				✔			
Hairy woodpecker				✔			

a. Bird species in italics obtain their food from aquatic habitats and therefore are at increased risk of exposure to PCBs.
b. T = threatened, E = endangered.

Table 2-1 (cont.)
Bird Species Recorded during the Wisconsin Checklist Project in Assessment Area Counties from 1982 until 1986

Species[a]	Breeding/ Summer Visitor	Migrant	Winter Visitor	Year Round Resident	Seasonal Status Uncertain	Federal Status[b]	State Status[b]
Northern flicker	✔	✔		✔			
Pileated woodpecker				✔			
Olive-sided flycatcher		✔					
Eastern wood-pewee	✔	✔					
Yellow-bellied flycatcher		✔					
Alder flycatcher	✔	✔					
Willow flycatcher	✔						
Least Flycatcher	✔	✔					
Eastern phoebe	✔	✔					
Great crested flycatcher	✔	✔					
Eastern kingbird	✔	✔					
Horned lark		✔		✔			
Purple martin	✔	✔					
Tree swallow	✔						
Northern rough-winged swallow	✔	✔					
Bank swallow	✔	✔					
Cliff swallow	✔						
Barn swallow	✔	✔					
Gray jay				✔			
Blue jay				✔			
American crow				✔			
Common raven				✔			
Black-capped chickadee				✔			
Boreal chickadee					✔		
Tufted titmouse					✔		
Red-breasted nuthatch				✔			
White-breasted nuthatch				✔			
Brown creeper				✔			
House wren	✔	✔					

a. Bird species in italics obtain their food from aquatic habitats and therefore are at increased risk of exposure to PCBs.
b. T = threatened, E = endangered.

Table 2-1 (cont.)
Bird Species Recorded during the Wisconsin Checklist Project in Assessment Area Counties from 1982 until 1986

Species[a]	Breeding/ Summer Visitor	Migrant	Winter Visitor	Year Round Resident	Seasonal Status Uncertain	Federal Status[b]	State Status[b]
Winter wren				✔			
Sedge wren	✔						
Marsh wren	✔						
Blue-gray gnatcatcher		✔					
Eastern bluebird	✔						
Veery	✔	✔					
Gray-cheeked thrush		✔					
Swainson’s thrush		✔					
Hermit thrush	✔	✔					
Wood thrush	✔	✔					
American robin	✔			✔			
Gray catbird	✔	✔					
Northern mockingbird	✔						
Brown thrasher	✔	✔					
Water pipit		✔					
Bohemian waxwing			✔				
Cedar waxwing	✔			✔			
Northern shrike			✔				
Loggerhead shrike	✔						E
European starling				✔			
Bell’s vireo	✔						T
Solitary vireo	✔	✔					
Yellow-throated vireo	✔						
Warbling vireo	✔	✔					
Philadelphia vireo	✔	✔					
Red-eyed vireo	✔	✔					
Blue-winged warbler		✔					
Golden-winged warbler	✔	✔					
Tennessee warbler		✔					
Orange-crowned warbler		✔					

a. Bird species in italics obtain their food from aquatic habitats and therefore are at increased risk of exposure to PCBs.
b. T = threatened, E = endangered.

Table 2-1 (cont.)
Bird Species Recorded during the Wisconsin Checklist Project in Assessment Area Counties from 1982 until 1986

Species[a]	Breeding/ Summer Visitor	Migrant	Winter Visitor	Year Round Resident	Seasonal Status Uncertain	Federal Status[b]	State Status[b]
Nashville warbler	✔	✔					
Northern parula		✔					
Yellow warbler	✔	✔					
Chestnut-sided warbler	✔	✔					
Magnolia warbler		✔					
Cape May warbler		✔					
Black-throated blue warbler	✔	✔					
Yellow-rumped warbler	✔	✔					
Black-throated green warbler	✔	✔					
Blackburnian warbler	✔	✔					
Pine warbler	✔	✔					
Palm warbler		✔					
Bay-breasted warbler		✔					
Blackpoll warbler		✔					
Cerulean warbler		✔					T
Black-and-white warbler	✔	✔					
American redstart	✔	✔					
Prothonotary warbler	✔						
Ovenbird	✔	✔					
Northern waterthrush	✔	✔					
Louisiana waterthrush		✔					
Connecticut warbler		✔					
Mourning warbler	✔	✔					
Common yellowthroat	✔	✔					
Hooded warbler		✔					T
Wilson’s warbler		✔					
Canada warbler		✔					
Yellow-breasted chat		✔					
Scarlet tanager	✔	✔					
Northern cardinal				✔			

a. Bird species in italics obtain their food from aquatic habitats and therefore are at increased risk of exposure to PCBs.
b. T = threatened, E = endangered.

Table 2-1 (cont.)
Bird Species Recorded during the Wisconsin Checklist Project in Assessment Area Counties from 1982 until 1986

Species[a]	Breeding/ Summer Visitor	Migrant	Winter Visitor	Year Round Resident	Seasonal Status Uncertain	Federal Status[b]	State Status[b]
Rose-breasted grosbeak	✔	✔					
Indigo bunting	✔	✔					
Dickcissel	✔						
Rufous-sided towhee	✔	✔					
American tree sparrow		✔	✔				
Chipping sparrow	✔						
Clay-colored sparrow	✔						
Field sparrow	✔						
Vesper sparrow	✔						
Lark sparrow					✔		
Savannah sparrow	✔						
Grasshopper sparrow	✔						
Le Conte's sparrow	✔						
Fox sparrow		✔					
Song sparrow	✔	✔		✔			
Lincoln's sparrow	✔	✔					
Swamp sparrow	✔	✔					
White-throated sparrow	✔	✔					
White-crowned sparrow		✔					
Harris's sparrow		✔					
Dark-eyed junco	✔	✔	✔				
Lapland longspur		✔					
Snow bunting		✔	✔				
Eastern meadowlark	✔						
Western meadowlark	✔						
Yellow-headed blackbird	✔						
Red-winged blackbird	✔						
Rusty blackbird		✔					
Brewer's blackbird	✔						
Common grackle	✔	✔		✔			

a. Bird species in italics obtain their food from aquatic habitats and therefore are at increased risk of exposure to PCBs.
b. T = threatened, E = endangered.

Table 2-1 (cont.)
Bird Species Recorded during the Wisconsin Checklist Project in Assessment Area Counties from 1982 until 1986

Species[a]	Breeding/ Summer Visitor	Migrant	Winter Visitor	Year Round Resident	Seasonal Status Uncertain	Federal Status[b]	State Status[b]
Brown-headed cowbird	✔	✔		✔			
Orchard oriole		✔					
Northern oriole	✔	✔					
Pine grosbeak			✔				
Purple finch				✔			
Red crossbill				✔			
White-winged crossbill					✔		
Common redpoll			✔				
Pine siskin		✔		✔			
American goldfinch	✔			✔			
Evening grosbeak		✔	✔	✔			
House sparrow				✔			

a. Bird species in italics obtain their food from aquatic habitats and therefore are at increased risk of exposure to PCBs.
b. T = threatened, E = endangered.

Source: Temple and Cary, 1987.

and the eastern Hudsonian godwit. The assessment area is also one of the easternmost breeding sites for the yellow-headed blackbird, and the white pelican (another western species) has recently colonized the assessment area as a breeding species (K. Stromborg, USFWS, personal communication, April 1999). This mixing of western and eastern birds adds to the avifaunal diversity of the assessment area.

Third, within Wisconsin there is a north-south shift in the major plant communities due to climate. The assessment area is located in a transitional zone called the “Tension Zone” (Curtis, 1959), where plant communities that are typical of both major ecoregions can be found. Areas north of the Lower Fox River are dominated by plant communities that are representative of higher, colder latitudes (e.g., an increased dominance of conifer forests). Northern Door County includes subarctic plant communities because of its low warmest daily average temperatures in summer, which are caused by a marked lake effect and Lake Michigan upwelling. Areas to the south have communities adapted to a warmer climate (e.g., hardwood forests). This results in the occurrence within the assessment area of bird species that are typical of both the more northern plant

Figure 2-2. The assessment area in relation to the Mississippi Flyway.

communities (e.g., gray jay, common raven, boreal chickadee, and several *Dendroica* warbler species), together with species more characteristic of southern habitat types (e.g., turkey vulture, mourning dove, and tufted titmouse).

Last, the bird species diversity found in the assessment area is supported by the area's high quality habitats. While many of the birds that breed in or use the area as a migratory staging post can be found elsewhere in the lower, industrialized Great Lakes region, the assessment area, because of its comparatively undisturbed nature and the quality and extent of its habitats, supports more diverse bird communities.

Wetlands are an important habitat for nesting and migratory birds in the assessment area. Figure 2-3 shows that extensive tracts of the west side of the bay comprise coastal wetlands. There are over 250,000 acres of wetlands within five miles of the bay, much of which (almost 52,000 acres) is protected as national forest, state forest, state park, or state wildlife area (complied from data from USGS, 1990; Wisconsin DNR, 1998b). These extensive and contiguous tracts of wetland provide ideal habitat for migratory and nesting birds.

Other important and abundant bird habitats in the assessment area are the small uninhabited islands of Green Bay, which provide nesting sites for colonial waterbirds. Figure 2-4 shows the distribution of these nesting sites and potential nesting sites. Such sites are favored by colonial waterbirds because of their freedom from human disturbance and from mammalian predators such as raccoons, mink, foxes, and coyotes. Many of these islands are well known as waterbird breeding sites and have supported colonies for many years (e.g., herring gulls on Big Sister Island, cormorants on Hat, Spider, and Cat Islands, Caspian terns on Gravelly Island).

2.3 AVIAN RESOURCES ON THE ONEIDA RESERVATION

The proximity of the Oneida Reservation to the Lower Fox River and Green Bay means that environmental changes to the Lower Fox River/Green Bay ecosystem directly impact the reservation ecosystems. The Oneida Reservation comprises approximately 65,400 acres located 3 miles east of the lower Fox River and 2.5 miles south of Green Bay, incorporating part of the city of Green Bay. It is directly connected to the larger assessment area through waterways. All of the major reservation waters are tributaries to the Lower Fox River and Green Bay. Land use within the exterior reservation boundaries ranges from commercial and light industrial to rural agriculture to residential. The northeastern quarter of the reservation is dominated by residential and commercial land uses, while the remaining areas of the reservation are low intensity agricultural, wetlands, and forested lands.

Birds that have been sighted on the Oneida Reservation in recent years include belted kingfisher, sandhill crane, great blue heron, great horned owl, barn owl, screech owl, northern harrier, rough-legged hawk, common nighthawk, turkey vulture, red-tailed hawk, bald eagle, double-crested cormorant, and 12 different species of ducks and geese. Migratory birds that use the reservation as a stopover site include tundra and trumpeter swans and snow geese. Several species of game birds, such as wild turkey, have been successfully reintroduced to the reservation. While this is not a comprehensive list of the birds that use the reservation ecosystem, it is a representative list of the types of birds on the reservation.

The environment of the reservation is important to both the resident and the migratory bird populations that use the Mississippi Flyway. Wetlands on the reservation become waterbird breeding colonies in the spring, and many species such as sandhill cranes have been found nesting in local wetlands. Threatened species such as bald eagles have used the open waters of the

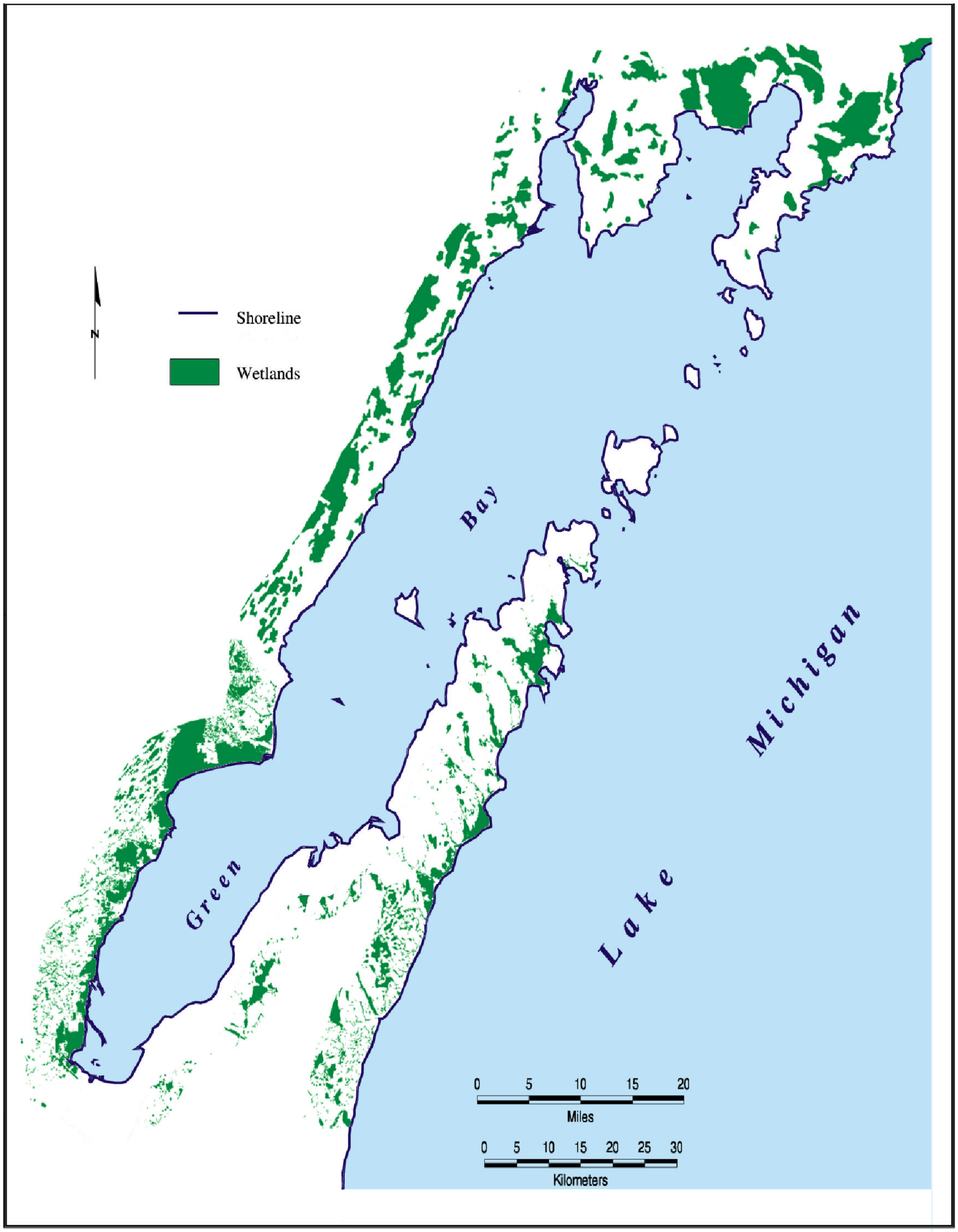


Figure 2-3. Distribution of wetland habitat within five miles of Green Bay or portions of Lake Michigan.

Source: Compiled from data from USGS, 1990, and Wisconsin DNR, 1998b.

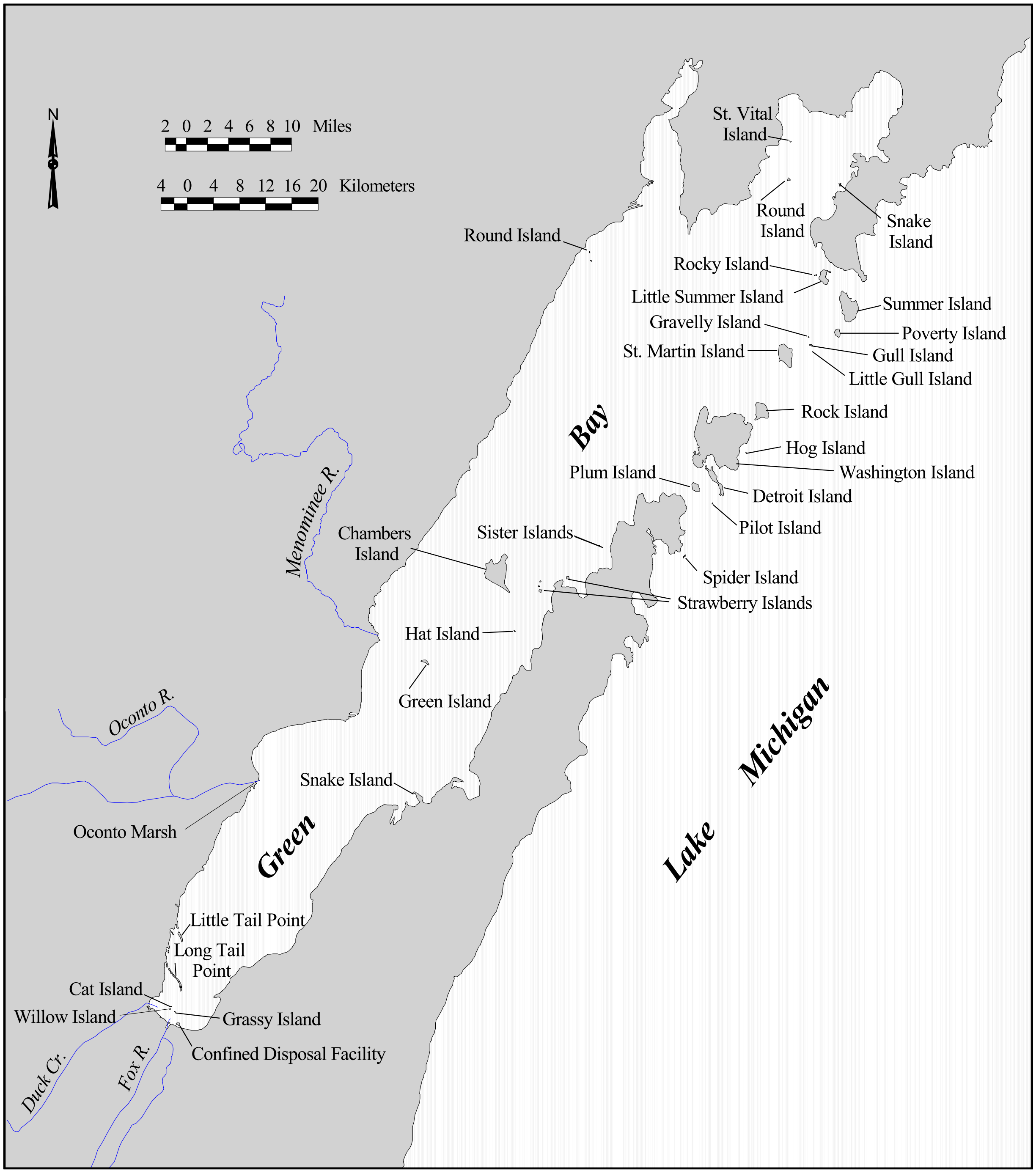


Figure 2-4. Nesting and potential nesting islands for colonial waterbirds in the assessment area.

reservation for winter feeding areas and double-crested cormorants have been seen foraging in reservation lakes and ponds. These examples illustrate the importance of the Oneida Reservation to the Lower Fox River/Green Bay environment.

The combination of agricultural fields and wetlands on the reservation provides ideal habitat for waterfowl. Reservation wetlands provide necessary cover for nesting waterfowl in close proximity to farm fields that supply food for many of the birds. Waterfowl use the many engineered ponds and wetlands on the reservation as a replacement for lost habitat along the Fox River and the shore of Green Bay.

The major waterway on the reservation, Duck Creek, received its name from the large number of ducks that use the river for nesting and rearing of their young. The Duck Creek corridor includes habitat that is ideal for the waterfowl. In the spring the water level is high enough to create nesting areas along many reaches, and the riparian zone in some areas is wide enough to provide the necessary cover for the young.

Why the Birds Are Important to the Reservation

Waterfowl and other game birds are important to the reservation as a food source. The Oneida Tribe chose this land when they were relocated from New York because of the abundant game and the similarity to lands they were leaving behind. The original name for the major river flowing through the reservation was “the place of many ducks.” Ducks and other waterfowl became an important part of the Oneida diet. Oral histories from tribal elders explain how they obtained most of their meat from the local population of game, including waterfowl, turkey, and other small game.

The local birds have always been spiritually important to the Oneida People. For example, the bald eagle was instructed by the Creator to head the bird kingdom, and appreciation for the fulfillment of these duties is expressed in the Oneida Thanksgiving Prayer. The eagle sits on top of the “Tree of Life,” ever vigilant against those who would harm the tree, and eagles carry their prayers up to the Creator. The beauty and songs of all the different birds help the Oneida people appreciate their purpose in life and remind them to enjoy their life cycle to its utmost.

Waterfowl have a special role in the creation story of protecting and safely bringing Mother Earth to the back of the Great Turtle who supports the land we walk on. Birds are also used by the tribal elders to explain many of life’s lessons to younger Oneida people. Some Oneida elders have expressed a sense of shame, loss, and sorrow because of the decreased numbers and diversity of birds. In the Oneida culture it is important to protect and preserve all of the animals, and birds have a special place in the Oneida culture. These feelings are expressed in the Oneida Thanksgiving Prayer.

> . . . Now then, I will mention . . . the water birds that swim about in the water. Now I will mention, that it is still that it can be for our good minds, good feelings, and a medicine for all people, many places the water has been polluted, and this was created by the human family, and it has created a great suffering in our minds, that we can no longer eat the fish safely, and surely it has caused great suffering to all the fish families as far as they carry on to. Now then we mention that we apologize to the waters and all her inhabitants on behalf of the humans, and pray that we will restore the waters to how it is intended to be.

Culturally, everything that the Oneida Tribe values is related to the earth's environment. The Thanksgiving Prayer implies that it is the responsibility of all Oneida people to preserve and protect their environment for future generations. The tradition of the "Seventh Generation Commitment" implies that we will honor the Creator and future generations through the protection of Mother Earth.

2.4 REGULATORY STATUS OF ASSESSMENT AREA BIRDS

Green Bay and the Lower Fox River support populations of 13 species that have been listed by the federal government or by the State of Wisconsin as either threatened or endangered (Table 2-1). The continued existence of these sensitive species in the assessment area (and the recent colonization by white pelicans) attests to the relatively undisturbed nature of the area and the quality and extent of its habitats.

2.5 BREEDING CYCLES AND LIFE HISTORIES OF ASSESSMENT AREA BIRDS

Of the birds that breed in the assessment area, colonial waterbirds, such as terns and cormorants, and bald eagles have been the most studied in terms of their PCB exposure and PCB-caused effects (see Chapter 5). All of the colonial waterbirds that breed in the assessment area are migratory, arriving at their breeding areas in the spring and leaving for their wintering areas in late fall. Thus, all of the components of their breeding cycles are contained within the short time span between approximately April and August. Figure 2-5 presents a typical breeding chronology for waterbirds within the assessment area.

The life histories of the principal colonial waterbirds of the area vary. Common and Caspian terns are exclusively ground nesters with little or no nest construction, whereas Forster's terns nest on substantial, reed nests on floating mats of vegetation. All of the terns typically nest in dense colonies with often only 2 or 3 feet between neighboring nests (Ehrlich et al., 1988). They lay clutches of two to three eggs, which are incubated over about three weeks. The young are semiprecocial in that they can leave the nest soon after hatching but depend on the adults for their food and, for the first few days after hatching, for thermoregulation (Ehrlich et al., 1988). The

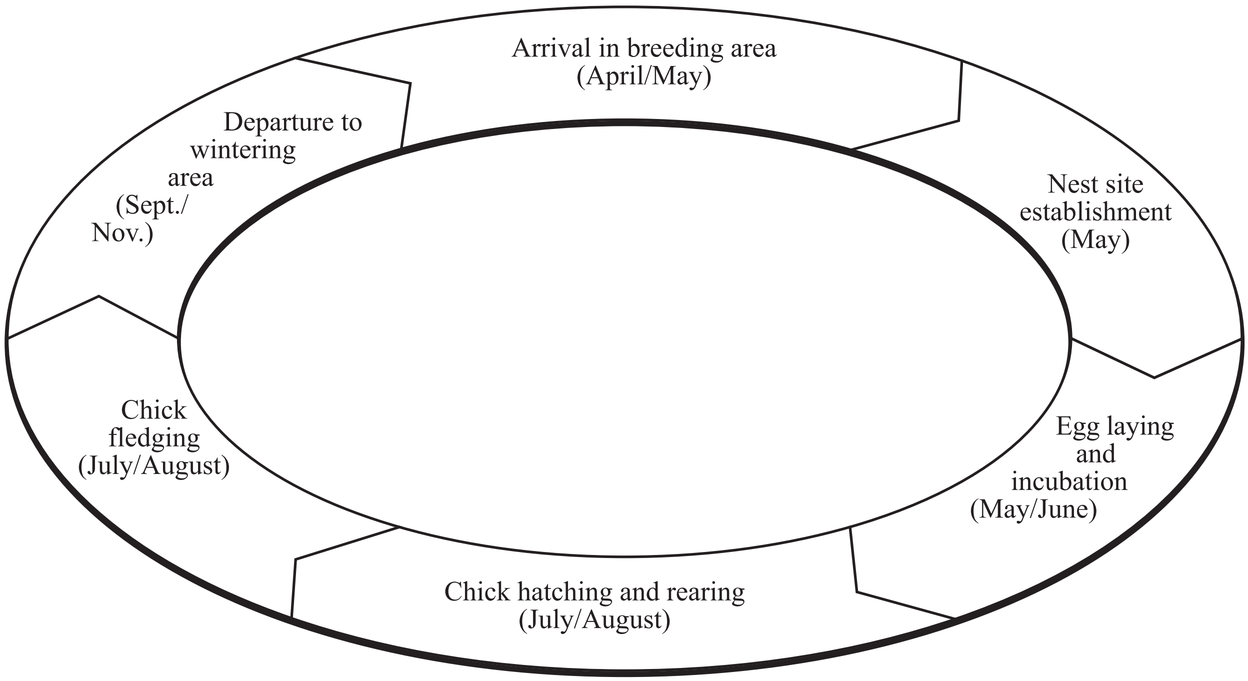


Figure 2-5. Typical breeding cycle of Green Bay colonial waterbirds.

diets of both the adults and the young are mainly forage fish (Cramp and Simmons, 1977). After about three to four weeks the young are capable of flight and leave the nesting areas in family parties. By late fall, they have left for their wintering areas in the Gulf of Mexico.

Double-crested cormorants are also summer visitors to the assessment area, arriving at their nesting colonies in April and May. They build their nests on the ground (at sites where trees are not available and mammalian predators are not a problem) or in trees (as at Cat Island, until recently). Like the terns, cormorants nest in dense colonies with only a few feet between nests. They lay three to five eggs, which are incubated for about four weeks. The nestlings are altricial, relying on their parents for their food and, in the first two weeks of life, for thermoregulation. After five to six weeks, the young are capable of flight and leave the colonies with their parents. The return to their wintering areas in the Mississippi Valley and the Gulf of Mexico begins in September. Like the three tern species, double-crested cormorants mainly eat forage fish (see Chapter 5).

Bald eagles differ substantially in their life histories from the four species discussed previously. Unlike the terns and cormorants, they are not colonial but are widely dispersed over the landscape. They may be year-round residents in the assessment area, depending on the availability of prey (Robbins, 1991). They also begin breeding much earlier in the year, with most nests being refurbished in February and March. Nests are typically bulky structures of twigs and branches and are built high in trees. Often the same nest may be used year after year. Neighboring nests may typically be up to several kilometers apart. One to three eggs are laid in March, and the young hatch after about 35 days of incubation. The young are completely altricial and dependent on their parents for food and, in the early stages of growth, for heat. Fledging takes place about 12 weeks after hatching. The prey of bald eagles comprises fish, carrion, and other birds such as herring gulls and cormorants (see Chapter 5).

2.6 CONCLUSIONS

Because of its diversity of habitats, geographical position relative to east-west and north-south gradients in bird communities, local climate patterns, and proximity to the Mississippi Flyway, the assessment area supports a rich diversity of bird species. Over 250 species have been recorded. These include breeding birds and summer visitors, fall and spring migrants, and winter visitors. Furthermore, at least 13 species that are listed by either the State of Wisconsin or the federal government as threatened or endangered are found in the assessment area.

Many of the species in the assessment area are dependent on the large tracts of relatively undisturbed habitat in the area. This is particularly true for birds that depend on wetlands or uninhabited islands for breeding, resting, and feeding sites. The assessment area provides these critical habitats in abundance.

The Oneida Reservation provides habitat for many bird species. Furthermore, birds are an important part of the Oneida Tribe culture, as reflected in oral histories and tribal prayers.

CHAPTER 3
TOXIC EFFECTS OF PCBS ON BIRDS

As described in the NRDA Assessment Plan (61 Fed. Reg. 43,558) and as noted in Chapter 1, the Fox River/Green Bay ecosystem has been contaminated with PCBs. This chapter provides an overview of the chemistry of PCBs and the toxic effects of PCBs on birds. This information provides background for the injuries assessed in subsequent chapters.

This chapter is organized as follows. Section 3.1 presents a brief overview of PCB chemistry. Section 3.2 provides a toxicological taxonomy of PCB congeners based on their mode of action. Section 3.3 discusses the effects of PCBs on birds and relates these effects to categories of injury established in the Department's NRDA regulations. Section 3.4 discusses the toxic potency of PCB congeners in birds. Section 3.5 summarizes published studies on the concentrations of PCBs shown to cause adverse effects and includes a discussion of avian sensitivity to PCB toxicity. Section 3.6 presents conclusions.

3.1 PCB CHEMISTRY

PCBs are a class of 209 chlorinated biphenyl congeners that differ in the total number and position of chlorine atoms substituted on the biphenyl structure (Figure 3-1).

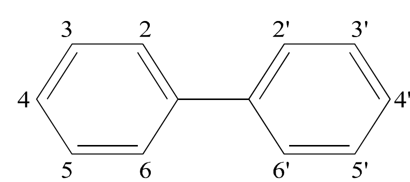


Figure 3-1. Biphenyl molecular structure.

As shown in Figure 3-1, the PCB structure is made up of two benzene (biphenyl) rings linked by a single bond. The 209 possible PCB congeners are identified by the location and number of chlorine atom substitutions on the biphenyl rings, with 10 chorine atoms being the maximum number possible. For example, the congener 2,2' dichlorobiphenyl has chlorine atoms at the 2 and 2' positions. PCB congeners are also identified by a sequential numbering system based on increasing chlorine substitution from PCB 1 (2-chlorobiphenyl; 1 chlorine) to PCB 209 (2,2',3,3',4,4',5,5',6,6'-decachlorobiphenyl; 10 chlorines).

PCBs do not occur naturally in the environment.[1] PCBs were introduced into the environment as commercial mixtures of congeners (e.g., trade names of Aroclor in the United States, Clophen in Germany, Kaneclor in Japan), with the congener composition dependent on the manufacturing process (U.S. EPA, 1980). Most commercial mixtures were differentiated by the average percentage chlorine by weight (e.g., Aroclor 1242 contained 42% chlorine). Aroclor 1242, the predominant commercial mixture involved in Fox River paper company processes (Carr et al., 1977), consists of a mixture of approximately 80 congeners (Schulz et al., 1989), with a mean number of 3.1 chlorine atoms per molecule (Eisler, 1986). Quantifiable congeners in Aroclor 1242 extend from 2,2'-dichlorobiphenyl (two chlorines; PCB 4) to 2,2',3,4,4',5,5'-heptachlorobiphenyl (seven chlorines; PCB 180) (Schulz et al., 1989).

In the environment, organisms may be exposed to mixtures of PCB congeners that no longer resemble the original commercial Aroclor. This is because physical and chemical environmental fate processes such as evaporation, transport, biodegradation, and partitioning onto sediments can alter the mixture of congeners to which biota may be exposed (Safe, 1994). Moreover, once accumulated by organisms, PCB congeners may be differentially excreted, distributed, biotransformed, or sequestered (e.g., deposition in lipids) (Rozemeijer et al., 1995; summarized by Barron et al., 1995).

3.2 TOXICOLOGICAL TAXONOMY OF PCB CONGENERS

Many individual PCB congeners have been found to cause adverse effects to biota. These effects can differ based on the specific chemical composition of the individual congener or congener mixtures. The different adverse effects caused by PCBs can be classified according to the manner in which they manifest toxicity, known as their “toxicological mode of action” (Table 3-1).

3.2.1 Dioxin-Like Toxicity

Dioxin-like congeners are known as co-planar PCBs because the biphenyl rings lie in the same two-dimensional plane, giving them a molecular configuration similar to 2,3,7,8-tetrachlorodibenzo-p-dioxin (dioxin; TCDD) (Safe, 1994). These congeners have chlorine substitutions in the 3,3',4,4' positions (meta-para substitutions) and have either zero (nonortho), one (mono-ortho), or two (di-ortho) chlorines in the 2 or 2' positions (Figure 3-2). This chlorine substitution pattern increases the structural similarity of the congeners to TCDD, inhibits metabolic transformation by organisms, and generally increases biological persistence (Safe, 1994). Dioxin-like PCBs have affinity for the same cellular receptor (the aryl hydrocarbon or Ah

1. Minor concentrations of PCBs may be produced by volcanic processes (Lamparski et al., 1990). However, data in Lamparski et al. (1990) from the Mount Saint Helen’s eruption suggest that PCBs observed in volcanic ash might be scavenged from the atmosphere rather than a product of vulcanism.

Table 3-1
Toxicological Taxonomy of PCB Congeners

Mode of Action	Toxic Effects	Example References
Dioxin-like	Edema, deformities, early life stage mortality, uroporphyrin accumulation	Safe, 1994 Bosveld, 1995 Barron et al., 1995
Phenobarbital-like	Tumor promotion, uroporphyrin accumulation	Safe, 1994 Rodman et al., 1991
Neurotoxic	Decreased dopamine, behavioral/ neuromuscular alterations	Safe, 1994 Choksi et al., 1997
Endocrine-disrupting	Estrogen mimics, metabolized to biphenylols (thyroxine mimics; Vitamin A effects)	Jansen et al., 1993 Korach et al., 1987 Walker, 1995

receptor) as TCDD. Like TCDD, dioxin-like PCBs are strong inducers of the chemical metabolizing enzyme system known as the P450 family, and exposure to a dioxin-like PCB congener can cause a substantial increase in the concentration and activity of P450 enzymes. Dioxin-like PCBs are strong inducers of the P4501A isoform, and this is the key to their toxicological characteristics. Figure 3-3 provides a simplified schematic representation of TCDD interaction with the Ah receptor showing (1) TCDD movement into a cell, (2) binding to the Ah receptor in the cytoplasm, (3) translocation of the TCDD:Ah receptor complex to the nucleus, (4) production of messenger RNA (mRNA) in the nucleus, (5) translocation of mRNA to the cytoplasm, and (6) synthesis of P450 in organelles (e.g., endoplasmic reticulum) and generation of other cellular responses.

PCBs elicit a suite of toxic effects that are similar to those of TCDD, such as pericardial and abdominal edema, and deformities of the heart, eyes, limbs, head, and body (Bosveld, 1995; Henshel et al., 1997). Although mono-ortho substituted biphenyls are less potent than nonortho congeners, they occurred at higher concentrations in commercial PCB mixtures, and thus may contribute substantially to toxicity in the environment (Braune and Norstrom, 1989; Brunstrom, 1990).

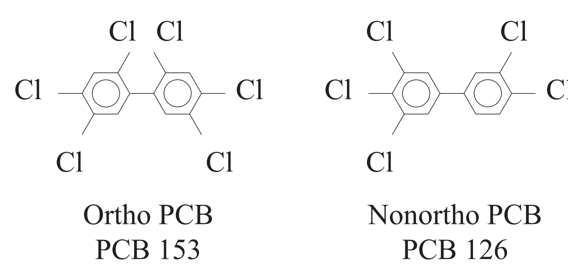


Figure 3-2. Example nonortho (coplanar) and ortho-substituted PCBs.

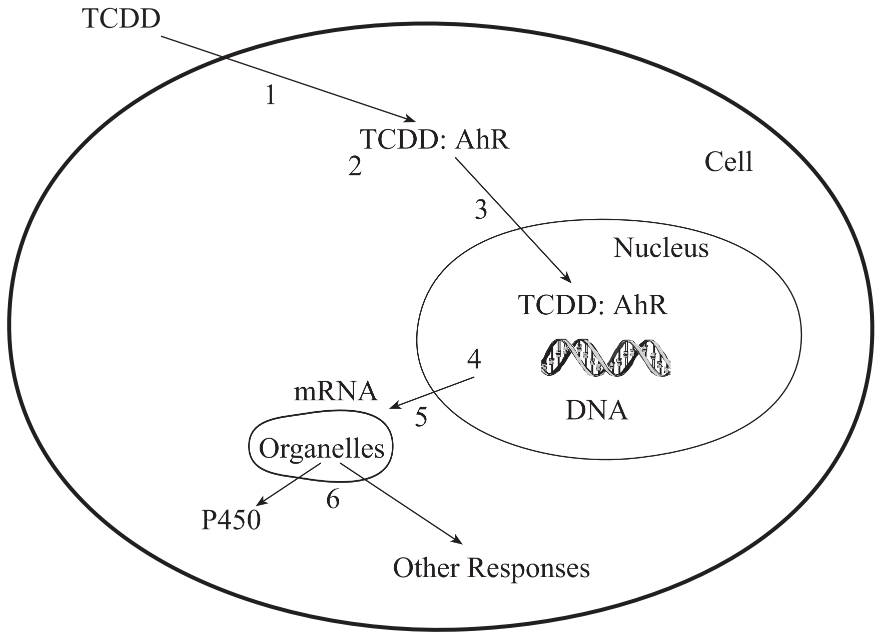


Figure 3-3. Schematic representation of TCDD interaction with the ah receptor.
(1) TCDD movement into a cell, (2) binding to the Ah receptor in the cytoplasm, (3) translocation of the TCDD:Ah receptor complex to the nucleus, (4) production of messenger RNA (mRNA) in the nucleus, (5) translocation of mRNA to the cytoplasm, and (6) synthesis of P450 in organelles (e.g., endoplasmic reticulum) and generation of other cellular responses.

Source: Simplified from Wilson and Safe, 1998.

3.2.2 Phenobarbital-Like Toxicity

The phenobarbital-like (PB-like) group of PCBs are di-ortho substituted congeners with a low affinity for the Ah receptor, and induce a different P450 isozyme (P4501B-like) in birds and other vertebrates than do the dioxin-like congeners (Safe, 1994; Van den Berg et al., 1994). The effects of PB-like PCBs include tumor promotion in rodents (Safe, 1994) and accumulation of uroporphyrin in chick embryo liver cells (hepatocytes) at high doses (Rodman et al., 1991).

3.2.3 Neurotoxicity

Neurotoxic PCBs are ortho-substituted, cause changes in neuromuscular activity and decreases in dopamine levels (Safe, 1994; Choksi et al., 1997), and are associated with behavioral changes and learning deficits (Tilson et al., 1990; Fisher et al., 1998). Heinz et al. (1980) suggested that depletion of brain neurotransmitter levels by neurotoxic PCBs may result in abnormal behavior in sensitive avian species.

3.2.4 Endocrine-Disrupting Toxicity

Current research indicates that specific PCB congeners and hydroxylated metabolites (Figure 3-4) can act as endocrine disruptors by altering normal hormonal dynamics (Sheffield et al., 1998). For example, exposure of mallards to Aroclor 1254 (20 mg/kg body weight twice per week for five weeks) caused a significant reduction in plasma levels of the thyroid hormone triiodothyronine (Fowles et al., 1997). Thyroid hormones modulate the rate of cellular metabolism (Zubay, 1983).

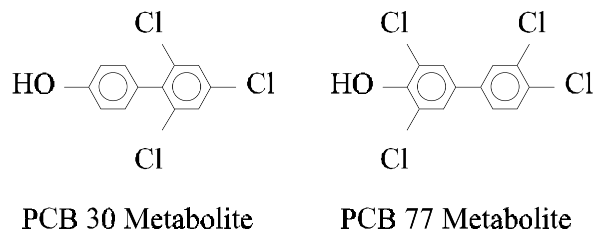


Figure 3-4. Two examples of hydroxylated PCB metabolites (biphenylols) with endocrine disrupting effects.

Biotransformation by an organism generally decreases the toxicity of PCB congeners by increasing their elimination through oxidation (e.g., hydroxylation) and/or conjugation (e.g., glutathione addition) (Figure 3-5). However, hydroxylation at specific sites on several PCB congeners produces hydroxylated biphenylol (HO group inserted into the 4 or 4' position) metabolites that can compete for and occupy hormone or vitamin binding sites (Korach et al., 1987) (Figure 3-4). These water soluble metabolites may be retained in bird eggs, exposing the developing embryo and affecting the functioning of the endocrine system (Fry, 1995). Avian endocrine disruptor effects of PCBs include hyperthyroidism (increased metabolic rate) and hypothyroidism (decreased metabolic rate) in murres (Jefferies and Parslow, 1976) and altered retinoid (vitamin A) dynamics in ring doves (Spear et al., 1989).

Fry (1995) concluded that some PCBs are estrogenic and are responsible for endocrine disruption in breeding birds and abnormalities in their offspring. However, other researchers have not found a relationship between PCBs and endocrine effects. For example, Nisbet et al. (1996) found no relationship between measured PCB congeners in common tern eggs and feminization of male embryos, although the study focused on congeners rather than their metabolites.

Estrogen mimics are thought to include PCBs 1, 9, 10, 30, 52, and 61. These PCBs have estrogenic activity (measured by increased rodent uterine weight; Jansen et al., 1993) or can be metabolized to hydroxylated biphenyls (e.g., 2',4',6'- trichloro-4-biphenylol; 2',3',4',5'-tetrachloro-4-biphenylol) with demonstrated estrogen receptor binding affinity (Korach et al., 1987). Biphenylol (one hydroxylation) and catechol (two hydroxylations) metabolites have been shown to cause endocrine disruption in several vertebrate systems (Fry, 1995; Guillette et al., 1995; Garner et al., 1999), but studies evaluating their estrogenicity in avian embryos have not been reported.

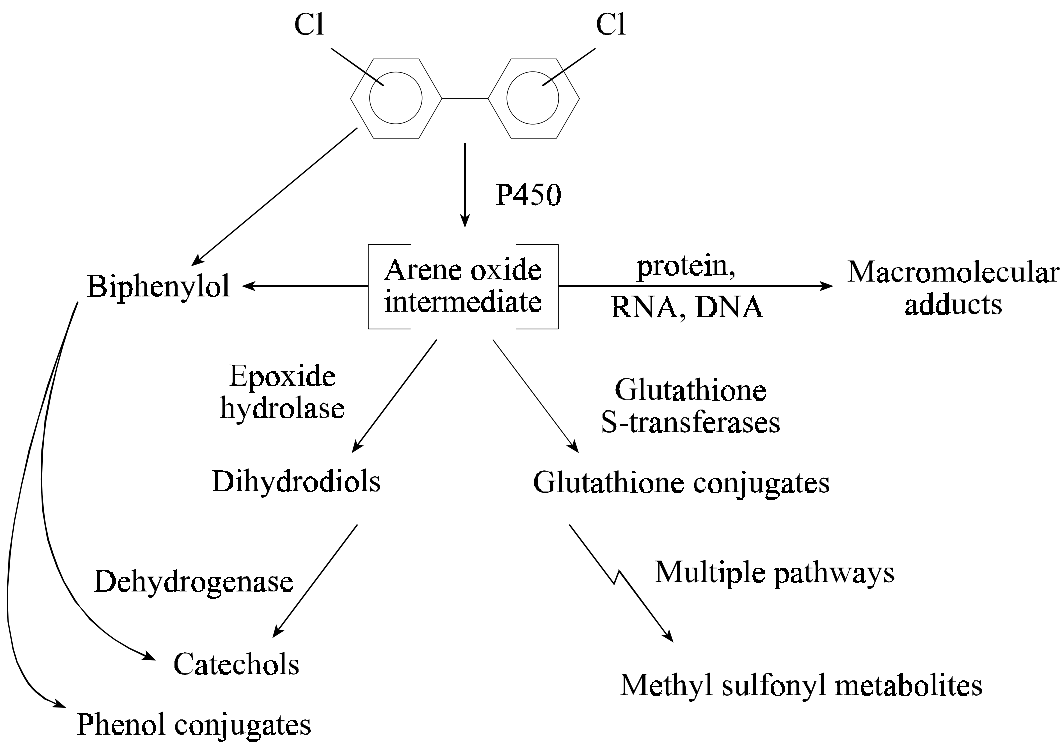


Figure 3-5. Generalized biotransformation pathway for PCBs.

Source: Safe, 1994.

In addition to these estrogen mimics, PCB 77 is thought to act as a thyroxine mimic and may modulate vitamin A levels, resulting in alteration of growth and development. This congener can be hydroxylated to biphenylol metabolites (e.g., 3,3',4,4'-tetrachloro-4-biphenylol; 3,3',4',5-tetrachloro-4-biphenylol) that compete with the binding of the thyroid hormone thyroxine to the transport protein transthyretin with subsequent alteration of growth and development (Brouwer, 1991; Walker, 1995). The activity of thyroid hormones is dependent on binding to carrier proteins in the blood, and competition with PCBs for binding sites may result in changes in metabolic rate through an increase in the "free" hormone (Zubay, 1983).

PCBs can also alter the concentrations of vitamin A in birds, which is important in vision, growth, and reproduction. Retinol is the principal natural form of vitamin A and is primarily stored in the liver as the fatty acid ester retinol palmitate (Environment Canada, 1991). Murk et al. (1994) suggested that PCBs and related contaminants may interfere with the regulation of storage and mobilization of retinoids in the livers of birds, resulting in decreased liver retinoid levels and increased plasma retinoid concentrations. Biphenylol metabolites of PCB 77 may occupy a retinol binding protein, resulting in reduction of blood vitamin A levels and thus altered growth and development (Brouwer, 1991; Walker, 1995).

3.3 PCB EFFECTS ON BIRDS

This section classifies the effects of PCBs on birds according to the injury categories identified in the Departmental NRDA regulations. These regulations define injuries to biological resources [43 CFR § 11.62 (f)], and these definitions can be used to categorize the biological effects of PCBs on birds. Table 3-2 presents, for the Department's NRDA injury categories, examples of scientific studies documenting biological effects of PCBs and concentrations causing adverse effects in birds.

3.3.1 Injury Category: Death [43 CFR § 11.62 (f)(4)(i)]

The experimental studies reported in Table 3-2 (together with many other studies not reported in Table 3-2) show that exposure to PCBs can cause death in avian embryos and juvenile and adult birds.

3.3.2 Injury Category: Disease [43 CFR § 11.62 (f)(4)(ii)]

PCBs are known to affect immune system function in mammalian systems (Safe, 1994), and may cause morphological changes in immune tissues in birds (Nikolaidis et al., 1988). Friend and Trainer (1970) reported increased mortality in mallard ducklings challenged with a duck hepatitis virus following a short-term (10 day) feeding of PCBs (25 to 100 mg/kg diet of Aroclor 1254).

3.3.3 Injury Category: Behavioral Abnormalities [43 CFR § 11.62 (f)(4)(iii)]

PCB-induced behavioral effects in birds include decreased parental incubation attentiveness in ring doves (Peakall and Peakall, 1973), and impaired courtship behavior in mourning doves (Tori and Peterle, 1983). Heinz et al. (1980) found reduced brain dopamine and norepinephrine levels in ring doves fed a 10 mg/kg diet of Aroclor 1254, and suggested that depletion of brain neurotransmitter levels may result in abnormal behavior in sensitive avian species. McCarty and Secord (1999) reported abnormal nest building behavior and lowered nest quality in tree swallows (5 to 7 mg/kg wet weight total PCBs in eggs). Subtle neurological effects such as impaired avoidance behavior in pheasants have also been reported (Dahlgren et al., 1972a).

3.3.4 Injury Category: Cancer and Genetic Mutations [43 CFR § 11.62 (f)(4)(iv)]

Long-term feeding studies of rodents have demonstrated that PCBs increase the incidence of tumors (Safe, 1994). However, our review of the literature identified only one study showing that PCBs cause tumors or genetic mutations in birds. Peakall et al. (1972) reported increased

Table 3-2
Summary of NRDA Injury Categories, Corresponding Biological Effects of PCBs on Birds, and Examples of Studies Documenting Effects

Injury Category [see 43 CFR § 11.62 (f)(4)]	Biological Response	Species Studied	Example Reference
Death	Increased adult/juvenile mortality	Mallard, pheasant, bobwhite, Japanese quail	Heath et al., 1972; Stickel et al., 1984
	Increased embryo mortality	Chicken, pheasant	Carlson and Duby, 1973; Brunstrom and Reutergårdh, 1986
Disease	Increased susceptibility to viral challenge	Mallard	Friend and Trainer, 1970
Behavioral abnormalities	Impaired mating behavior	Mourning dove	Tori and Peterle, 1983
	Impaired avoidance of visual cliff	Pheasant	Dahlgren and Linder, 1971
Cancer, genetic mutations	Chromosome alteration	Ring dove	Peakall et al., 1972
Physiological malfunctions	Reduced reproduction (reduced fecundity)	Pheasant, black- headed gull	Brunstrom and Reutergårdh, 1986
		Chicken	Carlson and Duby, 1973
		American kestrel	Lincer and Peakall, 1970
		Ring dove	Peakall et al., 1972
	Eggshell thinning	Mallard	Haseltine and Prouty, 1980
	Altered endocrine status (e.g., decreased estrogen levels)	Chicken	Chen et al., 1994
	Porphyria	Japanese quail	Elliott et al., 1990
	Enzyme induction (e.g., ED50 for P450)	Turkey	Brunstrom and Lund, 1988
Physical deformations	External malformation (e.g., small beak, eyes, unresorbed yolk sac)	Chicken, common tern, American kestrel	Brunstrom and Lund, 1988; Hoffman et al., 1998
	Skeletal deformities	Common tern, American kestrel	Hoffman et al., 1998
	Histopathological lesions	American kestrel nestlings	Hoffman et al., 1996b

chromosomal aberrations in the embryos of ring doves fed 10 mg/kg wet weight of Aroclor 1254 in their diet for three months. Chromosomal aberrations were 0.8% (range of 0 to 2%) in the control group and 1.8% (range of 0 to 9.4%) in the PCB treated group. A separate NRDA report (Barron et al., 1999) documents the increased frequency of liver tumors in fish (walleye; *Stizostedion vitreum vitreum*) exposed to elevated concentrations of PCBs in the assessment area.

3.3.5 Injury Category: Physiological Malfunctions [43 CFR § 11.62 (f)(4)(v)]

Physiological malfunctions caused by PCBs include reduced reproductive success, eggshell thinning, altered endocrine status, porphyria, altered vitamin A status, and enzyme induction (see Table 3-2).

Reduced Reproductive Success

A number of field and laboratory studies have associated PCB contamination in bird eggs with impaired reproduction, reduced fecundity and fertility, embryotoxicity, and reduced hatchling growth and development (summarized by Gilbertson et al., 1991; Barron et al., 1995; Hoffman et al., 1996a).

Eggshell Thinning

Eggshell thinning has been caused by high concentrations of PCBs in the maternal diet (summarized by Peakall and Lincer, 1996). For example, Haseltine and Prouty (1980) reported eggshell thinning (8.9% thickness reduction) in mallards fed a 105 mg/kg diet of Aroclor 1242. No effects on reproduction were observed. Peakall and Lincer (1996) concluded that PCBs do not cause significant eggshell thinning at environmentally realistic doses.

Altered Endocrine Status

The current understanding of PCB effects on bird endocrine systems is limited. However, reported avian endocrine effects of PCBs have included feminization, lowered estrogen levels, and changes in thyroid function (Colborn et al., 1993). For example, American kestrels fed a 33 mg/kg diet of Aroclor 1254 had reduced semen quality (Bird et al., 1983), and chickens fed a 250 mg/kg diet of Aroclor 1254 had significant reductions in comb and testicle weights (Platonow and Funnell, 1971). PCBs and dioxin-like compounds were associated with gonadal abnormalities in common terns, including the presence of ovarian tissue in the testes of male embryos (Hart et al., 1998). Lincer and Peakall (1970) observed a dose-dependent increase in microsomal metabolism of estradiol in American kestrels fed Aroclor 1254 or Aroclor 1262. Chickens orally administered 10 mg Aroclor 1254 daily for 5 days had reduced plasma estradiol and calcium levels, reduced egg production, decreased liver weight, and increased hepatic P450 content (Chen et al., 1994). Connor et al. (1997) reported that the estrogenicity of hydroxylated PCB congeners determined in multiple in vitro assays was complex and response-specific, with some assays indicating estrogenicity, no activity, or antiestrogenicity.

Porphyria

Porphyrins are precursors of heme (a component of hemoglobin) and normally occur in small quantities in the body (Environment Canada, 1991). Porphyria, evidenced by increased formation and excretion of porphyrins and precursors, has been reported in several avian species (Goldstein et al., 1976; Fox et al., 1988; Elliott et al., 1997). Accumulation of highly carboxylated porphyrins in herring gull livers has been linked to environmental exposures of birds to PCBs (Environment Canada, 1991). Rodman et al. (1991) demonstrated that specific PCB congeners with three or four ortho chlorines caused increased uroporphyrin in chicken embryo hepatocyte cultures, and Elliott et al. (1990) reported a significant accumulation of liver porphyrins in Japanese quail fed 0.05 mg/kg PCB 126 daily.

Altered Vitamin A (retinoid) Status

Retinol is the principal natural form of vitamin A and is primarily stored in the liver as the fatty acid ester retinol palmitate (Environment Canada, 1991). Murk et al. (1994) suggested that PCBs and related contaminants may interfere with the regulation of storage and mobilization of retinoids in the livers of birds, resulting in decreased liver retinoid levels and increased plasma retinoid concentrations. For example, ring doves exposed to PCB 77 exhibited altered retinoid dynamics, and females laying viable eggs exhibited compensatory retinoid mobilization from the liver and transfer to the eggs (Spear et al., 1989). Murk et al. (1994) concluded that hepatic and yolk sac retinoids may be suitable indicators of early effects of dioxin-like contaminants in common terns and other fish-eating birds.

Enzyme Induction

PCBs have been reported to increase the content or activity of several enzymes in birds, including P450 isozymes (Hoffman et al., 1996a). For example, planar PCBs strongly induce the P450 isozyme CYP1A [measured by increases in aryl hydrocarbon hydroxylase (AHH) or ethoxyresorufin-O-deethylase (EROD) activity]. Each isozyme has a slightly different affinity and capacity for metabolizing contaminants and endogenous biomolecules (e.g., steroids), and may be differentially induced by exposure to PCBs.

Numerous studies have reported EROD induction by dioxin-like PCBs, and some have suggested EROD induction as a sensitive measure of exposure. EROD induction may also be indicative of adverse PCB effects because P450 isozymes may activate or deactivate endogenous biomolecules (e.g., hormones). For example, Lincer and Peakall (1970) reported increased microsomal metabolism of estradiol in American kestrels fed a 0.5 mg/kg diet of Aroclor 1254. Recent in vitro studies demonstrate that EROD activity in bird tissues is suppressed at higher PCB concentrations (e.g., Lorenzen et al., 1997).

3.3.6 Injury Category: Physical Deformations [43 CFR § 11.62 (f)(4)(vi)]

Avian physical deformations caused by PCBs include external deformations, organ and tissue malformations, skeletal deformities, and histopathological lesions (Table 3-2). Physical deformations attributed to PCBs in birds include pericardial and subcutaneous edema, cardiovascular malformation, liver lesions, microphthalmia, beak and limb deformities, brain asymmetry, thymic hypoplasia, and inhibition of lymphoid development (Gilbertson et al., 1991; Bosveld and Van den Berg, 1994; Henshel, 1998; Hoffman et al., 1998). Dioxin-like PCBs may cause deformities in turkey embryos (e.g., microphthalmia and beak deformities) at egg doses lower those causing mortality**,** but may not in other bird species (Nosek et al., 1993). Additionally, exposure of birds to PCB congeners (e.g., PCB 77) and commercial mixtures (e.g., Aroclor 1254) may cause an increase in liver weights (Elliott et al., 1997).

3.4 TOXIC POTENCY OF ENVIRONMENTAL MIXTURES OF PCBS

Different PCB congeners have different potencies in producing toxic responses in birds, as do different mixtures of congeners. Since congener mixtures in the environment are complex and can vary over space, time, and environmental media, several methods have been developed to assess the toxicity of PCB mixtures. These methods are based either on applying information on the potency of individual congeners to congener concentration measurements in media, or on using bioassays to evaluate the toxic potency of the environmental mixture as a whole.

The potency of individual PCB congeners in birds can be determined by their toxicity to avian embryos (e.g., hatching success following injection into bird eggs; Brunstrom, 1990) or from the magnitude of P450 induction caused by a congener (e.g., in vitro induction in avian embryo hepatocytes; Kennedy et al., 1996a, b). Toxicity-based congener potency is derived from studies in which small quantities of graded doses of a PCB congener are injected into the yolk sac, albumin, or air cell of an egg and hatching success or some other response is measured.

Table 3-3
Relative Potency of Selected PCB Congeners in Inducing Chicken Hepatocyte P450

PCB Congener	Concentration Causing 50% P450 Induction (mM)
77	0.51
81	0.094
105	3.3
118	19
126	0.052
169	0.79

Sources: Kennedy et al., 1996b; Lorenzen et al., 1997.

Table 3-3 summarizes the potency of selected PCB congeners in inducing P450 in chicken hepatocyte cells. Of the tested congeners, PCB 126 typically is the most potent, followed by PCB 81, PCB 77, and PCB 169. A similar relative order of potency is observed in egg injection studies with embryomortality as the endpoint in different bird species (Brunstrom and Andersson, 1988; Brunstrom 1989, 1990; Powell et al., 1996). The general order of potency for congeners is nonortho > mono-ortho > di-ortho > tri-ortho > tetra-ortho.

The toxic potencies of PCB congeners are dependent on both the test species and the toxicity endpoint (e.g., P450 induction versus embryotoxicity). Responsiveness of P450 induction is dependent on the reporter system (e.g., enzyme activity, enzyme content, specific isozyme) and the assay system (e.g., rat versus avian tissue). For example, Kennedy et al. (1996a) concluded that P4501A induction in chicken hepatocyte cultures was more responsive to mono- and di-ortho-PCB congeners than the H4IIE rat hepatoma system, suggesting that birds are more sensitive to these congeners than mammals.

The potency of a PCB congener can be expressed relative to the potency of TCDD (generally the most toxic planar halogenated environmental contaminant) by estimating its toxicity equivalency factor (TEF). The TEF of a congener is determined by dividing the potency response (e.g., concentration at which 50% mortality or 50% P450 induction occurs) by the potency response of TCDD:

$$\text{TEF} = \frac{\text{Potency of PCB congener}}{\text{Potency of TCDD}}$$

Avian TEFs for 12 PCB congeners have been derived by the World Health Organization (WHO) and are summarized in Table 3-4. These values are "generic" in that they are not specific to an individual bird species. The WHO avian TEFs were derived from the results of multiple studies and a variety of experimental data (e.g., embryotoxicity studies, P450 induction), and represent the consensus opinion of an international group of toxicology experts.

TEFs can be used to calculate a TCDD equivalent concentration (TCDD-eq) of PCBs in an environmental sample. A TCDD-eq is the concentration of TCDD that has the same potency as the PCB congener mix and concentration in an environmental sample. A TCDD-eq is calculated by summing the

Table 3-4
Toxic Equivalency Factors (TEFs) of PCB Congeners in Birds Relative to TCDD

Congener Number	WHO[a] Avian TEF
77	0.05
81	0.1
105	0.0001
114	0.0001
118	0.00001
123	0.00001
126	0.1
156	0.0001
157	0.0001
167	0.00001
169	0.001
189	0.00001

a. Van den Berg et al., 1998.

product of the measured concentration and the TEF for each PCB congener measured in a sample (Giesy et al., 1994):

$$\text{TCDD-eq} = \sum ([\text{congener}]_i \times \text{TEF}_i)\ .$$

The TCDD-eq of environmental samples can also be determined experimentally using the H4IIE rat hepatoma system (e.g., Giesy et al., 1994) or the avian embryo hepatocyte system (e.g., Kennedy et al., 1996a,b). These in vitro systems generally require administration of small quantities of a chemical extract of an environmental sample to the bioassay system, with subsequent measurement of the P450 induction response. The response is then compared to the TCDD-induced response measured using the same method. Because of the apparent differences between mammals and birds in relative congener potency, a TCDD-eq measured using an avian bioassay is considered more relevant to the evaluation of the toxicity to birds than a TCDD-eq measured using a mammalian bioassay (U.S. EPA, 1998a).

There are two limitations to the TCDD-eq approach:

- ***Calculated values assume additive toxicity.*** An implicit assumption in the calculation of a TCDD-eq is that the contribution of individual congeners to the toxicity of a mixture is simply additive. However, both synergistic (more than additive toxicity) and antagonistic (less than additive) responses to PCBs and other contaminants have been reported (Petersen et al., 1993; Van den Berg et al., 1994). For example, Lorenzen et al. (1997) concluded that common terns may be more susceptible to CYP1A inducing effects of complex mixtures of dioxin-like contaminants than indicated by their response to individual contaminants. Synergistic and antagonistic interactions are not incorporated into the calculation of TCDD-eq because of uncertainty in the type and magnitude of contaminant interactions. There is some consensus (U.S. EPA, 1998a) that nonadditive effects may be relatively minor (i.e., toxicity of dioxin-like PCBs is approximately additive).

- ***Only dioxin-like toxicity is considered.*** The TCDD-eq approach generally accounts for only dioxin-like toxicity, whether determined directly from an in vitro bioassay system (e.g., EROD induction) or calculated using measured analyte concentrations and TEFs (derived from in vitro responses or acute in ovo exposures). The endpoints and modes of action used in determining TCDD-eq do not typically incorporate neurotoxicity, PB-like effects, endocrine disruption, or long-term responses such as cancer.

Despite these limitations, the TCDD-eq approach provides a generally accepted method for assessing the toxicity of mixtures of PCBs and other contaminants (U.S. EPA, 1998b).

3.5 PCB CONCENTRATIONS CAUSING TOXICITY TO BIRDS

This section summarizes literature studies on the concentrations of PCBs shown to cause adverse effects to birds. First, the sensitivity of avian life stages and species is discussed, after which toxicity studies using total PCBs dosed into bird eggs are summarized. Lastly, studies of the toxicity of individual PCB congeners, TCDD, or TCDD-eq are summarized.

3.5.1 Avian Sensitivity to PCBs

Embryos appear to be the life stage most sensitive to PCB toxicity, followed by nestlings, then adults (e.g., Hoffman et al., 1998). Embryotoxicity in birds has been studied primarily by direct injection of PCB congeners or commercial mixtures into the egg (air cell or yolk sac), and subsequent monitoring of embryo mortality and hatching success. Eggs injected after the period of organ development experience substantially less mortality and greater chick growth than eggs injected before completion of organ development (Carlson and Duby, 1973).

The relative sensitivity of the tested bird species to PCBs can be estimated by comparing concentrations of individual congeners that cause the same adverse effects. For example, Table 3-5 lists concentrations of PCB 77 and PCB 126 causing 50% embryo mortality when injected into the eggs of different bird species. The results presented in Table 3-5 show that the chicken is the most sensitive species tested to both PCB 77 and 126, with LD50 concentrations approximately two orders of magnitude less than those for other bird species. The higher sensitivity of chickens to TCDD-like toxicity has been documented in numerous studies (Eisler, 1986).

Table 3-5
PCB Congeners: Relative Species Sensitivity Based on Embryo Mortality

Species	Embryo Mortality (LD50; µg/kg egg)	
	PCB 77	PCB 126
Herring gull	>1,000[a]	—
Black-headed gull	<1,000[a]	—
Common tern	—	104[b]
Double-crested cormorant	—	158[c]
Mallard	>5,000[a]	—
Goldeneye, domestic goose	>1,000[a]	—
Kestrel	680[b]	65[b]
Pheasant; quail	100-1000[a]	—
Turkey	800[a]	—
Bobwhite	—	24[b]
Chicken	8.6[a]	0.4

a. Brunstrom and coworkers (summarized in Barron et al., 1995).
b. Hoffman et al., 1995, 1998.
c. Powell et al., 1997.

Bosveld and Van den Berg (1994) suggested that the general order of sensitivity to the embryotoxic effects of PCBs is chicken > pheasant/turkey > ducks > gulls. Kennedy et al. (1996b) similarly concluded that the general order of sensitivity was chicken > pheasant > turkey ≥ duck ≥ herring gull, based on EROD induction in primary hepatocyte cultures by a variety of planar PCBs. Brouwer (1991) concluded that herring gulls were insensitive to PCB exposure because of Ah receptor nonresponsiveness. Based on the results of egg injection studies with PCB 126, Hoffman et al. (1998) concluded that species responsiveness to P450 induction was chicken > common tern > American kestrel > bobwhite. Based on EROD induction by PCBs 77, 126, and 169 in primary hepatocyte cultures, common terns appeared to be of similar sensitivity to ducks and herring gulls, and 50 to >1600 times less sensitive than chickens (Lorenzen et al., 1997). However, common tern embryo hepatocytes were only 3.5 to 15 times less sensitive than chicken embryo hepatocytes to contaminant extracts of field collected tern eggs (Lorenzen et al., 1997). Lorenzen et al. (1997) suggested that hepatocyte cultures from common tern chicks indicated that this species may be only 6 to 79 times less sensitive than chickens, and may be more susceptible to CYP1A inducing effects of complex mixtures of dioxin-like contaminants than is indicated by their response to individual contaminants. However, common tern embryo hepatocytes were not sensitive to the commercial PCB mixture Aroclor 1254 (Lorenzen et al., 1997).

In conclusion, a limited number of bird species have been evaluated for their sensitivity to PCBs and dioxin-like toxicity. Early life stages (embryos and hatchlings) appear to more sensitive to PCB toxicity than adult or juvenile birds. Of the tested bird species, chickens are consistently the most sensitive to PCB congeners and mixtures, whereas gulls appear to be the least sensitive. The relative rankings of other tested species appear to vary based on congener (or congener mixture) and toxic endpoint studied. Most of the 87 bird species that obtain their food from the Green Bay aquatic environment have not been tested. It is likely that the sensitivity of these species varies and some may be sensitive to PCBs.

3.5.2 Adverse Effect Concentrations of PCBs

As discussed above, adverse effects of PCBs include adult and embryo mortality, impaired reproductive behavior, deformities, decreased female or male fertility, lower hatching success, impaired egg production, and reductions in population size or reproductive success. These effects have been determined in the following types of studies:

- ***Egg injection experiments.*** These studies typically use graded doses of PCBs injected into the yolk sac, air cell, or albumen of the eggs. Measurement endpoints may include embryo mortality, malformations, hatching success, and chick growth.

- ***Dietary toxicity tests.*** These studies involve administration of PCBs in the diet of the bird or by gavage. Measurement endpoints may include effects on behavior (e.g., courtship or parental attentiveness, avoidance behaviors); disease resistance; various measures of reproductive success (e.g., egg production, fertility, hatching success); and chick mortality and growth. Study duration may range from a few days to many weeks. However, studies that evaluate the effects of PCB exposure over a complete life cycle of birds have not been conducted.

- ***Field assessments.*** Field studies typically involve determining any differences in reproductive success of wild birds from contaminated sites and birds from selected reference locations. Measurement endpoints may include hatching success, chick mortality and growth, and fledgling success.

Adverse effect concentrations are typically expressed as a median effect concentration (e.g., LC50, the concentration causing 50% mortality in the test population, derived from the relationship between dose and toxic response) or as a toxicity threshold value (derived by a statistical assessment of control and treatment groups, e.g., tested concentration causing a significant increase in mortality). Toxicity thresholds are typically reported as a NOEC (no observed effect concentration) or LOEC (lowest observed effect concentration). Note that NOECs and LOECs are statistically determined, but they do not represent absolute thresholds because they are reflective of the experimental design and the doses used. For example, a LOEC of 10 mg PCB/kg egg may not represent the lowest toxicity threshold for a species because lower PCB concentrations were not tested. Traditionally, NOECs and LOECs have been used by the U.S. EPA and others to derive thresholds for chronic toxicity to protect sensitive species.

Total PCB Concentrations in Eggs Causing Toxicity

Table 3-6 presents NOECs and LOECs for total PCB concentrations in bird eggs. LOEC values are presented for the most sensitive reproductive effect measured. Excluding chickens, NOECs range upward from 1.3 mg total PCBs/kg egg (wet weight), and LOECs range upward from about 5-10 mg/kg. It should be noted that there are, apparently, large differences between species in their sensitivities to PCBs. For example, the mallard LOEC reported in Table 3-6 exceeds that of the most sensitive species, chicken, by a factor of more than 50. However, many of the study results listed in Table 3-6 may be confounded by the fact that they are based on field studies in which parameters other than PCBs (e.g., other contaminants, hatching and rearing conditions) could not be controlled.

PCB Congeners and TCDD-eq Concentrations in Eggs Causing Toxicity

Table 3-7 presents LD50, NOEC, and LOEC values for individual PCB congeners, TCDD, and TCDD-eq reported in the literature. The data presented in Table 3-7 indicate that, excluding chickens, toxicity (NOECs, LOECs) for most of the tested bird species occurs at TCDD or TCDD-eq concentrations in the range of approximately 0.2 µg/kg egg (wet weight) to 10 µg/kg. Estimated LD50s for TCDD-eq are also consistent with this range of values. However, it should be noted that because LD50s are concentrations causing effects to 50% of the test organisms; they are *not* effects *thresholds*. Effects thresholds typically will be lower.

Table 3-6
Total PCB No Observed and Lowest Observed Adverse Effect Concentrations (NOEC/LOEC) in Bird Eggs

Species	PCB	NOEC (mg/kg ww)	LOEC (mg/kg ww)	Adverse Effect	Laboratory (L) or Field (F) Study	Reference
Chicken	Total PCB	0.36	2.5	H	L	Scott, 1977
	Total PCB	0.95	1.5	H	L	Britton and Huston, 1973
	Total PCB	<5	5	P, F	L	Platonow and Reinhart, 1973
	Total PCB	—	4	D, H	L	Tumasonis et al., 1973
	A1242	0.67	6.7	G	L	Gould et al., 1997
	A1254	0.67	6.7	G	L	Gould et al., 1997
Tree swallow	Total PCB	—	5-7	B	F	McCarty and Secord, 1999
Common tern	Total PCB	7	8	S	F	Bosveld and Van den Berg, 1994
	Total PCB	4.8	[a]10	D, H	L	Hoffman et al., 1993
	Total PCB	5.2-5.6	7	H	F	Becker et al., 1993
Bald eagle	Total PCB	—	4	S	F	Ludwig et al., 1993
	Total PCB	1.3	7.2	S	F	Wiemeyer et al., 1984
	Total PCB	—	13	S	F	Bosveld and Van den Berg, 1994
Ringed turtle dove	A1254	—	16	H	L	Peakall and Peakall, 1973
Forster's tern	Total PCB	4.5	22.2	H	F/L	Kubiak et al., 1989
	Total PCB	7	[a] 19	S	F	Bosveld and Van den Berg, 1994
Caspian tern	Total PCB	—	4.2	S	F	Yamashita et al., 1993
Mallard	A1242	—	105	T	F	Haseltine and Prouty, 1980

a. Based on no apparent adverse effects in field population.

For adverse effect, A = adult mortality; B = reproductive behavior; D = deformities; F = female fertility; G = chick growth; H = hatching success; M = male fertility; P = egg production; S = population size or reproductive success; T = egg shell thinning. Data are organized by the general rank order of sensitivity (most to least sensitive species based on reported NOECs and LOECs).

Table 3-7
No Observed and Lowest Observed Adverse Effect Concentrations (NOEC/LOEC) and Median Lethal Concentrations (LD50s) for PCB Congeners, TCDD, and TCDD-eq (µg/kg egg wet weight) in Birds

Species	Toxicant	Measurement	Reported µg/kg egg (ww)	TCDD-eq[a] Toxicity Value	TCDD-eq[a] µg/kg egg (ww)	Adverse Effect	Laboratory (L) or Field (F) Study	Reference
Chicken	TCDD	NOEC	0.1	NOEC	0.1	H	L	Janz and Bellward, 1996
			0.2	NOEC	0.2	I	L	Peden-Adams et al., 1998
		LD50	0.15	LD50	0.15	H	L	Powell et al., 1996
	PCB 77	LD50	8.6	LD50	0.43	H	L	Brunstrom and Andersson, 1988
	PCB 105	LD50	2200	LD50	0.22	H	L	Brunstrom, 1990
	PCB 118	LD50	8000	LD50	0.08	H	L	Brunstrom, 1989
	PCB 126	LD50	3.2	LD50	0.32	H	L	Brunstrom and Andersson, 1988
		LD50	2.3	LD50	0.23	H	L	Powell et al., 1996
		LD50	0.4	LD50	0.04	H	L	Hoffman et al., 1995
	PCB 156	LD50	1500	LD50	0.15	H	L	Brunstrom, 1990
	PCB 157	LD50	2500	LD50	0.25	H	L	Brunstrom, 1990
	PCB 167	LD50	>4,000	LD50	>0.04	H	L	Brunstrom, 1990
	PCB 169	LD50	170	LD50	0.17	H	L	Brunstrom and Andersson, 1988
Osprey	TCDD-eq	NOEC	0.14	NOEC	0.14	S	F	Woodford et al., 1998
Bald eagle	TCDD-eq	NOEC	0.2	NOEC	0.2	S	F	Elliott et al., 1996

Table 3-7 (cont.)
No Observed and Lowest Observed Adverse Effect Concentrations (NOEC/LOEC) and Median Lethal Concentrations (LD50s) for PCB Congeners, TCDD, and TCDD-eq (µg/kg egg wet weight) in Birds

Species	Toxicant	Measurement	Reported µg/kg egg (ww)	TCDD-eq[a] Toxicity Value	TCDD-eq[a] µg/kg egg (ww)	Adverse Effect	Laboratory (L) or Field (F) Study	Reference
Bobwhite	PCB 126	LD50	24	LD50	2.4	H	L	Hoffman et al., 1995
Caspian tern	TCDD-eq	NOEC	0.75	NOEC	0.75	H	F	Ludwig et al., 1993
Domestic pigeon	TCDD	LOEC	3	LOEC	3	G, H	L	Janz and Bellward, 1996
Eastern bluebird	TCDD	NOEC	1	NOEC	1	B	F	Thiel et al., 1988
		LOEC	10	LOEC	10	B	F	
Common tern	PCB 126	LD50	104	LD50	10.4	H	L	Hoffman et al., 1998
	TCDD-eq	NOEC	<4	NOEC	<1 (assuming 25% lipid)	H	L	Bosveld and Van den Berg, 1994
Double-crested cormorant	PCB 126	LD50	158	LD50	16	H	L	Powell et al., 1997
	TCDD	LD50	4	LD50	4	H	L	Powell et al., 1997
	TCDD-eq	LD50	~0.55	LD50	0.55	H	F	Tillitt et al., 1992

Table 3-7 (cont.)
No Observed and Lowest Observed Adverse Effect Concentrations (NOEC/LOEC) and Median Lethal Concentrations (LD50s) for PCB Congeners, TCDD, and TCDD-eq (µg/kg egg wet weight) in Birds

Species	Toxicant	Measurement	Reported µg/kg egg (ww)	TCDD-eq[a] Toxicity Value	TCDD-eq[a] µg/kg egg (ww)	Adverse Effect	Laboratory (L) or Field (F) Study	Reference
Forster's tern	TCDD-eq	NOEC	2.2	NOEC	2.2	H	L/F	Kubiak et al., 1989
Great blue heron	TCDD	NOEC	2	NOEC	2	H	F	Janz and Bellward, 1996
	TCDD-eq	NOEC	0.02	NOEC	0.02	G, D	F	Hart et al., 1991
		LOEC	0.245	LOEC	0.245			
Ring-necked pheasant	TCDD	NOEC	1 (yolk sac injected)	NOEC	1	H	L	Nosek et al., 1993
		LOEC	1 (albumen injected)	LOEC	1	H	L	
	77	NOEC	100	NOEC	5	H	L	Brunstrom and Reutergårdh, 1986
Wood duck	TCDD-eq	NOEC	≤5	NOEC	≤5	H	F	White and Hoffman, 1995
		LOEC	>20-50	LOEC	>20-50	H	F	
American kestrel	PCB 77	LD50	680	LD50	34	H	L	Hoffman et al., 1998
	PCB 126	LD50	65	LD50	6.5	H	L	
Turkey	PCB 77	LD50	~800	LD50	40	H	L	Brunstrom and Lund, 1988
Black-headed gull	PCB 77	LD50	<1000	LD50	<50	H	L	Brunstrom, 1988

Table 3-7 (cont.)
No Observed and Lowest Observed Adverse Effect Concentrations (NOEC/LOEC) and Median Lethal Concentrations (LD50s) for PCB Congeners, TCDD, and TCDD-eq (µg/kg egg wet weight) in Birds

Species	Toxicant	Measurement	Reported µg/kg egg (ww)	TCDD-eq[a] Toxicity Value	TCDD-eq[a] µg/kg egg (ww)	Adverse Effect	Laboratory (L) or Field (F) Study	Reference
Herring gull	PCB 77	LD50	>1000	LD50	1-2	H	L	Brunstrom, 1988
	TCDD-eq	NOEC	1-2	NOEC	1-2	H	F	Ludwig et al., 1993
Domestic goose	PCB 77	LD50	>1000	LD50	>50	H	L	Brunstrom, 1988
Goldeneye	PCB 77	LD50	>1000	LD50	>50	H	L	Brunstrom and Reutergårdh, 1986
Mallard	PCB 77	LD50	>5000	LD50	>250	H	L	Brunstrom, 1988

a. Calculated using WHO TEFs (U.S. EPA, 1998b).

For adverse effect, A= adult mortality; B = reproductive behavior; D = deformities; F = female fertility; G = chick growth; H = hatching success; I = immunological changes; M = male fertility; P = egg production; S = population size or reproductive success; T = egg shell thinning. Data are organized by the general rank order of sensitivity as TCDD-eq toxicity values (most to least sensitive species).

3.6 CONCLUSIONS

The results of this review support the following conclusions.

- ***PCBs cause a number of adverse effects in birds that meet the NRDA definitions of injury.*** PCB-caused adverse changes in viability in birds can include death, disease, behavioral abnormalities, physiological malfunctions, and physical deformities. Increasing PCB exposure results in an increase in the number and severity of effects from EROD induction to embryotoxicity to adult mortality (Figure 3-6).

  Laboratory and field studies have shown that exposure of birds to PCBs causes a suite of toxic effects:

  - At high doses, PCBs may cause death in adult and juvenile birds (summarized by Prestt et al., 1970; Dahlgren et al., 1972b; Barron et al., 1995).

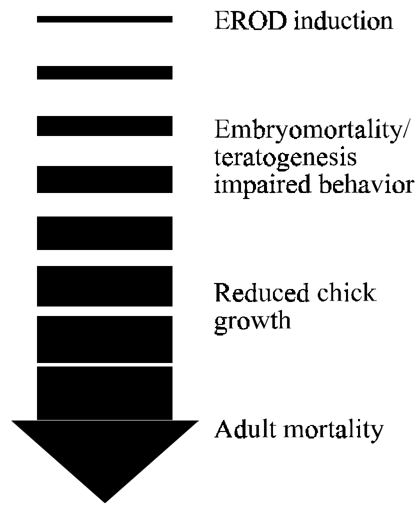


Figure 3-6. Increasing PCB exposure results in an increase in the number and severity of adverse effects in birds.

  - At lower exposures, PCBs may cause death in avian embryos (Barron et al., 1995).

  - Sublethal effects of PCBs can include reproductive and developmental toxicity: (1) altered reproductive behavior, (2) reduced fertility and egg production, (3) reduced or delayed chick growth. PCB effects also include subtle neurological effects such as impaired avoidance behavior (Dahlgren et al., 1972b).

  - P450 induction is the most consistently sensitive in vitro measure of PCB exposure, but P450 activity (e.g., EROD) can be inhibited at higher exposure concentrations (Lorenzen et al., 1997).

  - Eggshell thinning does not appear to be caused by PCB exposure (Peakall and Lincer, 1996), except at high dietary concentrations (e.g., 105 mg/kg wet weight; Haseltine and Prouty, 1980).

  - Depending on the dose and exposure scenario, PCBs and related contaminants may act as estrogen or thyroxine agonists or antagonists or may alter circulating hormone levels (McKinney et al., 1985; Gilbertson et al., 1991).

- Field studies have associated PCB exposure in birds with increased EROD, decreased thyroxine in plasma, decreased hepatic retinoid levels, increased relative liver weight, decreased head and femur size in hatchlings, reduced embryo growth, and delayed hatching (Hoffman et al., 1986; Van den Berg et al., 1994; Bosveld et al., 1995).

▸ ***PCBs in eggs cause toxicity at low parts-per-million concentrations of total PCBs (Table 3-6):***

- Although there is much variability in species sensitivity, toxicity thresholds for total PCBs in the eggs of sensitive bird species range upward from 5 to 10 mg/kg egg, resulting in reproductive malfunctions, embryo mortality, and embryo deformities.

▸ ***PCBs in eggs cause toxicity at low, or sub parts per billion, concentrations as TCDD-eq in eggs (Table 3-7):***

- Toxicity thresholds for TCDD-eq in the eggs (NOECs, LOECs) of many bird species range from 0.2 to 10 µg/kg egg, resulting in reproductive malformations, embryo mortality, and deformities.

In conclusion, PCBs cause multiple adverse effects in bird species, including death, deformities, and reproductive malfunctions. Low mg/kg wet weight concentrations of total PCBs in eggs and low ppb concentrations of TCDD-eq in eggs can cause embryo mortality, malformations, and impaired reproduction.

CHAPTER 4
PCB EXPOSURE IN ASSESSMENT AREA BIRDS

4.1 INTRODUCTION

This chapter presents information on the exposure of assessment area birds to PCBs. Exposure is characterized using data on PCB concentrations measured in the tissues of bird species nesting in the Fox River/Green Bay assessment area. The purpose of this chapter is to determine if assessment area birds have been exposed to PCBs and the likelihood that their exposure levels may have been sufficient to result in injuries. The occurrence of injuries to birds in the assessment area is assessed in the following chapters of this report.

Our approach in this chapter is as follows:

- We compare PCB tissue residues in assessment area birds to those from reference areas to determine whether assessment area birds have been exposed to elevated PCB concentrations.
- We evaluate PCB tissue residues in assessment area birds over time to characterize temporal trends in PCB exposure.
- We compare PCB tissue residues in assessment area birds to ranges of PCB concentrations shown to cause toxicity in laboratory or field studies. These comparisons provide information regarding the likelihood that PCBs have caused adverse effects in assessment area birds.

Figure 4-1 depicts the pathways by which assessment area birds can be exposed to PCBs. Because PCBs accumulate in biota and "biomagnify" in the food chain, the dietary pathway is the primary route by which birds are exposed. Also, of the birds that nest and feed on and near the assessment area, piscivorous species (i.e., those that consume fish) and predatory species (i.e., those that consume other birds) are expected to be most highly exposed to PCBs, since their food items are more highly contaminated with PCBs.

Exposure analysis is a fundamental component of pathway determination. The NRDA regulations indicate that confirmation of biological pathways can be characterized by direct measurement of the hazardous substance in tissues of exposed organisms [43 CFR §11.63(f)(4)(i)]. Thus, using measurements of PCBs accumulated in bird tissue is the most direct method of confirming exposure, as it takes into account such factors as contaminant bioavailability, foraging areas,

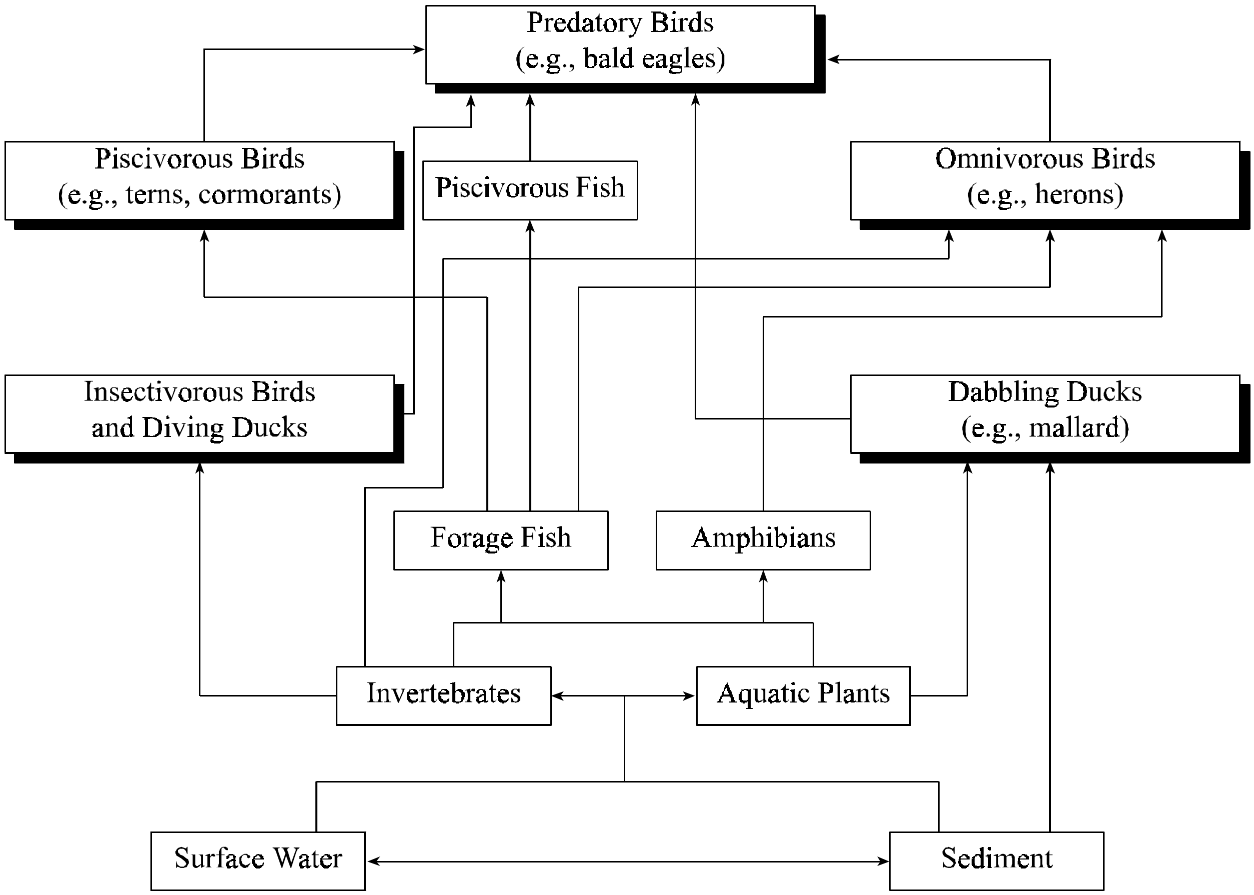


Figure 4-1. Generalized PCB exposure pathways for assessment area birds.

and contamination of prey items. More detailed pathway evaluations for each bird species assessed are presented in Chapter 5.[1]

Numerous studies have been conducted on PCB concentrations in Green Bay bird tissues. Figure 4-2 shows the locations in the assessment area at which bird tissues have been collected for PCB analysis, and Table 4-1 lists the species and tissues (egg, adult, or chick) that have been measured. Table 4-1 shows that more PCB concentration data are available for eggs than for adult or chick tissue, and since egg PCB data can be used to assess the potential for embryo toxicity (a sensitive PCB toxic endpoint), this chapter focuses on PCB concentration data in eggs.

1. The pathways by which PCBs are transported from Fox River paper company facility sources through the Fox River, into Green Bay, and into assessment area fish tissue are described in a separate NRDA report.

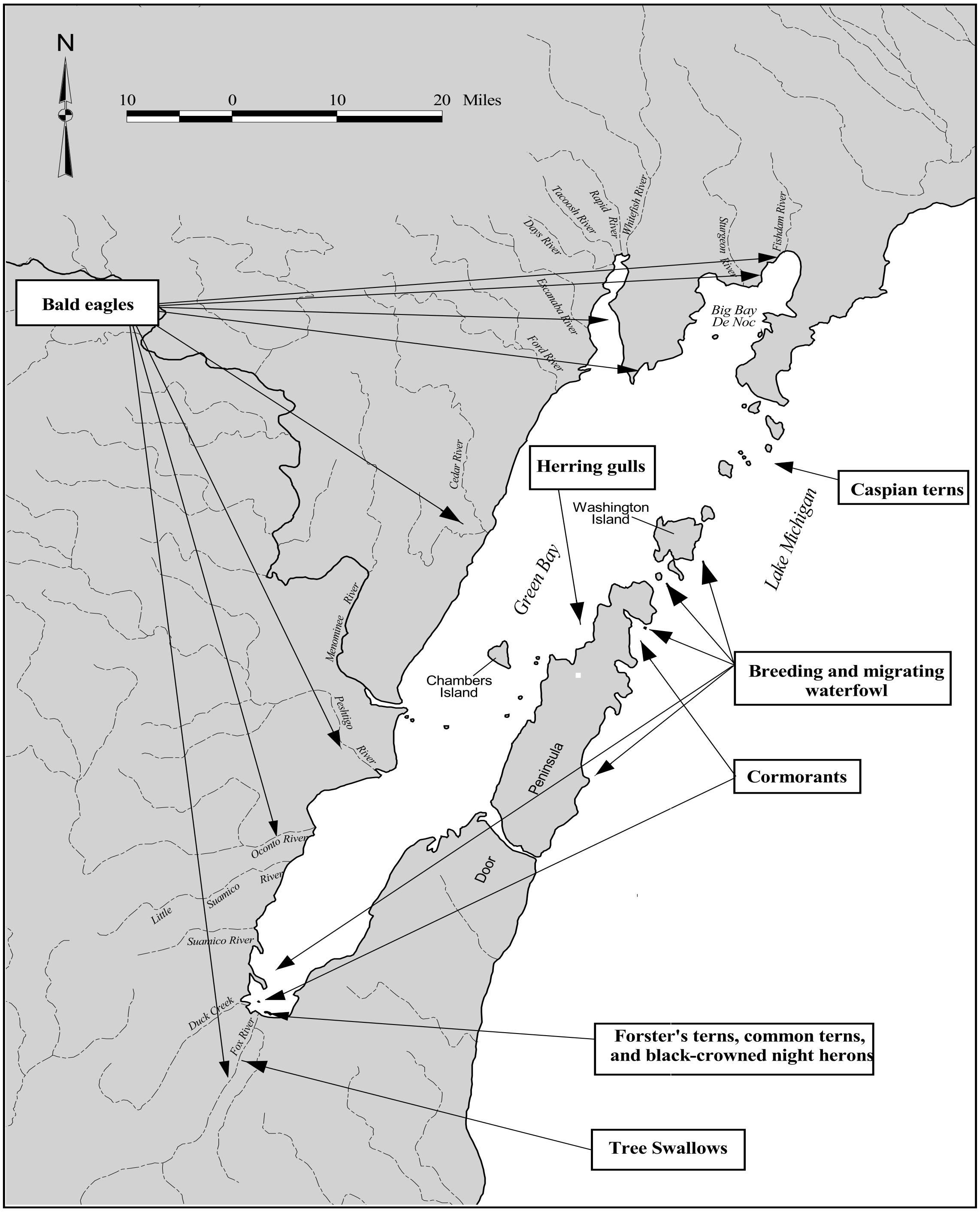


Figure 4-2. Selected locations at which bird tissues have been collected for PCB analysis.

Table 4-1
Green Bay Assessment Area Birds and Their Life Stages in Which PCBs Have Been Measured

	Eggs/Embryos	Adults	Chicks
Double crested cormorant	✓[a]	✓[b]	✓[c]
Black-crowned night heron	✓[d]		✓[d]
Green heron	✓[e]		
Canada goose	✓[b]		
Mallard	✓[b]	✓[b]	
Pintail		✓[b]	
Gadwall	✓[b]	✓[b]	
Lesser scaup		✓[q]	
Greater scaup		✓[q]	
Common goldeneye		✓[q]	
Ruddy duck		✓[q]	
Bufflehead		✓[q]	
White-winged scoter		✓[q]	
Canvasback		✓[q]	
Common merganser		✓[f]	
Red-breasted merganser	✓[g]	✓[q]	
Bald eagle	✓[h]		✓[i]
Herring gull	✓[j]		
Ring-billed gull	✓[e]		
Little gull	✓[e]		
Common tern	✓[e]		✓[k]
Forster's tern	✓[l]		✓[m]
Caspian tern	✓[n]	✓[o]	
Black tern	✓[e]		
Tree swallow	✓[p]		✓[p]
Red-winged blackbird	✓[k]		
Yellow-headed blackbird	✓[r]		✓[r]
Marsh wren	✓[s]		

a. Heinz et al., 1985; Tillitt et al., 1992; Williams et al., 1995a; Larson et al., 1996; T. Custer, USGS, pers. comm., 1998; Custer et al., in press.
b. USFWS, 1993.
c. Custer et al., 1997.
d. Custer and Custer, 1995; Rattner et al., 1993.
e. Heinz et al., 1985; Ankley et al., 1993; USFWS, 1993; Hoffman et al., 1993; Stratus Consulting unpublished data.
f. Amundson, undated.
g. White and Cromartie, 1977; Haseltine et al., 1981; USFWS, 1993; Williams et al., 1995b; Heinz et al., 1994.
h. Dykstra and Meyer, 1996.
i. W. Bowerman, Lake Superior State University, pers. comm., June 1998.
j. Bishop et al., 1992; Pekarik et al., 1998.
k. Ankley et al., 1993.
l. Heinz et al., 1985; Kubiak et al., 1989; USFWS, 1993; Harris et al., 1993; Jones et al., 1993; Stratus Consulting unpublished data.
m. Harris et al., 1993.
n. Yamashita et al., 1993.
o. Mora et al., 1993.
p. C. Custer et al., 1998.
q. USFWS, unpublished data.
r. Rattray, 1997.
s. B. Harris, University of Wisconsin, Green Bay, pers. comm., 1999.

4.2 COMPARISON OF ASSESSMENT AREA BIRD PCB CONCENTRATIONS TO REFERENCE AREA CONCENTRATIONS

In many of the studies of PCB concentrations in assessment area birds, the investigators also collected and analyzed eggs from reference areas. Within each study, similar collection, handling, storage, preparation, and analysis methods were used for both the assessment area and reference areas. Therefore, these studies can be used for direct comparison of PCB concentrations between the assessment area and reference areas.

Table 4-2 presents a comparison of bird tissue PCB concentrations in assessment and reference areas from studies in which both were measured. The table shows that for all species and studies where a statistical comparison was made between PCB concentrations in assessment and reference area tissues, concentrations were significantly greater in tissues from the assessment area. Mean assessment area PCB concentrations were up to approximately eight times greater than reference area concentrations for species such as double-crested cormorant, black-crowned night heron, and bald eagle. PCB concentrations in other species were two to five times greater in the assessment area than in reference areas.

Many of the studies listed in Table 4-2 used different reference areas for comparison. Reference areas used in studies of Caspian terns, common terns, and herring gulls are in northern Great Lakes areas where no PCB point sources such as those of the Fox River paper companies are present. Reference areas used in studies of Forster's terns, mallards, bald eagles, tree swallows, and red-winged blackbirds represent PCB exposure in inland Wisconsin. Reference areas used in studies of black-crowned night herons and double-crested cormorants are distant from the Great Lakes, reflecting lower PCB exposure in areas not influenced by Great Lakes PCB releases. Regardless of the reference area used, Table 4-2 demonstrates that PCB concentrations in birds from the Fox River/Green Bay assessment area exceed those in the reference areas.

4.3 TEMPORAL TRENDS IN PCB EXPOSURE

Because releases of PCBs into the Fox River/Green Bay environment have not been constant over time, exposures of assessment area birds to PCBs have also varied over time. Characterizing temporal trends in PCB exposure of assessment area birds helps define the time span over which injuries have occurred. The Canadian Wildlife Service has collected herring gull eggs from Big Sister Island in Green Bay (along the eastern shore; see Figure 2-4) almost every year since 1972 as part of regular monitoring of contaminant concentrations in the Great Lakes (Bishop et al., 1992; Pettit et al., 1994; Hughes et al., 1998; Pekarik et al., 1998). This dataset is the most complete dataset available with which to evaluate temporal trends in Green Bay bird PCB exposure.

Figure 4-3 plots the PCB concentrations measured in Big Sister Island herring gull eggs from 1971 through 1996 (Hughes et al., 1998). Also included in the plot is an estimate of PCB

Table 4-2
Comparison of PCB Concentrations in Bird Tissues from the Assessment and Reference Areas

Species	Tissue	Mean PCB Concentration (mg/kg wet weight, wet weight)		Statistically Higher in Assessment Area?[a]	Reference Area	Source
		Assessment Area	Reference Area			
Double-crested cormorant	Egg	7.8	1.0	Yes	Lake Winnipegosis, Manitoba	Larson et al., 1966
	Egg	5.3-14.8	0.8	Yes	Lake Winnipegosis, Manitoba	Tillitt et al., 1992
Black-crowned night heron	Whole-body	9.3	1.1	Yes	Chincoteague NWR, Virginia	Rattner et al., 1993
Herring gull	Egg	104.2	51.4	Yes	Lake Superior	Bishop et al., 1992
Forster's tern	Egg	19.2	4.6	Yes	Lake Poygan, WI	Kubiak et al., 1989
Common tern	Egg	10.0	4.0-4.7	Yes	N. Lake Michigan	Hoffman et al., 1993
Caspian tern	Egg	36.2	18.5-30.9	Yes	N. Lake Huron	Struger and Weseloh, 1985
	Egg	10.8	5.6-10.0	NA	N. Lake Huron	Yamashita et al., 1993
	Egg	15.8	8.6-14.5	NA	N. Lake Huron	Ewins et al., 1994
	Plasma	3.5	1.0-1.4	Yes	N. Lake Huron	Mora et al., 1993
Mallard	Muscle	0.43	0.19	Yes	Inland Wisconsin	Amundson, undated
Bald eagle	Egg	35	4.3	Yes	Inland Wisconsin	Dykstra and Meyer, 1996
Tree swallow	Egg	3.2	0.3	Yes	Lake Poygan, WI	Custer et al., 1998
Red-winged blackbird	Egg	1.1	0.3	NA	Inland Wisconsin	Ankley et al., 1993

a. Statistical significance as reported by study authors. In all cases significance was determined at $p < 0.05$.

NA = study authors did not conduct statistical tests, and raw data are not available to use in conducting tests.

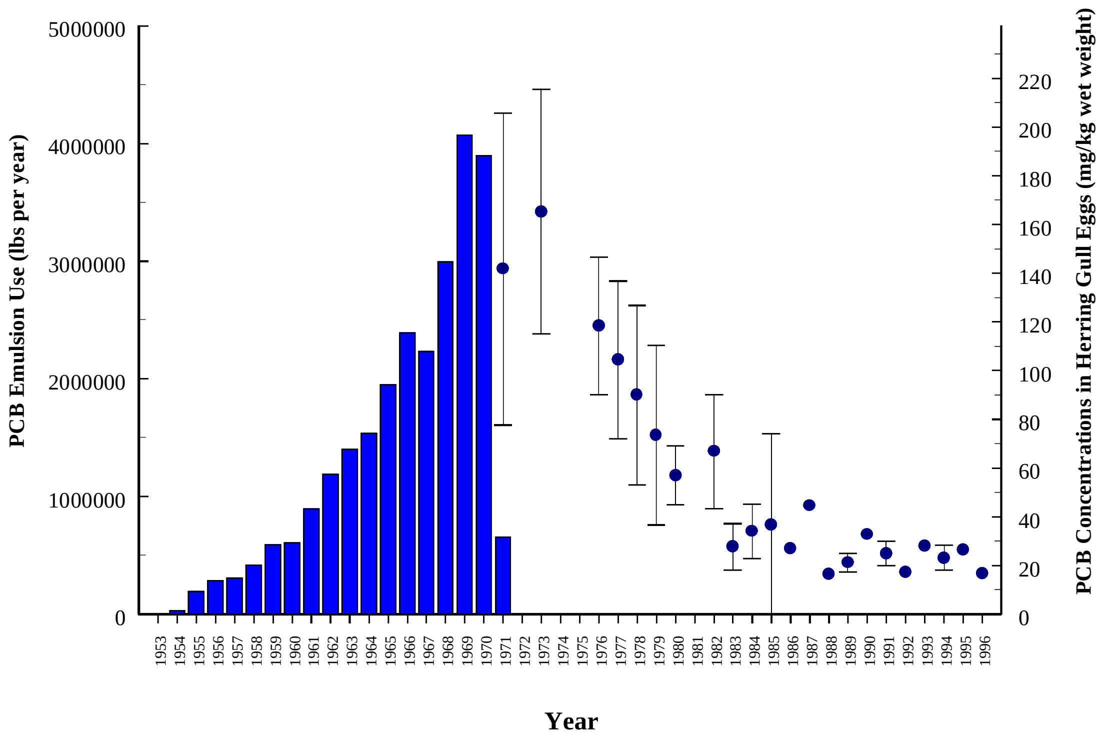


Figure 4-3. Estimated PCB emulsion use at Appleton Paper (solid bars) and mean PCB concentrations in herring gull eggs (plus or minus 1 SD) from Big Sister Island, Green Bay. PCB emulsion use is expected to closely match the temporal pattern of direct PCB discharges from paper companies into the Fox River (G. Amendola, Amendola Engineering, Inc., personal communication, 1999). PCB emulsion use data from G. Amendola (personal communication, 1999). Herring gull data from Hughes et al. (1998).

emulsion use by Appleton Paper, which is on the Fox River. The timeframe for direct PCB discharges into the Fox River is expected to closely match the timeframe of PCB emulsion use by Appleton Paper (G. Amendola, Amendola Engineering, Inc., personal communication, 1999). Direct PCB releases increased from 1954 to 1969 and dropped dramatically from 1970 to 1971, when PCB use in carbonless copy paper was discontinued (G. Amendola, Amendola Engineering, Inc., personal communication, 1999).[2] PCB concentrations in Big Sister Island herring gull eggs were highest when they were first measured in the early 1970s, with mean concentrations of approximately 170 mg/kg wet weight. Concentrations dropped from the early 1970s through the mid-1980s, reaching a mean concentration of approximately 30 mg/kg wet weight. Since the mid-

2. Direct releases into the Fox River did continue after 1971, although the estimated mass of PCBs released was much less than that released before 1971 (Wisconsin DNR, 1998a). In addition, re-releases of PCBs from contaminated river and bay sediments continue (DePinto et al., 1994).

1980s, PCB concentrations have stabilized or are declining only very slightly, with concentrations varying within the approximate range of 15 to 40 mg/kg wet weight.

Figure 4-4 shows the Big Sister Island herring gull egg PCB data broken into two time segments: 1971 to 1982, and 1983 to 1996. A comparison of the two plots shows that before 1983, PCB concentrations were clearly declining. Since approximately 1983, the decline has reached a plateau, although there is an almost significant negative trend ($r = 0.5$, $p = 0.07$).

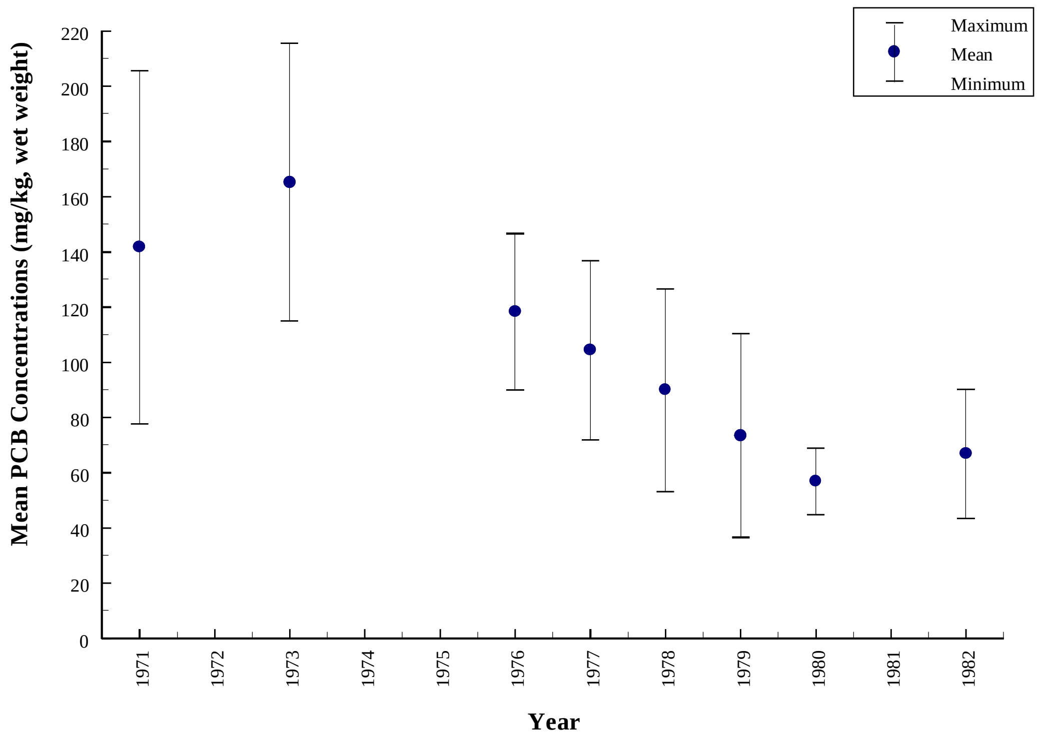
Comparison of the PCB loadings to the herring gull data (Figure 4-3) indicates that the decline in herring gull PCB concentrations followed the sharp reduction in paper company releases in 1971. Following the decline in paper company direct releases, herring gull PCB concentrations decreased, although not as rapidly. Since PCBs do not degrade readily in the environment (Erickson, 1997), they remain in the system for many years following initial release. This is reflected in the fact that the decline in herring gull PCB concentrations lagged behind the drop in loadings to the system, and that since the initial decline, concentrations have stabilized or are declining at a much lower rate. Figures 4-3 and 4-4 indicate that it took approximately 15 years for PCB concentrations in Green Bay herring gulls to respond fully to the sharp reductions in direct PCB loadings. Following the initial 15-year response, PCB concentrations have now stabilized at levels that reflect a state where direct PCB loadings are much lower than in the past, but PCBs stored in the system continue to result in exposure to biota.

Piscivorous birds in the assessment area, including herring gulls, feed on a variety of forage fish species (Ludwig and Ludwig, undated report a; Ewins et al., 1994). Figures 4-5 and 4-6 show PCB concentrations measured over time in yellow perch and alewife, respectively, in Green Bay. These forage fish data, although not as complete as the herring gull data, show a temporal pattern similar to that observed in Green Bay herring gulls. PCB concentrations were higher in the 1970s and have remained relatively constant or have declined slightly since the mid-1980s. Therefore, PCB exposure and accumulation for other assessment area piscivorous birds is expected to follow the pattern observed for herring gulls: a decline through the 1970s until approximately the early 1980s, and a stabilization of PCB concentrations since then. For example, Figure 4-7 shows such a pattern for red-breasted mergansers in the assessment area.

4.4 COMPARISON TO TOXIC EFFECTS RANGES

In this section, PCB concentrations measured in eggs of Green Bay birds are compared with toxicity threshold values obtained from the literature. The purpose of the comparison is to evaluate whether the PCB concentrations measured in Green Bay bird eggs are at or above concentrations shown to cause adverse effects. Because of uncertainties in applying toxic thresholds obtained from literature studies, such as differences in species studied, mode and timing of PCB dosing, differences in environmental mixtures of PCBs, toxic endpoints examined,

(a)



(b)

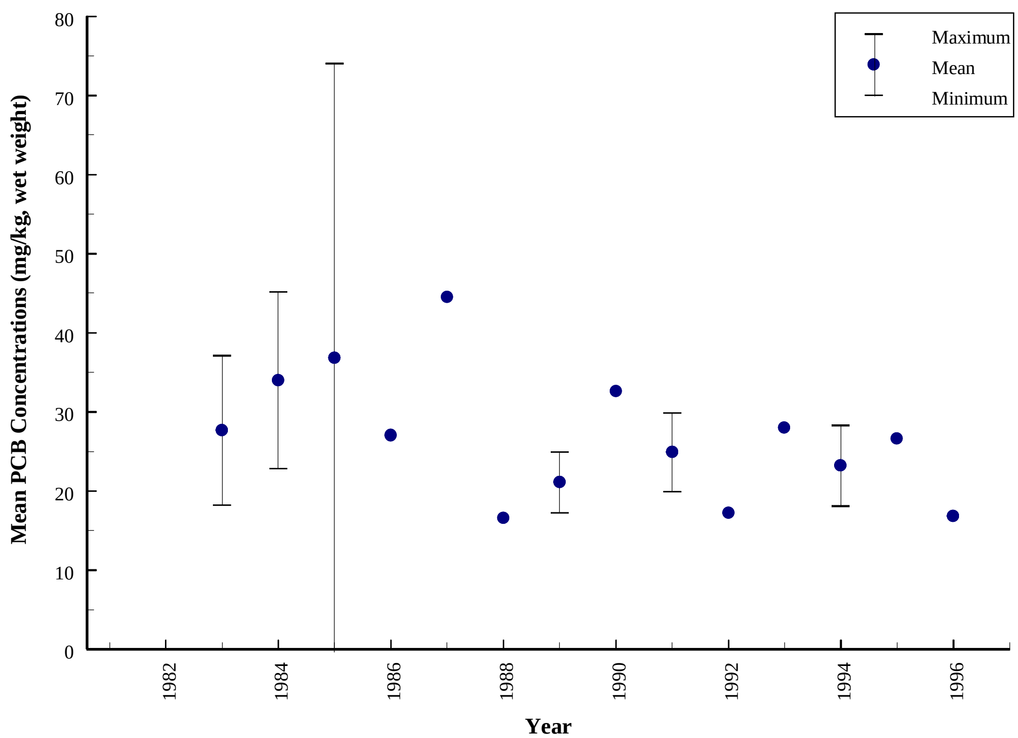


Figure 4-4. Mean and maximum and minimum PCB concentrations in Big Sister Island (Green Bay) herring gull eggs from (a) 1971 to 1982 and (b) 1983 to 1996.

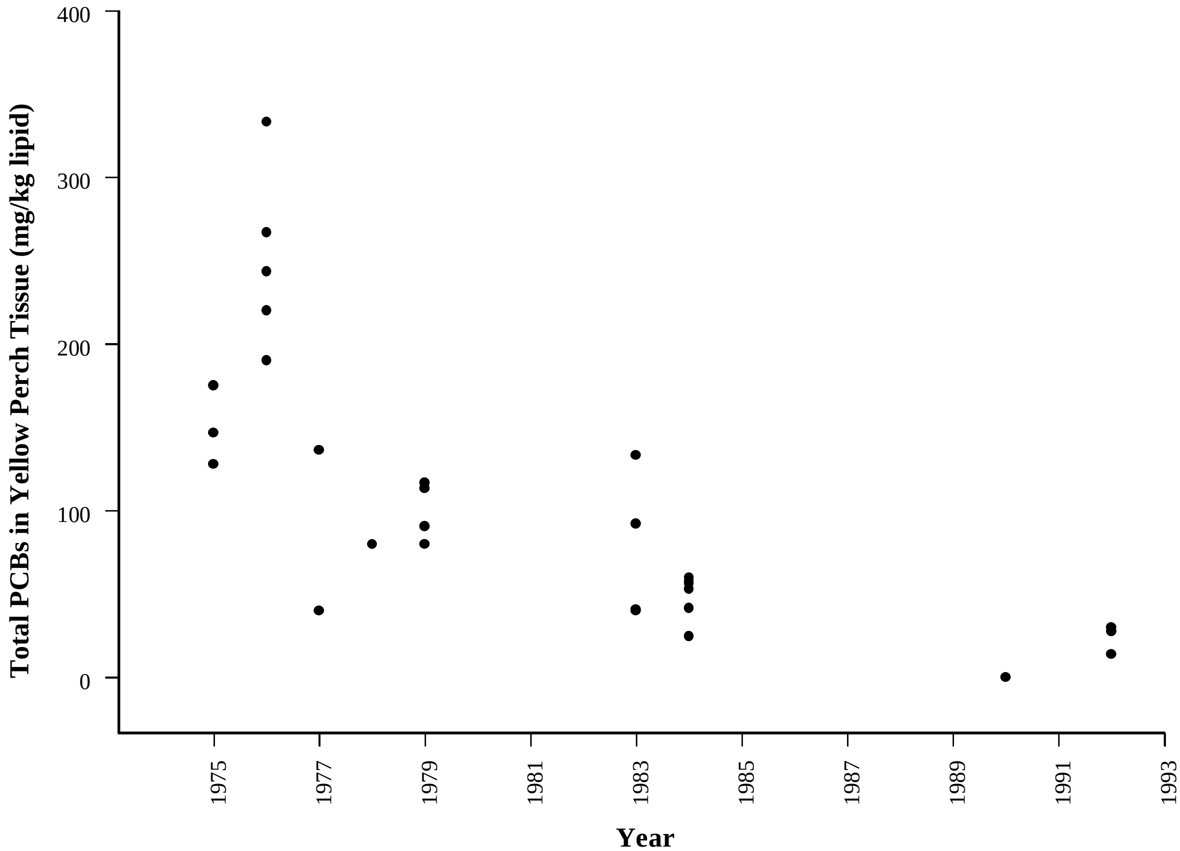


Figure 4-5. Total PCB concentrations measured in Green Bay yellow perch tissue. Each point represents a separate fish sample.
Data sources: Connolly et al., 1992; Wisconsin DNR, 1971-1995.

and the presence or absence of other stressors (e.g., other contaminants, environmental stressors), this comparison is not, in itself, a definitive determination of injury. However, it provides an indication as to whether PCB concentrations in Green Bay birds may be sufficient to cause adverse effects. A detailed evaluation of field studies examining actual adverse effects in Green Bay birds is presented in Chapter 5.

The comparison presented in this section is similar to the hazard quotient approach used in ecological risk assessment (e.g., U.S. EPA, 1998a). Hazard quotients are calculated as the ratio of exposure concentrations to toxicity threshold concentrations. Hazard quotients greater than one mean that contaminant exposure is at levels above toxic thresholds, thereby indicating risk. Although we do not actually calculate hazard quotients in this analysis, our approach is analogous to the hazard quotient approach.

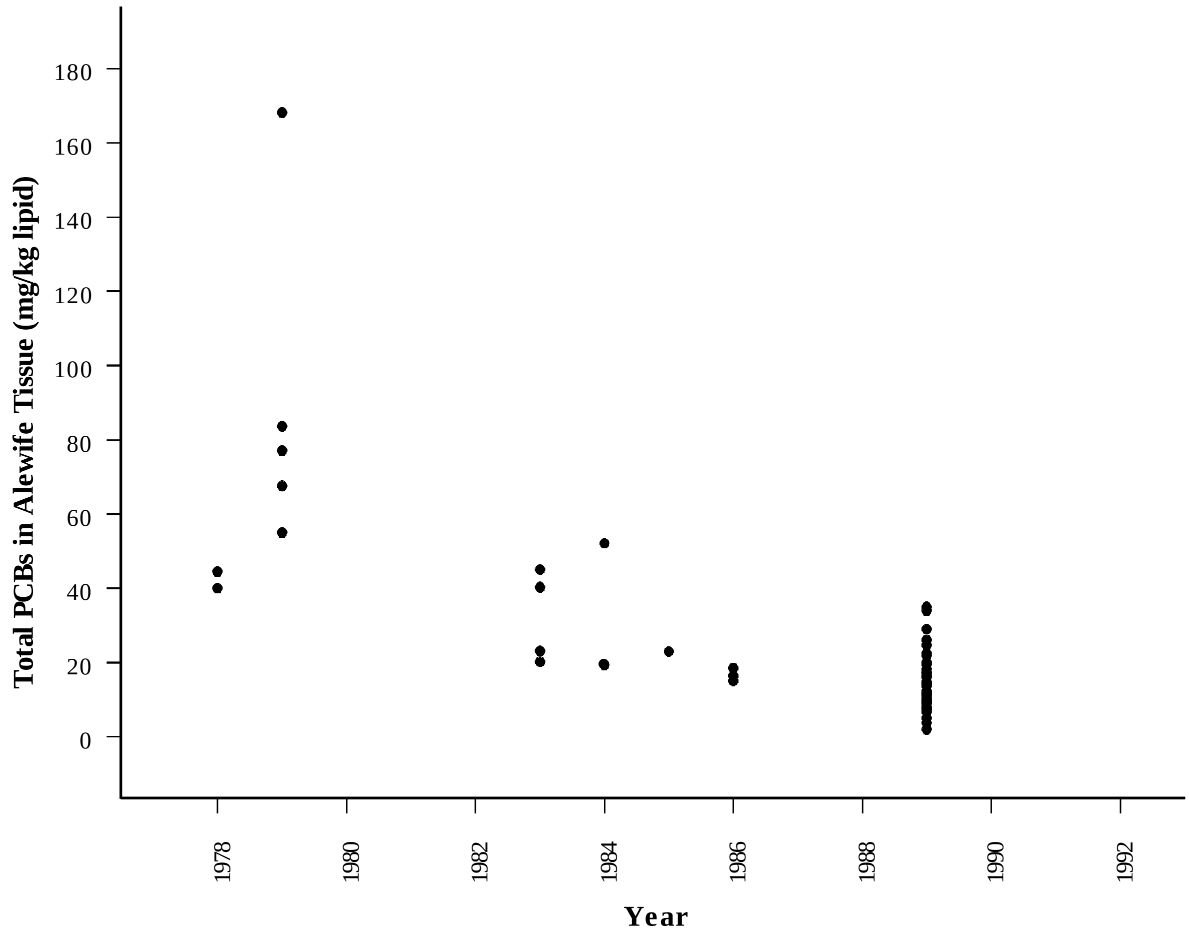


Figure 4-6. Total PCB concentrations measured in Green Bay alewife tissue. Each point represents a separate fish sample.
Data sources: Connolly et al., 1992; Wisconsin DNR, 1971-1995.

Green Bay bird egg PCB exposure data are available for both total PCBs and individual congeners. In the following analysis, total PCB concentrations measured in eggs are compared directly with the results of laboratory and field studies that quantified egg exposure as total PCBs. Measured PCB congener concentrations are converted to TCDD-eq and compared with toxicity studies that expressed exposure as TCDD or TCDD-eq concentration. Because individual PCB congeners can vary greatly in both their potency and relative concentrations in environmental samples, using TCDD-eq accounts for variations in congener potency and concentration that are not considered with total PCBs. WHO avian TEFs, which are TEFs developed by an international group of toxicology experts for use in avian risk assessments, were used to calculate TCDD-eq from congener concentrations (Van den Berg et al., 1998) (Table 3-4). Not all of the PCB congeners that have measurable TCDD-like effects (i.e., have nonzero TEFs) were measured in all of the Green Bay bird egg samples, leading to an underestimation of TCDD-eq. On the other

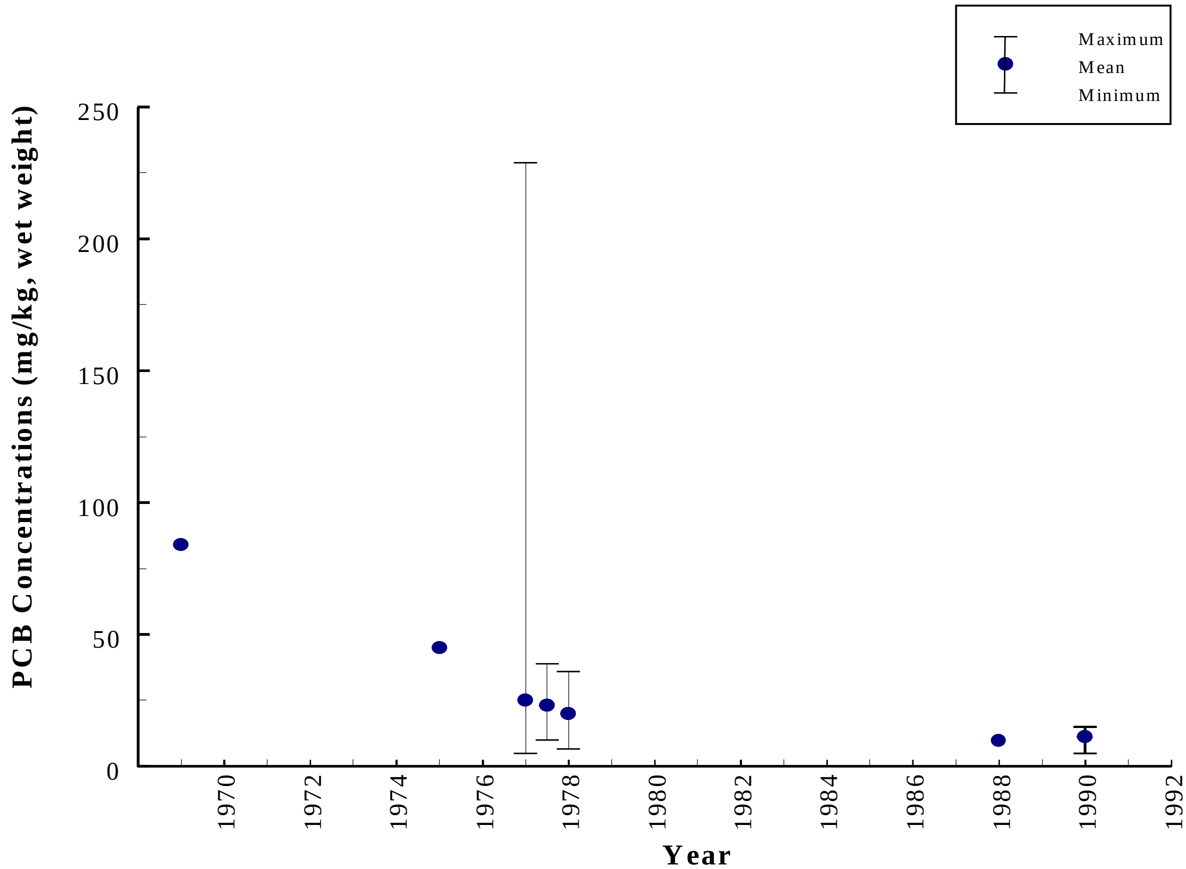


Figure 4-7. Mean PCB concentrations in red-breasted merganser eggs from Green Bay between 1968 and 1990.
Data sources: White and Cromartie, 1977; Haseltine et al., 1981; Heinz et al., 1983; Williams et al., 1995b.

hand, the calculation of TCDD-eq from congener concentrations assumes strict additivity of TCDD-like congener effects and does not take into account possible antagonism (Bosveld, 1995), although effects appear to be close enough to additive to justify the TEF approach in risk assessment (Van den Berg et al., 1998). Finally, only PCB concentrations measured in Green Bay bird eggs since 1983 are included, since the annual survey of Green Bay herring gull eggs and data on PCBs in Green Bay red-breasted mergansers (Figures 4-3, 4-4, and 4-7) show that PCB concentrations prior to the mid-1980s were still declining. Therefore this analysis underestimates pre-1983 effects.

Based on the values shown in Tables 3-6 and 3-7, egg total PCB concentrations of between approximately 5 and 10 mg/kg egg (wet weight) may result in adverse effects in sensitive wild bird species. This range is used as an overall estimate of the range at which toxic effects may begin to be seen in wild birds. Below 5 mg/kg wet weight total PCBs in eggs, effects appear to be unlikely. At and above this range, adverse effects are likely for at least sensitive wild bird species.

The adverse effects to birds documented as occurring at and above this range include reduced reproductive success, deformities, and behavioral abnormalities. It should be noted that concentrations of less than 5 mg/kg wet weight total PCBs in eggs have been shown to cause reduced hatching success in the domestic chicken. However, because the chicken is more sensitive to PCB toxicity than any wild bird species tested to date (Bosveld, 1995), it was not included in the derivation of the toxic threshold range so that the threshold is more relevant for bird species of concern in Green Bay.

From the information presented in Table 3-7, 200-10,000 pg TCDD-eq/kg egg (wet weight) is a representative toxic effects range. As with total PCBs, this range represents concentrations below which toxicity appears to be unlikely, and within and above which adverse effects to many species have been documented. Again, this range does not incorporate data for the chicken, which is much more sensitive than any wild bird species tested to date.

Data on total PCB exposure in assessment area bird eggs are shown in Figure 4-7 for red-breasted mergansers, Figure 4-8 for double-crested cormorants, Figure 4-9 for common terns, Figure 4-10 for Forster's terns, Figure 4-11 for Caspian terns, and Figure 4-12 for bald eagles. Data shown are for eggs collected from Green Bay, ranging from the inner bay (common and Forster's terns) to the outer bay (Caspian terns and bald eagles). In all the figures the mean total PCB concentrations reported are presented, and in some cases the minimum and maximum values were also available and are plotted.

Figures 4-7 through 4-12 show that for all six of these species, *average* total PCB concentrations measured in eggs after 1983 are within or above the 5-10 mg/kg range. These data indicate that the total PCB concentrations measured in eggs of red-breasted mergansers, double-crested cormorants, common terns, Forster's terns, Caspian terns, and bald eagles within the assessment area are within or, in some cases, exceed the range where adverse reproductive effects have been reported in sensitive species.

Figure 4-13 shows TCDD-eq concentrations calculated from PCB congener measurements made in assessment area bird eggs. PCB congener measurements (including coplanar congeners) are available for red-breasted mergansers, double-crested cormorants, common terns, and Forster's terns. PCB congener data were converted to TCDD-eq using both the WHO Avian TEFs (Van den Berg et al., 1998). For each species, all assessment area congener data since 1983 are combined. The sources of the congener data used in Figure 4-13 are listed in Table 4-3.

Figure 4-13 shows that the average TCDD-eq concentrations in eggs of all of the species are within or exceed the 200-10,000 pg TCDD-eq/kg egg (wet weight) toxic effects range for sensitive species derived from Table 3-7. These data are consistent with the total PCB data, and indicate that the mixture of PCB congeners in assessment area bird eggs is of sufficient potency and concentration to potentially cause adverse effects.

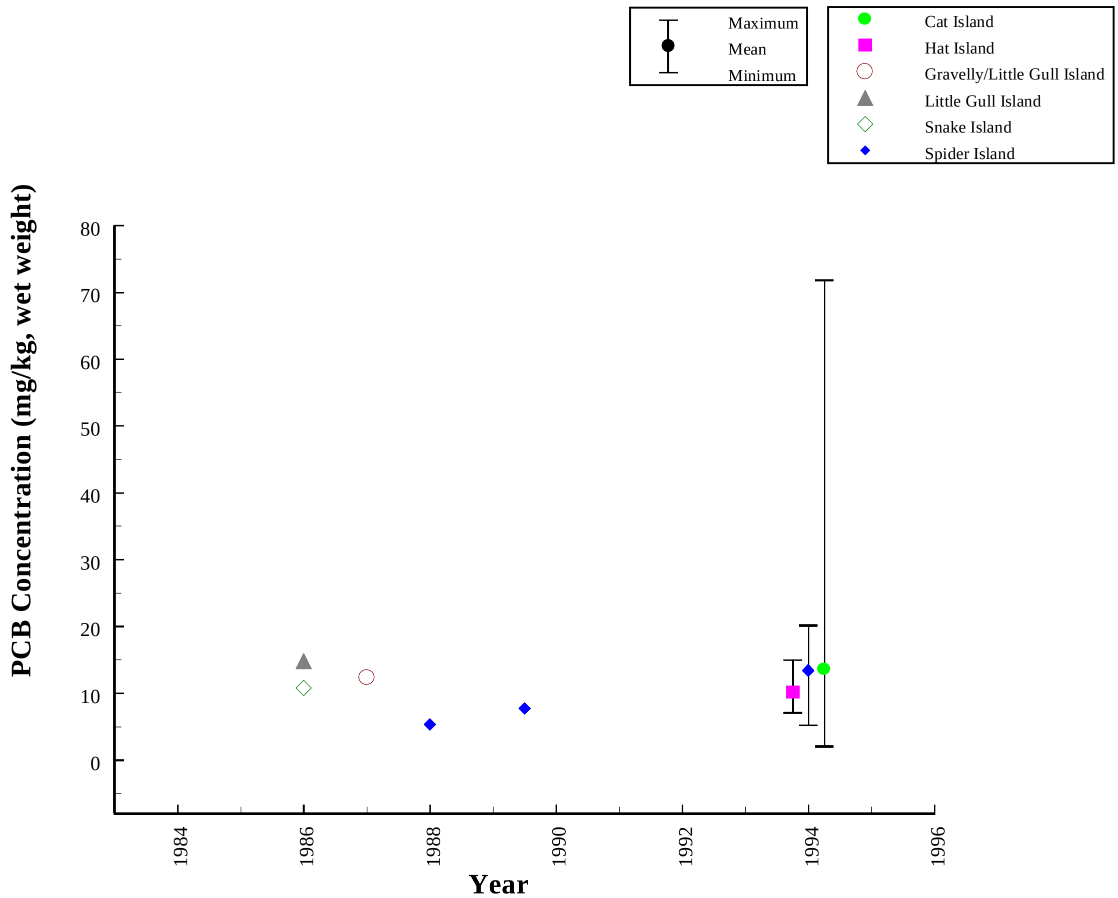


Figure 4-8. Total PCB concentrations measured in assessment area double-crested cormorant eggs, 1983-1996. See Table 4-1 for data sources.

4.5 CONCLUSIONS

The information presented and evaluated in this chapter supports the following conclusions:

- Numerous species of birds throughout the assessment area are exposed to PCBs. The primary route of exposure for most assessment area bird species is dietary.

- PCB concentrations measured in the tissues of assessment area bird species are statistically significantly greater than concentrations measured in reference areas. Every species tested has been found to have greater concentrations in the assessment area, including double-crested cormorant, black-crowned night heron, herring gull, Forster's tern, common tern, Caspian tern, mallard, bald eagle, tree swallow, and red-winged blackbird.

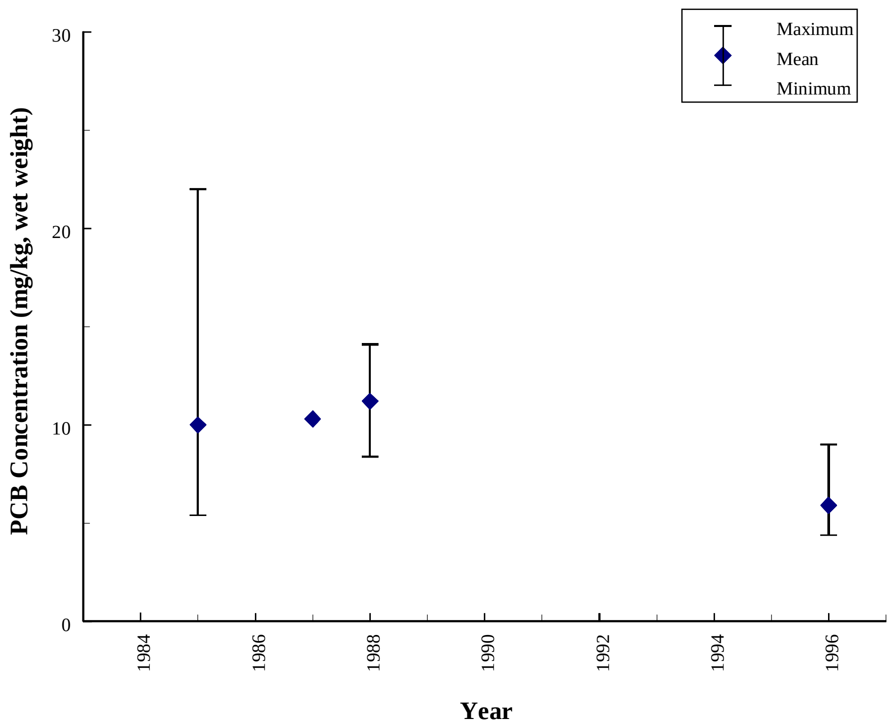


Figure 4-9. Total PCB concentrations measured in assessment area common tern eggs, 1983-1996. See Table 4-1 for data sources.

- PCB exposure of assessment area birds, as measured by PCB accumulation in bird tissue, was greatest in the early 1970s (the first dates for which data are available), declined through the 1970s and through the early 1980s, and has remained relatively stable since then.

- Total PCB concentrations measured in eggs of assessment area red-breasted mergansers, double-crested cormorants, common terns, Forster's terns, Caspian terns, and bald eagles from 1983 to 1996 are within or, in many cases, exceed the range where adverse reproductive effects have been reported in sensitive species.

- TCDD-eq concentrations calculated from PCB congener concentrations measured in assessment area bird eggs are within or exceed a TCDD-based toxicity threshold range. These data indicate that assessment area bird eggs contain a mixture of PCB congeners of sufficient potency and concentration to cause adverse effects.

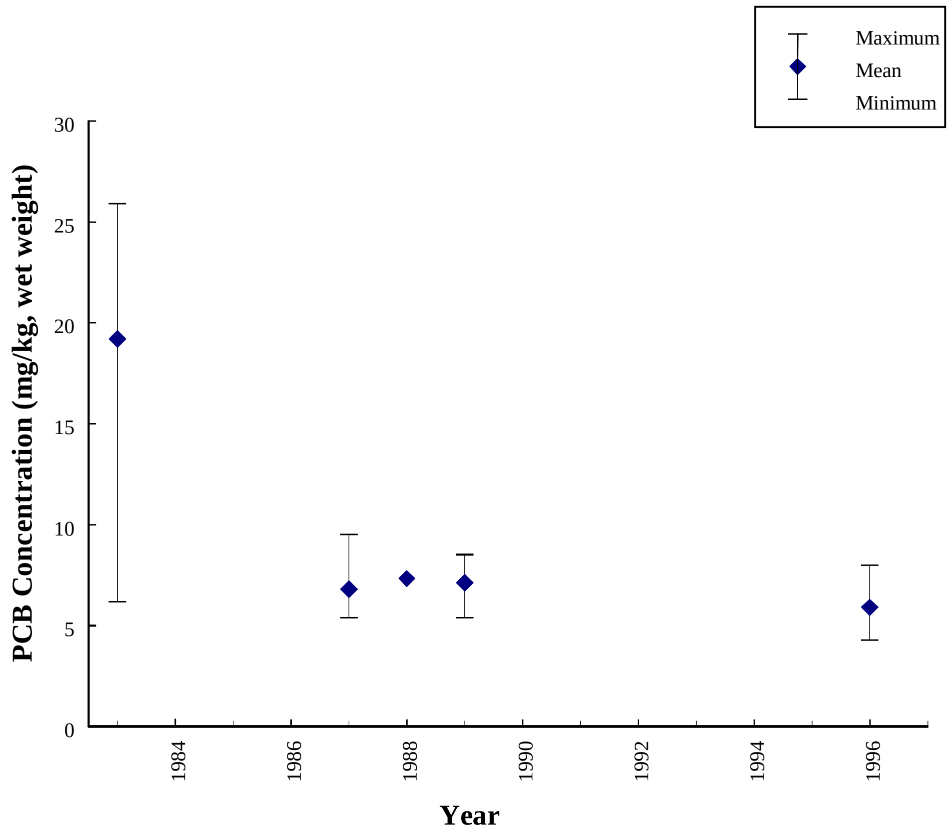


Figure 4-10. Total PCB concentrations measured in assessment area Forster's tern eggs, 1983-1996. See Table 4-1 for data sources.

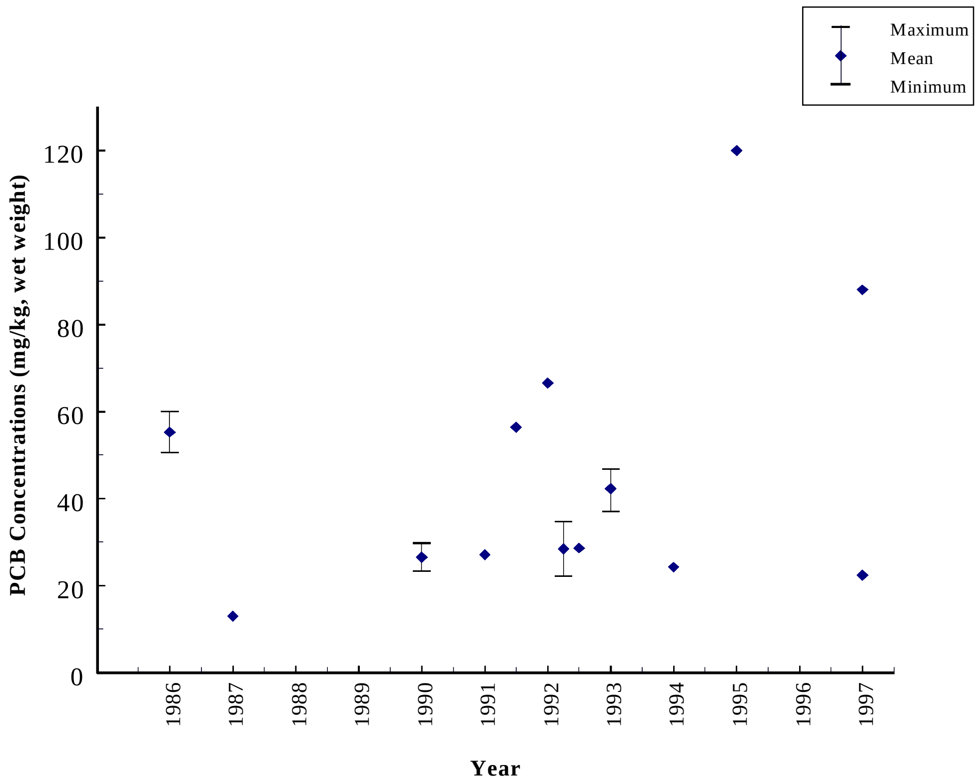


Figure 4-11. Total PCB concentrations measured in assessment area Caspian tern eggs, 1983-1996. See Table 4-1 for data sources.

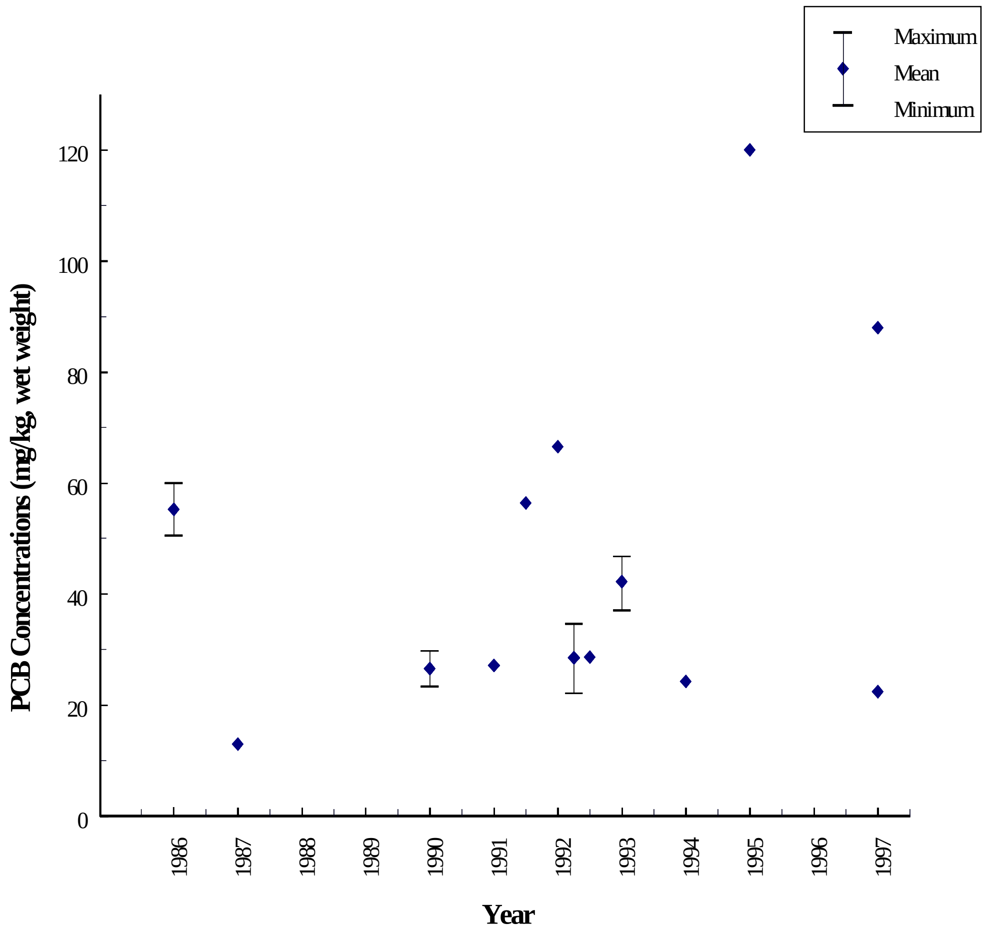


Figure 4-12. Total PCB concentrations measured in assessment area bald eagle eggs, 1986-1997. See Table 4-1 for data sources.

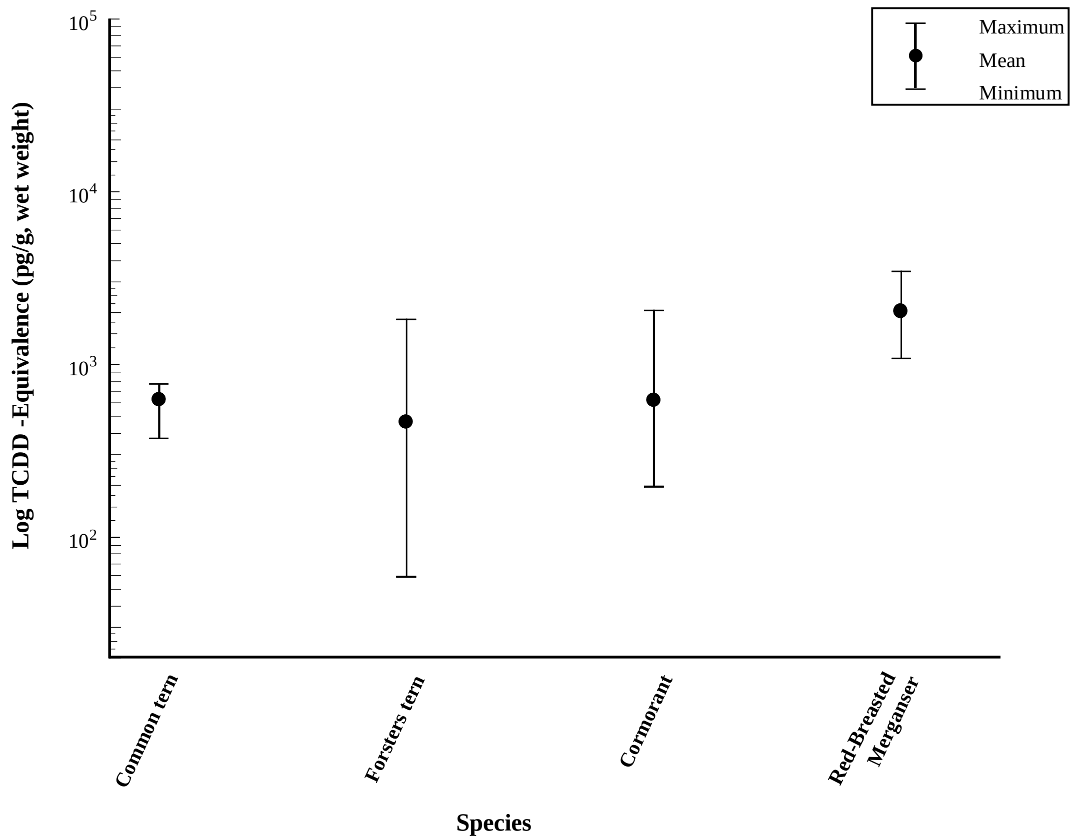


Figure 4-13. PCB TCDD-eq concentrations in assessment area bird eggs, 1983-1996.
TCDD-eq concentrations are calculated from measured PCB congener concentrations using the WHO Avian (U.S. EPA, 1998b) and Kennedy et al. (1996a) TEFs. See Table 4-3 for data sources.

Table 4-3
Sources of Assessment Area PCB Congener Data in Bird Eggs[a]

Species	Year of Collection	Number of Samples	Source
Red-breasted merganser	1990	12	Williams et al., 1995a
Double-crested cormorant	1988 1989 1994-1995	pool of 18 eggs 11 pools of 33 eggs 10	Yamashita et al., 1993 Williams et al., 1995a Custer, pers. comm., 1998
Common tern	1988 1996	2 6	Ankley et al., 1993 Appendix B of this report
Forster's tern	1982 1983 1988 1989 1996	2 6 5 5 6	Smith et al., 1990 Kubiak et al., 1989 Harris et al., 1993 Jones et al., 1993 Appendix B of this report

a. Only studies that included analysis of nonortho congeners (e.g., PCB 77, PCB 81, PCB 126, PCB 169) were used.

CHAPTER 5
INJURY EVALUATION

The previous chapter demonstrated that birds in the assessment area have been exposed to PCBs and that concentrations of PCBs measured in their tissues have exceeded concentrations that are reported to result in toxicological effects in sensitive species. In this chapter, we evaluate evidence from field studies that birds in the assessment area have suffered adverse effects as a result of PCB exposure. The species for which injuries are assessed are Forster's, common, and Caspian terns (Section 5.1); double-crested cormorant (Section 5.2); black-crowned night heron (Section 5.3); tree swallow (Section 5.4); red-breasted merganser (Section 5.5); and bald eagles (Section 5.6). These species are evaluated because of the field data available on adverse effects in the assessment area. However, it is emphasized that, with the exception of tree swallows, these birds are representative of a broader guild of birds for which fish are an important dietary component. This guild also includes other species that inhabit Green Bay, such as great blue herons, green-backed herons, white pelicans, ospreys, and gulls.

Assessment of injury to waterfowl according to the injury definitions related to exceedences of PCB tolerance levels or establishment of waterfowl consumption advisories is addressed in Chapter 6.

5.1 FORSTER'S, COMMON, AND CASPIAN TERNS

5.1.1 Status and Ecology in Green Bay

Forster's, common, and Caspian terns arrive at their nesting colonies in Green Bay in April and May and depart for their winter habitats in the southern United States and Central and South America in September and October (Ludwig, 1965; Burger and Gochfeld, 1991). Their nesting areas are usually on islands, where they are safe from land-based predators such as raccoons, foxes, and mink (Burger and Gochfeld, 1991). In the assessment area, the primary nesting areas for Forster's terns currently are the Confined Disposal Facility (CDF) near the mouth of the Fox River, Long Tail Point (approximately 3 miles from the mouth of the Fox River along the western shore of the bay), and Oconto Marsh (at the mouth of the Oconto River on the western shore of the bay). Common terns nest on the CDF, and Caspian tern colonies are located on Gravelly and Gull islands (between northern Green Bay and Lake Michigan) (see map, Figure 2-4).

Population Status in Green Bay

In 1935, Forster's terns were rare breeders in Wisconsin (Mossman, 1988). The first annual statewide census in 1978 found 136 pairs nesting in Green Bay. This total increased to 435 pairs by 1987. The lack of rigorous census data before the late 1970s makes it difficult to evaluate long-term population changes of Forster's terns in Green Bay. The only conclusion that is supported by the data is that the population increased between the late 1970s and the late 1980s. The State of Wisconsin listed the Forster's tern as endangered in 1979 (Mossman, 1988).

The Green Bay breeding population of common terns also increased over the same time period (late 1970s until the late 1980s). In 1979 there were 60 pairs breeding in Green Bay, and in 1986 there were 600 pairs (Matteson, 1988). Most of this increase took place between the 1984 and 1985 censuses (66 to 427 pairs). This rapid rate of increase could not have been supported by local productivity alone and must have been at least partly caused by immigration from outside of the area. Data on Green Bay common tern populations before this period of increase are sparse, but there is some evidence for a breeding population of several hundred pairs in the 1940s (Matteson, 1988). The State of Wisconsin listed the common tern as endangered in 1979 (Matteson, 1988).

Caspian tern breeding numbers in Green Bay and Lake Michigan have also increased over the last 20 years. In 1977 and 1978 there were 602 nests on Gravelly and Gull islands; by 1991, there were over 1,000 nests (Ewins et al., 1994). This increase is part of a general increase in the Great Lakes Caspian tern metapopulation, which has grown by at least 90% since the late 1970s (Ewins et al., 1994). The State of Wisconsin listed the Caspian tern as endangered in 1989 (Matteson, 1993).

5.1.2 Pathway and Exposure Analysis

Data presented in Chapter 4 of this report show that Forster's, common, and Caspian terns in the assessment area have been exposed to elevated concentrations of PCBs relative to birds collected from reference areas. The purpose of the supplemental pathway analysis presented in this section is to identify the environmental components through which this exposure has occurred. Specifically, we address the following questions: What are the principal prey items of the three species? Where do the species feed? Are their prey items contaminated with PCBs?

Diets and Foraging Areas

Forster's, common, and Caspian terns are mainly piscivorous (Salt and Willard, 1971; Cramp, 1985; Burger and Gochfeld, 1991; Fraser, 1994). Although few data quantitatively describe their diets in Green Bay, several studies carried out elsewhere in the Great Lakes provide evidence of their probable diets in the assessment area.

Fraser (1994) found that during courtship feeding and chick provisioning, Forster's terns at Lake Osakis, Minnesota, mainly ate yellow perch, shiners, and sunfish. Most of these fish were 7 cm or less in length. Trick (1982) reported that Forster's terns in Green Bay generally forage in littoral areas (i.e., areas of shallow water) adjacent to marshes or coastlines. This was also true at Lake Osakis, where Fraser (1994) found that Forster's terns generally foraged over shallow water within 5-20 m of the shore. At Lake Osakis, Forster's terns generally foraged within 5 km of the breeding colony (Fraser, 1994). While foraging distance is likely to be affected by site-specific factors such as the size of the water body, shoreline configuration, and the spatial distributions of feeding and nesting sites, foraging close to the colony is also likely to apply to Foster's terns in Green Bay.

In Lake Ontario, 90% of the diet of breeding common terns was alewives (*Alosa pseudoharengus*) and smelt (*Osmerus mordax*), whereas in Lake Erie, smelt, emerald shiners (*Notropis atherinoides*), and trout-perch (*Percopsis omiscomaycus*) were the main items (Courtney and Blokpoel, 1980). During 1990 and 1991, the diet of breeding common terns on Lake Erie was dominated by smelt and emerald shiner (Burness et al., 1994). Common terns also typically forage within a few kilometers of the breeding colony (Cramp, 1985; Burness et al., 1994). Birds nesting on the Green Bay CDF would probably obtain most of their food locally and within a few kilometers of the mouth of the Fox River.

Data have been reported on the diets of Caspian terns in Green Bay. Ewins et al. (1994) collected 31 regurgitated pellets from the vicinities of nests on Gravelly Island in 1991. All pellets contained the remains of alewives, 10% contained smelt, and 3% contained centrarchid remains. Using pellets to investigate avian diet can be difficult (e.g., Ewins et al., 1994); however, it seems likely that the diets of adult Caspian terns (the pellets were collected before chick hatching) nesting at Gravelly Island in 1991 comprised, in large part, alewives. Alewives and smelt have been shown in other studies to be important components of Caspian tern diet in Lake Michigan waters (Ludwig, 1965). No data have been reported on the foraging ranges of Caspian terns breeding in the assessment area, or elsewhere in the Great Lakes. However, given their larger body size, their foraging ranges may be larger than those of common or Forster's terns.

PCBs in Prey Items

Whole-body PCB concentrations in alewives, gizzard shad, and smelt were measured in 1989 as part of the development of the Green Bay Mass Balance Model (Connolly et al., 1992). Fish were collected from six zones (Figure 5-1) within the assessment area: the Fox River, the eastern and western halves of the inner bay from the Fox River mouth to Little Tail Point (approximately 10 miles north of the Fox River mouth), the eastern and western halves of the inner bay from Little Tail Point to Chambers Island, and the outer bay (beyond Chambers Island).

Figures 5-2, 5-3, and 5-4 show mean total PCB concentrations measured in the three forage fish species. In general, mean concentrations were higher for gizzard shad and alewives, which are

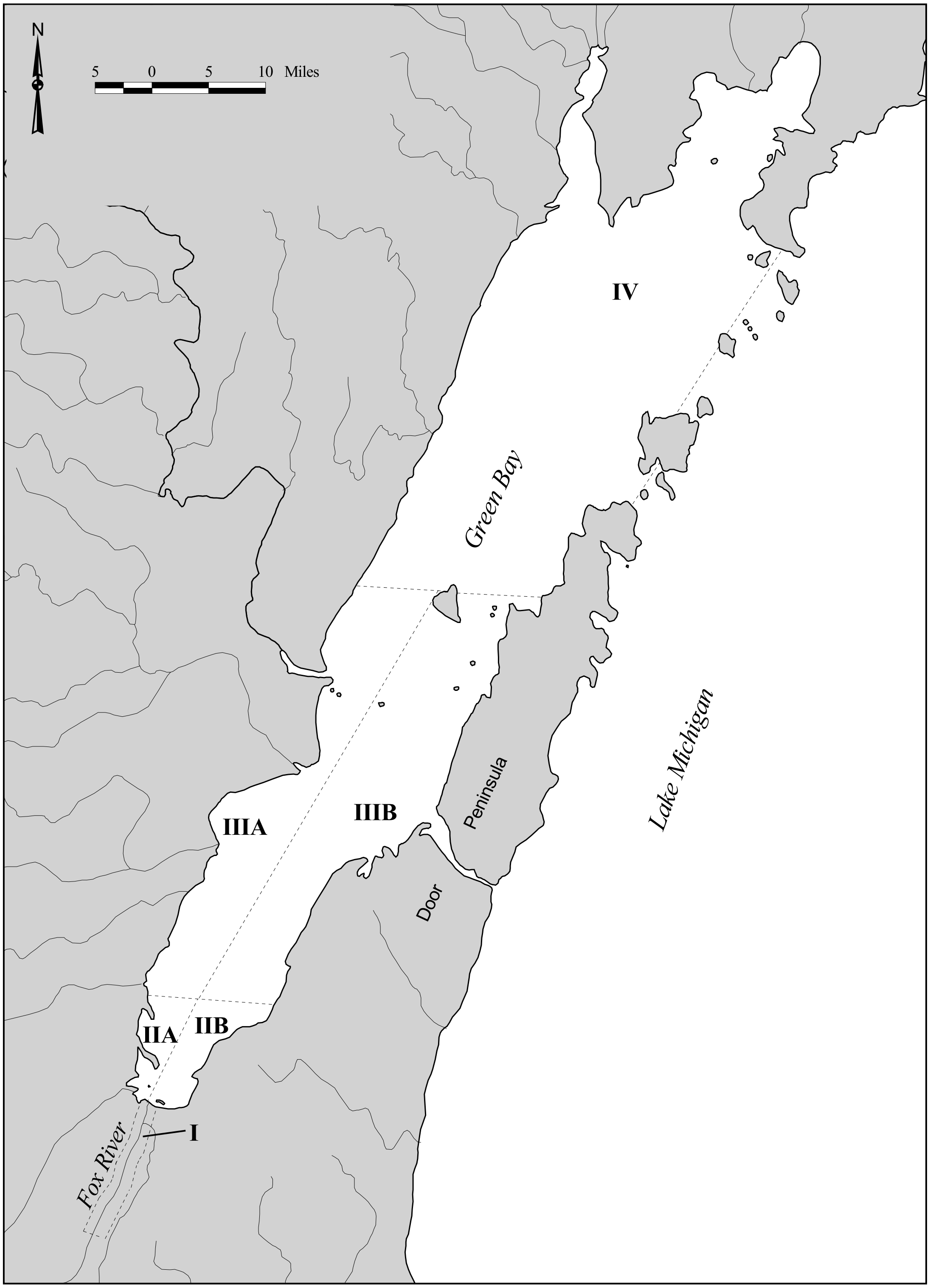


Figure 5-1. Forage fish sampling zones in 1989.

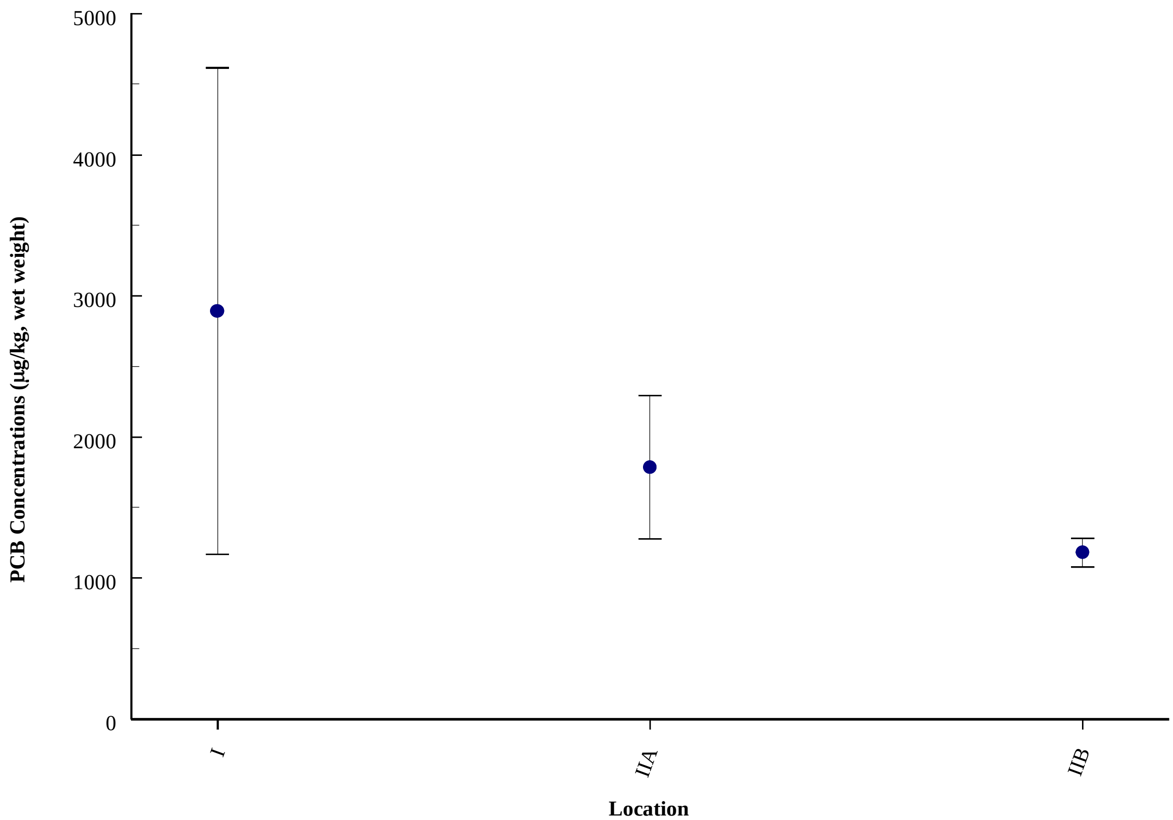


Figure 5-2. Total PCB concentrations in Green Bay gizzard shad, 1989. Bars equal means plus or minus 1 standard deviation. Data from Green Bay Mass Balance Model (Connolly et al., 1992).

relatively lipid-rich (Rottiers and Tucker, 1982; Oliver and Niimi, 1988), than for smelt. A general spatial pattern of decreasing concentrations with increasing distance from the Fox River is also evident (i.e., from zones I to IV). Concentrations in alewives also appear to be higher along the eastern shore of the bay (zones IIB and IIIB) than along the western shore (zones IIA and IIIA). This spatial pattern of PCBs in forage fish is consistent with that observed in sediment and is indicative of the Fox River being the primary source of PCBs to the bay (Manchester-Neesvig et al., 1996). A similar PCB concentration gradient has been observed in young-of-the-year littoral fishes collected from wetlands and beaches along Green Bay (Bruzner and DeVita, 1998). The forage fish data indicate that piscivorous bird exposure to PCBs in prey items tends to decrease with distance from the Fox River, yet is elevated throughout the bay. These data confirm that piscivorous birds in the assessment area are exposed to PCBs in their diet.

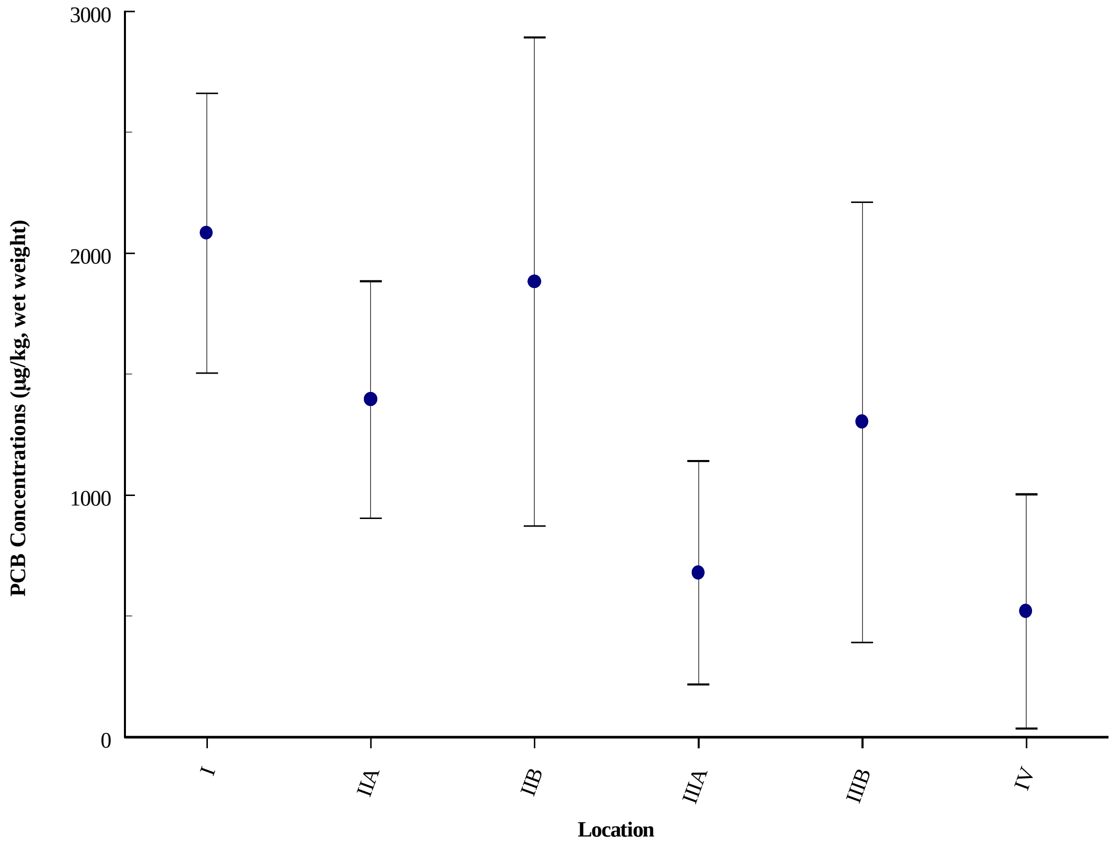


Figure 5-3. Total PCB concentrations in Green Bay alewives, 1989. Bars equal means plus or minus 1 standard deviation. Data from Green Bay Mass Balance Model (Connolly et al., 1992).

Evidence of PCB Uptake

Harris et al. (1993) monitored PCB concentrations in Forster's tern chicks on the Kidney Island CDF from hatching to fledgling. They found that the PCB concentration during this rapid growth period remained relatively constant, showing that the chicks were ingesting PCBs at a rate sufficient to keep pace with the increase in body weight. These data demonstrate that Forster's tern chicks were being fed PCB-contaminated food.

5.1.3 Field Studies of Injuries to Green Bay Terns

Forster's Tern

Field study descriptions. Two sets of studies of the potential effects of contaminants on the reproduction of Forster's terns in Green Bay have been performed.

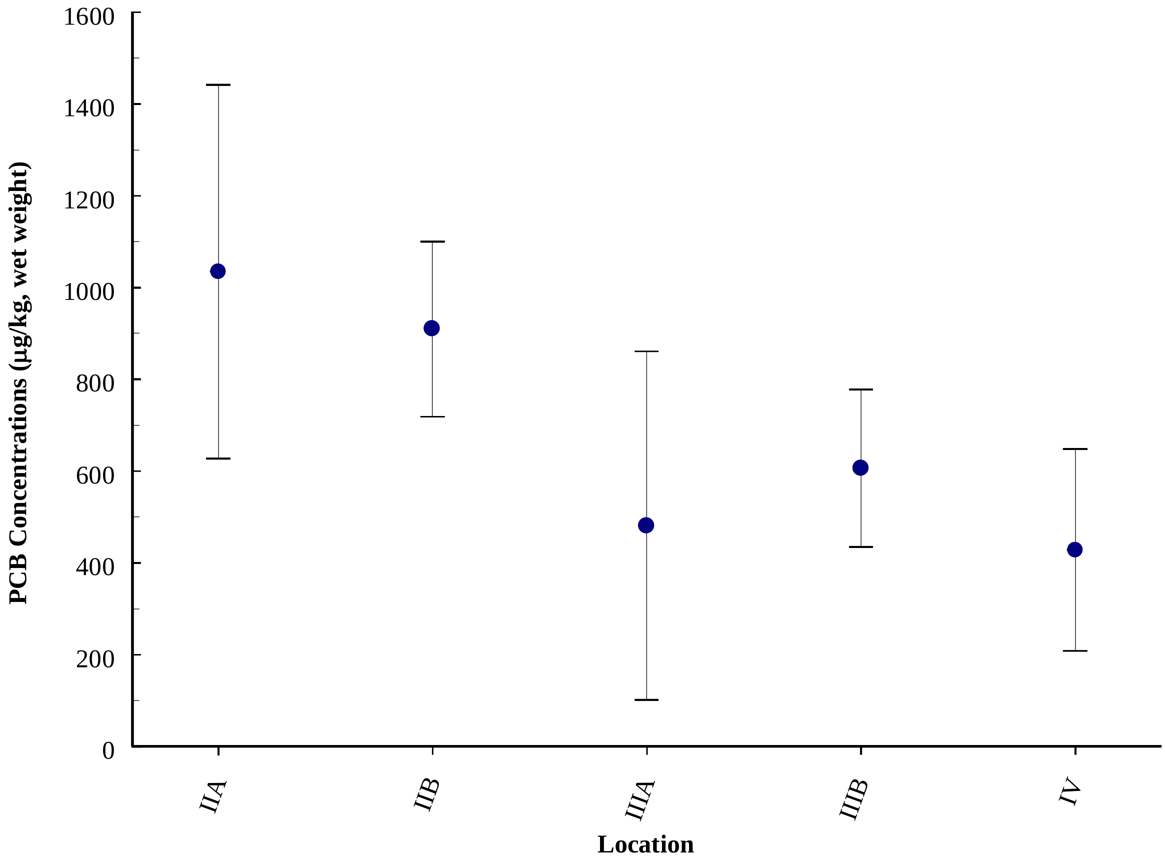


Figure 5-4. Total PCB concentrations in Green Bay smelt, 1989. Bars equal means plus or minus 1 standard deviation. Data from Green Bay Mass Balance Model (Connolly et al., 1992).

Hoffman et al. (1987) and Kubiak et al. (1989). In these companion studies, contaminant concentrations, reproductive performance, deformity rates, and biochemical responses were compared between Forster's terns nesting in Oconto Marsh, Green Bay (at the mouth of the Oconto River) and Forster's terns nesting at Lake Poygan, Wisconsin, an inland lake located in the Fox River drainage upstream of paper company PCB sources. Reproductive performance (but not contaminants, deformity rates, or biochemistry) was also monitored in Forster's terns nesting on Long Tail Point in inner Green Bay. The field work was performed in 1983.

Six tern eggs were analyzed for contaminants from both the Oconto Marsh (Green Bay) and Lake Poygan (reference) colonies. Eggs from the Oconto Marsh colony had a mean PCB concentration of 19.2 mg/kg wet weight (median of 22.2 mg/kg wet weight), and the Lake Poygan eggs had a mean of 4.6 mg/kg wet weight (median 4.5 mg/kg wet weight). The mean PCB concentrations between eggs from the two colonies were reported as being significantly different ($p < 0.05$). No egg contaminant data were collected for the Long Tail Point (Green Bay) colony.

In a companion paper to Kubiak et al. (1989), Tillitt et al. (1993) reported results of H4IIE bioassays on Green Bay and Lake Poygan Forster's tern eggs. The Green Bay eggs averaged 214.5 pg/g TCDD-EQ, compared to 23.4 pg/g TCDD-EQ at Lake Poygan. This difference was reported as being statistically significant.

The reproductive successes of the colonies are summarized in Table 5-1. Egg hatching rates were significantly lower in the Green Bay colonies than in the Lake Poygan colony. Of the eggs monitored, 40% and 55% hatched successfully at the Oconto Marsh and Long Tail Point colonies, respectively, whereas 88% of the eggs laid at the Lake Poygan site hatched. The percentage of nesting pairs that produced at least one fledgling was also lower in the Green Bay colonies, as was the average number of fledglings produced per nest. The reproductive success of terns at the Long Tail Point colony was intermediate between that of the Oconto Marsh and Lake Poygan colonies.

Table 5-1
Reproductive Success of Green Bay and Reference Colonies of Forster's Terns

Colony	Mean PCB Concentration (n = 6, mg/kg, wet weight)	Percent of Eggs that Hatched	Percent of Nesting Pairs that Were Successful[a]	Number of Fledglings/Nest
Oconto Marsh (Green Bay)	19.2	40% (14/35)	0% (0/12)	0
Long Tail Point (Green Bay)	—[b]	55% (18/33)	42% (5/12)	0.58
Lake Poygan (reference)	4.6	88% (30/34)	91% (10/11)	1.55

a. Successful means producing at least one fledgling. Only pairs from which no eggs were removed or exchanged are included.
b. No chemistry measurements were made at this colony.

Source: Kubiak et al., 1989.

The causes of the reduced hatching success in the Green Bay colonies were investigated using a combination of laboratory incubation of eggs collected at Oconto Marsh and Lake Poygan and field experimentation in which eggs were transferred between the colonies. In laboratory incubators, only 37% of the 19 Oconto Marsh eggs hatched, compared with 75% of the 20 Lake Poygan eggs. This statistically significant difference indicates that factors that were intrinsic to the eggs themselves affected hatchability under controlled conditions. Our own statistical analysis of the Kubiak et al. (1989) data showed that the hatching success rates in the incubators did not differ significantly from those in the natal colonies in the field ($\chi^2 = 1.45$ and 1.64, respectively; $p > 0.25$, 1 df).

In the egg transfer experiment, Kubiak et al. (1989) found that eggs removed from the Oconto Marsh colony and incubated by Lake Poygan adults had a significantly higher hatching success rate (94%) than Oconto Marsh eggs incubated in their natal colonies (55%), or in the laboratory (37%). This indicates that factors extrinsic to the eggs themselves were also important in reducing hatching success in Green Bay. This conclusion is supported by the fact that eggs transferred from the Lake Poygan colony to Oconto Marsh had a significantly lower hatching success (11%) than Lake Poygan eggs incubated by Lake Poygan adults (88%) or in the incubator (75%). These results are summarized in Table 5-2. The most plausible explanation for this extrinsic effect is that the reproductive behavior of the Oconto Marsh adults was less likely to result in a successful reproductive outcome than the reproductive behavior of Lake Poygan adults.

Table 5-2
Percent Hatching Success Results of Forster's Tern Egg Transfer and Laboratory Hatching Study

	Colony Where Eggs Incubated		
Source Colony	**Lake Poygan**	**Oconto Marsh**	**Laboratory**
Lake Poygan	88%	11%	75%
Oconto Marsh (Green Bay)	94%	55%	37%

Source: Kubiak et al., 1989.

Additional evidence that the reproductive performance was poorer in the Green Bay colonies was provided by data on incubation periods and nest abandonment rates. Oconto Marsh eggs took 4.6 days longer than Lake Poygan eggs to hatch in the laboratory. In the field, Oconto Marsh eggs incubated by their own parents took significantly longer to hatch (by 8.2 days) than Lake Poygan eggs incubated by their own parents. There was no difference in the time required for incubation between Lake Poygan eggs hatched in the natal colony and those hatched in the laboratory incubator. From these data, Kubiak et al. (1989) concluded that "about half of the longer incubation period for dirty eggs in the field . . . must have been due to intrinsic factors and about half to extrinsic factors."

In a companion study to Kubiak et al. (1989), Hoffman et al. (1987) reported incidences of deformities and liver microsomal aryl hydrocarbon hydroxylase (AHH) activity in embryos from the Forster's tern eggs from the Oconto Marsh and Lake Poygan colonies. In addition to lower body weights of hatchlings (also found by Kubiak et al., 1989), Hoffman et al. (1987) found that the Oconto Marsh eggs had significantly higher AHH activity (by a factor of 3), significantly greater liver-to-body weight ratios, and significantly shorter femurs. Three instances of structural deformities were also found in the Oconto Marsh embryos. These were one embryo with a crossed bill, one with a poorly ossified foot and a short lower mandible, and one with an incompletely ossified ilium. No deformities were found in Lake Poygan embryos. In total, 16.7% of Oconto Marsh hatchlings and embryos had structural deformities, compared with 0% of the Lake Poygan neonates. This difference was reported as being statistically significant. Also, 27.7% of Oconto Marsh hatchlings and embryos had edema, compared with 13.3% of Lake Poygan neonates, although this difference was not statistically significant.

The total PCB concentrations reported by Kubiak et al. (1989) in Oconto Marsh Forster's tern eggs exceed concentrations shown to cause toxicity (Chapter 3) and are significantly higher than concentrations measured in Lake Poygan eggs. Kubiak et al. (1989) found that in addition to PCBs, several other contaminants were also higher in Oconto Marsh tern eggs than in Lake Poygan eggs. These contaminants were oxychlordane + heptachlorepoxide (median of 0.20 mg/kg wet weight in Oconto Marsh eggs vs. 0.04 mg/kg wet weight in Lake Poygan eggs); *p,p'* DDE (median of 1.8 mg/kg wet weight vs. 0.45 mg/kg wet weight); cis-nonachlor + *p,p*' DDD (median of 0.12 mg/kg wet weight vs. 0.01 mg/kg wet weight); hexachlorobenzene (median of 0.10 mg/kg wet weight vs. 0.02 mg/kg wet weight); heptachlor (median of 0.09 mg/kg wet weight vs. 0.02 mg/kg wet weight); toxaphene (median of 1.10 mg/kg wet weight vs. 0.37 mg/kg wet weight); dioxins (median of 101.5 pg/g vs. 25.0 pg/g); and furans (median of 18.5 pg/g vs. 9.0 pg/g). Based on the TEF approach, Kubiak et al. concluded that dioxins and furans contributed less than 10% of the total TCDD-eq in the eggs, with nonortho and mono-ortho PCB congeners contributing the rest. Similarly, Kubiak et al. concluded that the measured concentrations of toxaphene and hexachlorobenzene in the tern eggs were below toxic thresholds. Therefore, Kubiak et al. (1989) concluded that PCBs were the primary cause of the toxic effects observed in Green Bay Forster's terns.

Harris et al. (1993) and Ankley et al. (1993). These companion studies monitored the reproductive success and measured egg contaminant concentrations for a Forster's tern colony on the Kidney Island CDF located at the mouth of the Fox River (Figure 2-4). No reference colonies were evaluated. The field work was conducted in 1988.

The mean total PCB concentration measured in eggs from the CDF was 7.3 mg/kg wet weight (n = 5, median of 7.4) (Harris et al., 1993). The Kidney Island CDF Forster's terns had an egg hatching rate of 81% (65 of 80), similar to that found by Kubiak et al. (1989) at the Lake Poygan reference colony (88%). However, only 65% of the pairs monitored at the CDF produced at least one fledgling, whereas 91% of the pairs at the Lake Poygan colony monitored by Kubiak et al. (1989) produced at least one fledgling. Similarly, the average number of fledglings per nest was 1.0 for Forster's terns at the CDF and 1.5 for those at Lake Poygan in the Kubiak et al. study.

Harris et al. (1993) found that many of the CDF Forster's tern chick deaths occurred at a comparatively late stage of development (>20 days after hatching). These deaths were, in many cases, preceded by weight loss. Harris et al. (1993) noted that the pattern of weight loss was characteristic of the "wasting syndrome" caused by organochlorine compounds. Furthermore, Harris et al. noted that as many chicks from nests with one and two young died as did those from nests with three young, and suggested that this implied that starvation was not the cause of the deaths. However, chick mortality due to food shortages and starvation, even late in development, is not uncommon in tern colonies (Langham, 1972; Burger and Gochfeld, 1991). Absent a breakdown of the hatch order of the young that died, it is not possible to exclude starvation due to local food shortage.

Ankley et al. (1993) showed that nesting Forster's terns on the CDF accumulated PCB residues during growth, indicating that they were obtained from local sources.

Conclusions from Forster's tern field studies. The types of adverse effects observed in the field and their relationship to measured egg PCB concentrations show that Green Bay Forster's terns have been adversely affected by exposure to PCBs. Effects and PCB exposure were most severe at the Oconto Marsh colony, where hatching success and number of fledglings per nest were lower than those of a reference colony (Kubiak et al., 1989). Specific effects included embryonic deformations, skeletal deformities, and edema, all of which can be caused by PCBs, as discussed in Chapter 3. Reproductive success (percentage of eggs hatching and number of fledglings per nest) was also lower at the Long Tail Point colony (Kubiak et al., 1989). At the Kidney Island CDF colony, where egg PCB concentrations were lower than at Oconto Marsh, egg hatching was not reduced (Harris et al., 1993). The number of fledglings per nest also was reduced, although the cause of the reduction is not clear.

The controlled egg switching experiments by Kubiak et al. (1989) show that extrinsic factors, e.g., decreased parental attentiveness, contributed to the lowered reproductive success. The adverse behavioral effects of PCBs on nesting adults have been documented in several studies. In a laboratory study using ring doves, Peakall and Peakall (1973) found that PCBs caused decreased parental attentiveness during incubation. Fox et al. (1978) showed that Lake Ontario herring gulls, which had higher PCBs and a lower rate of reproductive success than those from reference areas, also showed increased time away from nests and decreased nest defense during egg incubation. These documented adverse effects of PCBs on adult behavior during nesting are consistent with the findings of Kubiak et al. (1989) that extrinsic factors contributed to the reduced reproductive success of Green Bay Forster's terns.

Contaminants other than PCBs measured in the eggs were not significant contributors to the observed toxicity. Kubiak et al. (1989) determined that dioxins and furans, which can cause effects similar to those observed, accounted for less than 10% of the TCDD-eq in the eggs compared with PCBs. Other contaminants present in the eggs (e.g., DDE) are not known to cause the behavioral abnormalities or deformities that were observed, or were not present at concentrations sufficient to cause the observed effects (Kubiak et al., 1989).

Further evidence that DDE was not responsible for the adverse effects observed in Green Bay Forster's terns is provided by a study performed by King et al. (1991) in Texas. In this study the DDE concentrations in Forster's tern eggs from the contaminated and reference colonies were similar to the levels reported by Kubiak et al. (1989). PCB concentrations in the Texas Forster's tern eggs were low relative to the Green Bay eggs (1.2-2.3 mg/kg wet weight). Neither PCBs nor DDE were correlated with any measure of breeding success.

The field studies show that within the assessment area PCB exposure and effects were most severe at the Oconto Marsh colony and lowest at the Kidney Island CDF colony, which is consistent with the PCBs causing the observed effects. The reason for the lower PCB exposure and severity of effects observed in Forster's terns at the Kidney Island CDF compared with those at the Oconto Marsh is most likely a combination of both spatial and temporal variability. Based on the reproductive success endpoints, Forster's terns nesting at Long Tail Point, which is approximately 3 miles north of the Kidney Island CDF, may also have had lower contaminant exposure than did those at Oconto Marsh (Kubiak et al., 1989) (contaminants were not measured in Long Tail Point eggs). Data on PCBs in herring gull eggs from the Big Sister Island colony in Green Bay support the conclusion that temporal variability could also contribute to the observed variation in PCB concentrations of the Kidney Island CDF and Oconto Marsh eggs. Although PCB concentrations in herring gull eggs show no long-term trend from 1983 through 1996, concentrations vary from year to year by over a factor of two. Sample sizes of Forster's tern eggs for PCB analysis were smaller (six for Oconto Marsh and five for CDF) than those for herring gull eggs (10), which would increase between-year variability in the tern data. The herring gull egg PCB data also support the conclusion that the lower PCB concentration in Forster's tern eggs in 1988 at the CDF may not be indicative of a trend from 1983 to 1988 of declining PCB exposure for fish-eating birds in Green Bay.

Common Tern

Field study description. One study has been performed that is relevant to evaluating the potential effects of PCBs on the reproductive biology of common terns in Green Bay.

Hoffman et al. (1993). In this study, 35 newly laid, unincubated eggs of common terns were collected in 1985 from a colony situated on the Kidney Island CDF in Green Bay. Eggs were also collected from two reference colonies in nonindustrialized areas of northern Lake Michigan, Cut River, and Pointe aux Chenes. All eggs were artificially incubated in the laboratory, and hatching success, neonate morphology, biomarker activity, and contaminant concentrations were compared between colonies. No monitoring of reproductive success in the field was conducted as was done in the Forster's tern studies.

The Green Bay eggs had higher PCB concentrations (geometric mean of 10.0 mg/kg, wet weight; n = 10) than did eggs from the Cut Island (geometric mean PCB concentration of 4.7 mg/kg, wet weight) or Pointe aux Chenes (geometric mean PCB concentration of 4.0 mg/kg, wet weight) colonies. Mean liver AHH activity, a measure of exposure to TCDD-like contaminants, was also significantly higher in Green Bay eggs (mean of 23 pmol/min/mg protein, n = 22) than in either reference colony (10 and 9 pmol/min/mg protein and n = 22 and n = 12 at Cut River and Point aux Chenes, respectively).

Table 5-3 summarizes the parameters measured in the study that differed in the Green Bay and the reference area eggs, as well as the mean PCB concentrations. Hatching success of eggs from the Green Bay CDF (71%) was significantly lower than that of eggs from the Cut River colony (85%), but not statistically different from that of eggs from the Point aux Chenes colony (73%). Similarly, the femur length to body weight ratio in 1-day-old chicks was lower for Green Bay chicks than for Cut River chicks, but was not different from Point aux Chenes chicks. Four of the 35 (11%) Green Bay embryos or chicks were deformed, whereas no deformities were observed in any of the 55 reference embryos or chicks. Other morphological parameters measured, including hatching weight, liver weight, liver weight to body weight ratio, crown-rump length, and femur length, were not different between eggs from the different colonies.

Table 5-3
Differences between Eggs from Green Bay and Reference Common Tern Colonies When Incubated in the Laboratory

Source of Eggs	PCB Concentration (geometric mean, mg/kg wet weight)	Percent Hatching Success	Femur Length to Body Weight Ratio (x 100)	Percent Deformed Embryos and Hatchlings
Kidney Island CDF, Green Bay	10.0[a]	71%[b]	93.5[b]	11%[b]
Cut River, Michigan	4.7	85%	108.9	0%
Point aux Chenes, Michigan	4.0	73%	101.0	0%

a. Reported as statistically significantly different from Cut River and Point aux Chenes colonies.
b. Reported as statistically significantly different from Cut River colony.

Source: Hoffman et al., 1993.

There were no significant differences between the areas in egg concentrations of DDE, indicating that DDE was not the cause of the observed differences between colonies. Mercury was measured at significantly higher concentrations in the Green Bay eggs (0.76 mg/kg, wet weight) than in Cut River eggs (0.33 mg/kg, wet weight) or Pointe aux Chenes eggs (0.37 mg/kg, wet weight). However, mercury does not induce AHH activity, which was higher in Green Bay chicks than in reference area chicks.

Conclusions from common tern field study. As shown in Table 5-3, the single field study that has been conducted in the assessment area demonstrated that common terns had elevated tissue residues of PCBs, increased deformity rates, and perhaps reduced egg hatching success.

Caspian Tern

Field study descriptions. Five field studies have been reported that are relevant to evaluating the potential effects of contaminants on Caspian terns in the assessment area.

Ludwig and Ludwig (undated report b). This field study was performed on Gravelly and Gull islands in northern Green Bay, at three colonies in northern Lake Michigan, and at colonies in Thunder Bay and Saginaw Bay, Lake Huron, in 1986. The study compared clutch sizes, hatching success, productivity, and the incidences of developmental defects among the colonies. Mean clutch sizes were similar in the colonies on Gravelly and Gull islands and in the three Lake Michigan colonies (2.1, 1.9, 2.0, 2.1, and 2.0, respectively). Hatching success on Gravelly and Gull islands was 72% and 71%, respectively, compared to 81%-84% reported for the three Lake Michigan colonies. Productivity on Gravelly and Gull islands was 0.73 and 0.95 young fledged per nest, respectively, and 0.8-0.91 in the three Lake Michigan colonies. No developmental defects were found in chicks in any of the colonies.

Yamashita et al. (1993). Yamashita et al. collected 18 Caspian tern eggs from Gravelly and Gull islands in 1988. Of these, 13 (72%) contained "live normal" embryos with mean total PCB and DDE concentrations of 11 and 4 mg/kg wet weight, respectively. Three eggs (17%) were infertile, with mean total PCB and DDE concentrations of 10 and 3.2 mg/kg wet weight, respectively, and two eggs (11%) contained deformed embryos and mean total PCB and DDE concentrations of 11 and 6.3 mg/kg wet weight, respectively. The results of this study indicated no clear relationship between PCB and DDE concentrations and egg or embryo viability at the concentrations found in the two colonies.

Ludwig et al. (1996). In this study, live and dead Caspian tern eggs and chicks from five colonies throughout the Great Lakes (including from Green Bay) were examined between 1987 and 1991 for egg death rates and embryonic abnormalities. Egg mortality varied among the five study areas (North Channel of Lake Huron 25%, northern Lake Michigan 27%, Georgian Bay 27%, Green Bay 34%, and Saginaw Bay 42%). Egg mortality rates were highly correlated with TCDD-eq ($r = 0.8$), but not with total PCBs.

Of the 601 Green Bay dead eggs opened and examined, 124 (20.6%) of the embryos had developmental abnormalities. This compares with 17.3% in northern Lake Michigan, 13.2% in the North Channel of Lake Huron, 14.5% in Georgian Bay, and 22.8% in Saginaw Bay (which also is contaminated with PCBs). Of the abnormalities recorded in embryos from dead Green Bay eggs, 19% were edema, 39.2% were gastroschisis, 14.2% were bill defects, 4.7% were foot deformities, and 8.2% were other skeletal deformations.

Of the 162 Green Bay fertile live eggs opened and examined by Ludwig et al. (1996), 94% contained normal embryos and 5.5% contained deformed embryos. This compares with deformity rates of 11.8% in fertile live eggs from northern Lake Michigan and 30.4% in Saginaw Bay fertile live eggs. Five of the Green Bay deformities were club feet, three were gastroschisis.

In this study, 12,124 live Caspian tern chicks were also examined in the five study areas. Of these, 29 (0.02%) had deformities. The deformity rate varied little between Green Bay and the other areas (0.16% Green Bay, 0% Georgian Bay, and 0.18% North Channel of Lake Huron). Green Bay chick deformities comprised 62% clubbed feet and 38% gastroschisis. Ludwig et al. (1996) stated that no cross bills have been recorded in 26,819 Caspian tern chicks banded in the Great Lakes since 1960.

Thus, elevated rates of deformities were observed in dead eggs in Green Bay, whereas eggs and chicks that survived had lower deformity rates. This suggests that the deformities in Green Bay were associated with the viability of the embryos.

Mora et al. (1993). In this study performed in 1990, organochlorine concentrations in adult Caspian tern plasma were compared with age, productivity, and site fidelity (i.e., the proportion of birds that return to the natal area to breed) at eight colonies in Lakes Huron and Michigan (including Gravelly and Gull islands in Green Bay).

Mean total PCB concentrations varied from 0.91 mg/kg wet weight to 3.5 mg/kg wet weight among the study colonies, with the highest concentration in Green Bay. There were no significant intercolony differences in clutch size, hatching success, or fledging success. Of the 4,075 chicks examined at the Green Bay colonies, 0.17% had deformities (four had club feet, three had gastroschisis). This compares with 0.23% in colonies in northern Lake Michigan and 0.94% at Saginaw Bay.

On the basis of recapture rates of banded terns, Mora et al. (1993) argued that Caspian terns hatched in the Green Bay colonies displayed less site fidelity than terns in other regions. Mora et al. (1993) attributed this difference to contaminants, particularly PCBs.

Ewins et al. (1994). In this study, Caspian tern eggs were collected from 10 colonies across the Great Lakes (including Gravelly Island in Green Bay) and analyzed for organochlorine contaminants. Total PCB and DDE concentrations were highest in eggs from Gravelly Island and from Saginaw Bay. Nevertheless, hatching and fledging success were not significantly different at Gravelly Island compared with other areas (Table 5-4).

Conclusions from field studies on Caspian terns. Overall, less evidence exists for depressed reproductive rates among Green Bay Caspian terns than for Forster's and common terns. Of the four studies that examined reproductive injuries among Green Bay Caspian terns (Ludwig and Ludwig undated report b; Mora et al., 1993; Ewins et al., 1994; Ludwig et al., 1996) only one

Table 5-4
Hatching Success and Productivity among Great Lakes Caspian Terns

Study Site	Nests Studied	Egg PCB Concentration (mg/kg, wet weight)	% Hatching Success	Young Fledged/Nest
Gravelly Island, Green Bay	59	15.8	79%	1.07
High Island, northern Lake Michigan	56	not reported	85%	1.13
Cousins Island, North Channel of Lake Huron	28	14.6	47%	0.79
South Watcher Island, Georgian Bay	41	10.2	52%	0.83
Source: Ewins et al., 1994.				

(Ludwig et al., 1996) found reduced reproduction relative to reference conditions. The other three studies found no evidence of adverse effects on reproduction (though Mora et al. report possible behavioral effects among adult Caspian terns in Green Bay). Ludwig et al. (1996) found higher rates of deformities in Green Bay Caspian terns than in colonies not exposed to point source releases of PCBs. Other studies that investigated deformities did not find differences, although the Ludwig et al. study was the most detailed and comprehensive of the studies.

The available studies do not provide strong evidence that the reproductive success of Caspian terns nesting on Gravelly and Gull Islands has been adversely affected by PCB exposure. However, there is some evidence of increased deformity rates in Green Bay Caspian terns.

5.2 DOUBLE-CRESTED CORMORANT

5.2.1 Status and Ecology in Green Bay

Double-crested cormorant population trends in the Great Lakes can be divided into three temporal phases.

Initial colonization and early increases. Before the beginning of the 20th century, double-crested cormorants were unknown as a breeding bird in the Great Lakes. The colonization of the area began between 1913 and 1920 in Lake Superior (Environment Canada, 1995; Weseloh et al., 1995a, b), and probably involved birds spreading from colonies farther west. From this initial bridgehead, double-crested cormorants spread rapidly throughout the region until about 1950, when approximately 1,000 pairs bred in the Great Lakes, and control measures were initiated in an effort to protect fish stocks (Weseloh et al., 1995a).

Mid-century population declines. After 1950, the initial increases in cormorant numbers were followed by spectacular population reductions; by 1972, the Great Lakes population had been reduced by more than 80% (Weseloh et al., 1995a, b). From 1970 through 1974, double-crested cormorants had disappeared, or were close to disappearing, as a breeding species on Lake Michigan, and the total Great Lakes population was reduced to fewer than 150 pairs (Ludwig, 1984). In Wisconsin, the number of cormorants had decreased to 66 pairs by 1972 and the species was listed by the state as endangered (Hatch, 1995). These precipitous declines were accompanied by significant eggshell thinning and breakage. By 1970, eggshells in Ontario colonies were 30% thinner than normal, and in 1972, 95% of the eggs in Lake Huron colonies either disappeared or were broken (Environment Canada, 1995). Because of these losses, productivity in Great Lakes colonies had fallen to about 0.1 to 0.24 fledglings per breeding pair; 0.5 to 1.0 fledglings are required to maintain a breeding population (Ludwig, 1984; Ludwig et al., 1995). Based on the widespread and severe eggshell thinning and breakage, it is likely that the population decreases of the 1950s through early 1970s were caused by the toxic effects of DDE.

Post-1960s population resurgence. In the 1970s, following the ban on the use of DDT in North America, DDE levels in cormorant eggs in the Great Lakes began decreasing. By the late 1980s, egg DDE residues had decreased by more than 80% (Environment Canada, 1995), and populations of double-crested cormorants again increased. By 1992, approximately 3,000 pairs were breeding in Green Bay (Hatch, 1995). Thus, in only 20 years, the Green Bay population increased by a factor of at least 45 (assuming that the 66 Wisconsin pairs in the early 1970s were all in Green Bay). In the Great Lakes over this same period the increase was even greater, about 250-fold from about 150 nests to 38,000 nests (Weseloh et al., 1995a), a doubling time of about 2.5 years. This increase continued into 1994, when Weseloh et al. (1995b) estimated a total Great Lakes population of 60,000 pairs. As a result of this rapid rate of increase, approximately 60% of the world's population of double-crested cormorants currently breed in the Great Lakes (Hatch, 1995).

Residence Patterns and Migrations

Double-crested cormorants breeding in Green Bay are migratory, and most winter in the lower Mississippi Valley and the Gulf of Mexico (Dolbeer, 1991). In his analysis of band recoveries, Dolbeer found a high degree of mixing of midwestern nesting populations during winter; birds from Lakes Huron and Ontario and from Saskatchewan all wintered in the same areas of the lower Mississippi and coastal Texas.

The main breeding colonies of double-crested cormorants in the assessment area are on Cat, Jack, Hat, and Snake islands in Green Bay, and on Spider Island on the east side of the Door Peninsula (see Figure 2-4). Breeding cormorants arrive in Green Bay in April and remain in the area until September/October, when the return migration to the wintering area begins. First year and second year (nonbreeding) birds either remain in their wintering areas during their first summer or return later in the season than the breeding adults (Dolbeer, 1991).

5.2.2 Pathway and Exposure Analysis

A number of the ecological traits of double-crested cormorants predispose them to being potentially highly exposed to contaminants. First, double-crested cormorants begin to arrive in their breeding areas in Green Bay in April, approximately 3-4 weeks before the beginning of egg laying. There is no published information on the length of time it takes double-crested cormorants to form and lay a clutch of eggs. However, the closely related European shag is similar in size (hence metabolic rate), lays similarly sized eggs and clutches, and takes about 22 days from the beginning of egg formation to laying the last egg of a three-egg clutch (Grau, 1996). It is likely that the double-crested cormorant requires a similar time span. Thus, the birds arriving back in Green Bay in April, 3-4 weeks before egg laying begins, have sufficient time to form their eggs using food obtained locally rather than relying on reserves built up in the wintering area. Also, if cormorants forage close to their colonies during the pre-laying period, as they do during incubation and chick rearing, it is likely that the majority of females undergoing oogenesis will obtain their food from inner Green Bay, in the case of the Cat Island birds, or from the northeastern coast of the Door Peninsula, in the case of Spider Island birds. These ecological traits render the Green Bay double-crested cormorants vulnerable to exposure to local contaminants during the formation of the most sensitive life stage, the embryo.

As fish-eating predators, double-crested cormorants feed high in aquatic food chains. This renders them vulnerable to exposure to bioaccumulative contaminants. Also, alewives, one of the cormorants' major prey species in Green Bay and the Great Lakes (Ludwig and Ludwig, undated report a; Weseloh and Ewins, 1994; Neuman et al., 1997), are richer in lipid than other forage fish species (Oliver and Niimi, 1988; Rottiers and Tucker, 1982). By consuming lipid-rich prey, Green Bay cormorants increase their exposure to lipophilic contaminants such as PCBs.

Data reviewed in Chapter 4 show that double-crested cormorants in Green Bay have elevated PCB residues in their tissues. In this section we identify the environmental components through which Green Bay cormorants have been exposed to these PCBs. Specifically, we address three questions: What organisms constitute the principal diet of cormorants in the assessment area? Where do Green Bay cormorants feed? Are the prey of cormorants in Green Bay contaminated with PCBs?

Diet

A number of studies have shown that double-crested cormorants in the Great Lakes and adjacent areas eat mainly fish, in particular forage fish such as alewives and smelt (Ludwig and Ludwig, undated report a; Belonger, 1983; Hobson et al., 1989; Neuman et al., 1997). Of these, Neuman et al., Ludwig and Ludwig, and Hobson et al. showed that crayfish (*Orconectes* spp.) are also regularly found in small numbers in double-crested cormorant food samples; however, this could be due to secondary consumption (i.e., from the stomachs of fish that had been consumed).

The specific composition of cormorant diets can vary spatially and temporally (Neuman et al., 1997). This variability may reflect differences in the availability of prey species. For example, in Green Bay in 1983, alewives and yellow perch comprised more than 90% of 1,073 identifiable fish obtained from regurgitates of nestling cormorants on Willow Island (alewives, 51.6%; yellow perch 39.3%) (Belonger, 1983). At Gravel, Fish, and Spider islands, alewives similarly comprised a high proportion of the diet (69.2% of identifiable fish); however, yellow perch comprised only 1.9% (Belonger, 1983). Yellow perch were replaced in the cormorant diet at these locations by sculpin (*Cottus sp.*) (11.8% of identifiable fish), ninespine stickleback (*Pungitius pungitius*) (8.1% of identifiable fish), Johnny darter (*Etheostoma nigrum*) (5.2% of identifiable fish), and spottail shiner (*Notropis hudsonius*) (1.9% of identifiable fish). The few yellow perch in the latter samples was probably because the shallow water habitat preferred by this fish species is not readily available at Gravel, Fish, and Spider islands. Alewives and smelt were the most frequent food items in regurgitates from adult and young cormorants in northern Green Bay colonies from 1986 to 1988 (Ludwig and Ludwig, undated report a).

Foraging Areas

Custer and Bunck (1992) tracked the foraging flights of cormorants from the Cat and Spider Island colonies in Green Bay and found that the birds typically obtain their fish prey from waters relatively close to the colonies. Foraging flights from Cat Island were restricted to within 40 km of the colony, and the mean foraging flight distance was 2 km. Most of the foraging flights from Cat Island ended with the birds landing in the central and western inner bay area (Figure 5-5). Many foraging flights ended at the confluence of the Fox River with Green Bay, and less than 1% of birds flew up the river. Cormorants from Cat Island tended to forage in shallow water areas (less than 1.8 m deep) and avoided deeper water.

Double-crested cormorants from Spider Island also tended to forage close to the colony (Custer and Bunck, 1992). The maximum distance flown from the colony was 12 km, and the mean was 2.4 km. The majority of the Spider Island birds foraged off of the east coast of the Door Peninsula (Figure 5-6). They preferred water depths of less than 9.0 m, but avoided depths of less than 1.8 m (the preferred depth for the Cat Island birds). No Spider Island cormorants were recorded flying into Green Bay to forage during the study.

Prey Contamination

USFWS (1993) collected stomach contents from adult cormorants on Cat Island during the 1988 breeding season and found concentrations of total PCBs averaging 3.3 mg/kg (wet weight). These data show that, at least during the breeding season, double-crested cormorants in Green Bay ingest prey contaminated with PCBs. USFWS (1993) also showed that adult cormorant PCB body residues approximately doubled during the 1988 breeding season (Figure 5-7), indicating a local source of the PCBs. These data confirm that exposure to PCBs is dietary.

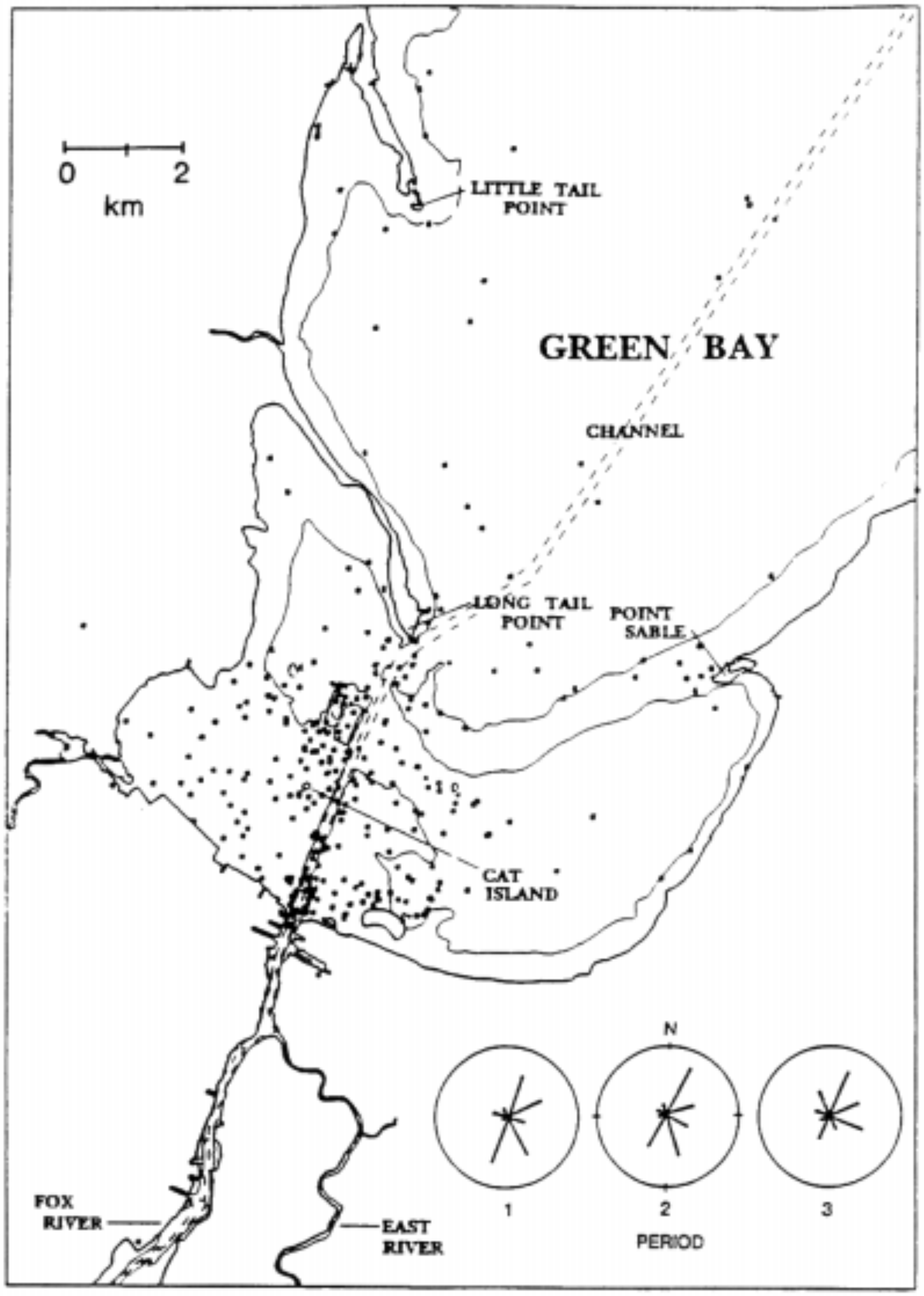


Figure 5-5. Foraging sites of double-crested cormorants from the Cat Island Colony in 1990. Each point represents the foraging destination of a bird tracked from the colony.
Source: Custer and Bunck, 1992.

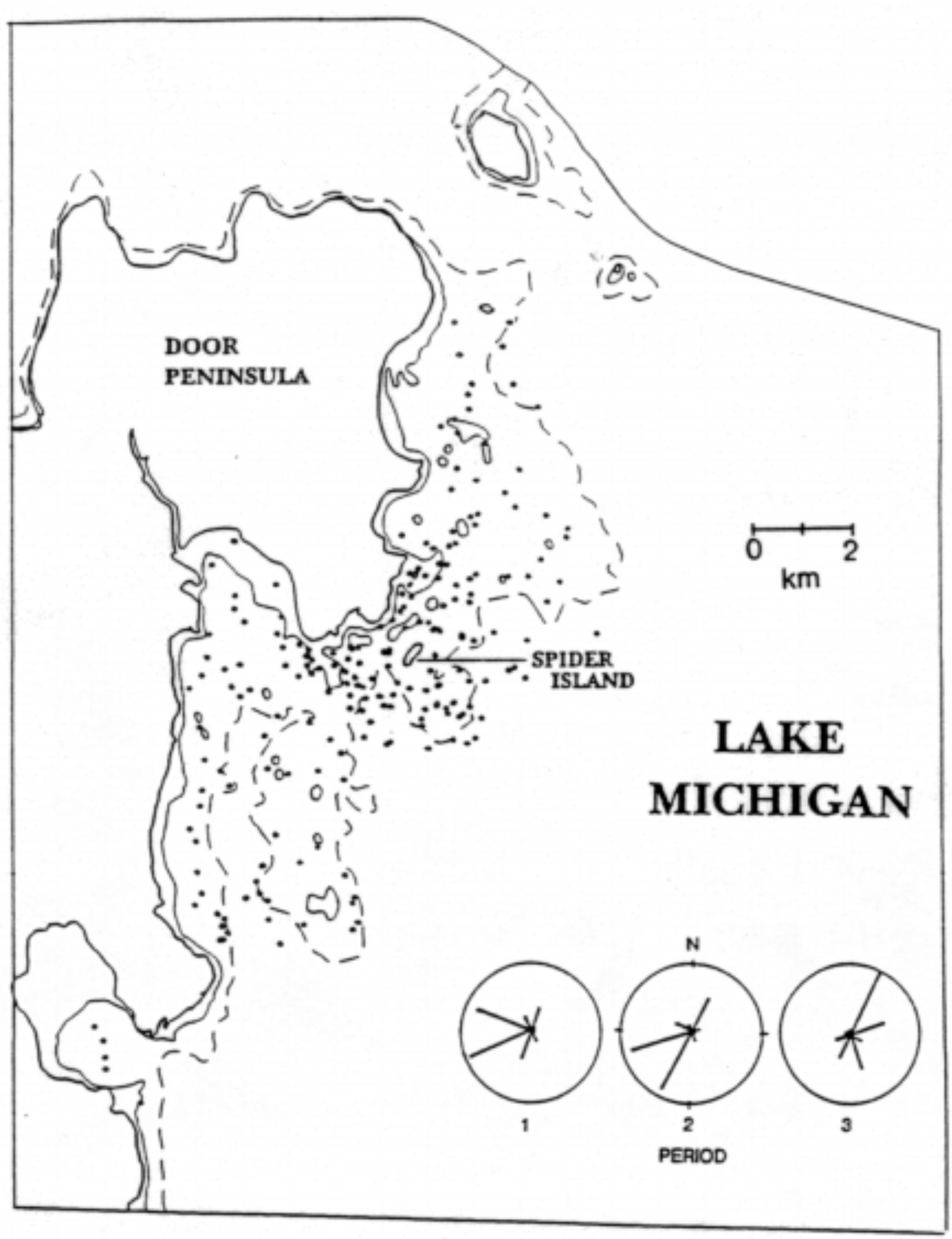


Figure 5-6. Foraging sites of double-crested cormorants from the Spider Island Colony in 1990. Each point represents the foraging destination of a bird tracked from the colony.
Source: Custer and Bunck, 1992.

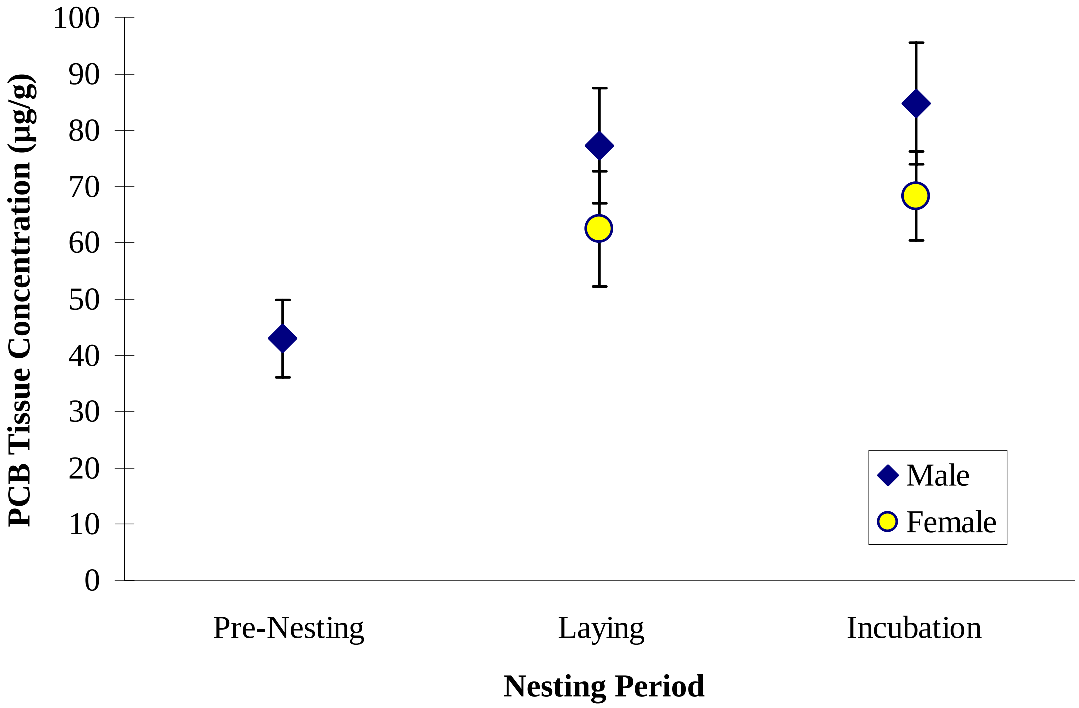


Figure 5-7. Whole-body concentrations of PCBs in cormorants during three phases of the nesting cycle in 1988. Bars equal means plus or minus 1 standard error.
Source: USFWS, 1993.

As already shown, the fish diet of double-crested cormorants in Green Bay is restricted to a relatively small number of forage fish species. These are mainly alewives and yellow perch at the Cat Island colony, and alewives and sculpin at the Spider Island colony. Sampling data show that alewives in Green Bay are contaminated with PCBs (Figure 5-3).

Thus, the PCB pathway documentation for Green Bay double-crested cormorants includes observations that during the incubation and chick-rearing phases of the breeding cycle, adult double-crested cormorants forage in areas of Green Bay that contain PCB-contaminated fish, cormorants ingest Green Bay fish that are contaminated with PCBs, and cormorant PCB tissue residues increase during the breeding season.

5.2.3 Field Studies of Injuries to Green Bay Double-Crested Cormorants

This section describes field studies addressing reproductive malfunctions and physical deformations in Green Bay double-crested cormorants.

Malfunctions in Reproduction

Reproductive success in double-crested cormorants nesting in Green Bay has been compared with that in reference colonies in three studies. In two of these studies, reproductive success was found to be lower in Green Bay colonies.

Ludwig and Ludwig (undated report b). This 1986 study compared hatching success rates in double-crested cormorant colonies in Lakes Huron, Superior, and Michigan, and northern Green Bay (Gravelly, Little Gull, and Snake islands). Hatching success varied from 63% of eggs laid to 74% of eggs laid (Table 5-5). Neither the proportions of eggs that failed to hatch nor hatching success differed between colonies.

Table 5-5
Hatching Success of Double-Crested Cormorants in Great Lakes in 1986[a]

Colony Location	Number of Eggs Studied[b]	Number (%) of Eggs Hatched	Number (%) of Eggs Disappeared	Number (%) of Eggs Dead
All Lake Huron colonies	126	96 (76)	8 (6)	22 (17)
Lake Michigan, Beaver Island	196	142 (72)	23 (12)	31 (16)
Northern Green Bay	173	114 (66)	20 (12)	39 (22)
Lake Superior colonies	65	45 (69)	6 (9)	14 (21)

a. Adapted from data in Ludwig and Ludwig, undated report b.
b. Excludes eggs that were accidentally pierced by parent birds.

Tillitt et al. (1992). This study compared hatching success rates between 1986 and 1988 in 12 double-crested cormorant colonies in Lakes Huron, Michigan, Superior, Ontario, and Winnipegosis and in Green Bay, and investigated the relationships between egg mortality, PCB concentrations, and egg H4IIE activation. Egg mortality varied between 8% and 39%, with the highest rates in Green Bay colonies (Little Gull, Snake, Gravelly, and Spider islands) and the lowest at Lake Winnipegosis. Regression analysis revealed a significant, though relatively modest, positive relationship between total PCB concentrations in eggs and egg mortality ($r = 0.319$, $p = 0.045$). However, when the analysis compared H4IIE bioassay-derived TCDD-eq in eggs with hatching success, the relationship was strengthened ($r = 0.703$, $p = 0.0003$). The H4IIE sample preparation process used in this investigation screened out both dioxins and furans (H4IIE is not sensitive to DDE). The authors of the study concluded that the elevated egg mortality rates and reduced hatching success in the more contaminated colonies were caused by the effects of dioxin-like PCBs.

Larson et al. (1996). This study compared the hatching success of double-crested cormorant eggs at Spider Island in Green Bay in 1988, 1989, and 1990 with the hatching success at Lake Winnipegosis in 1989 and 1990. Hatching success at Spider Island for the three consecutive years was 65.4% (1988), 55.2% (1989), and 57.7% (1990), compared with 75.7% (1989) and 64.1% (1990) at Lake Winnipegosis. Hatching success was significantly greater in larger clutches, and Lake Winnipegosis clutches were, on average, 0.2 eggs larger than Spider Island clutches. However, covariance analysis (in which clutch size was included as a categorical variable) revealed that hatching success was significantly lower at Spider Island in 1989 and 1990, even when clutch size was controlled for. Of 5,759 chicks examined at the Spider Island colony, 0.8% had bill deformities, compared with 0.06% at Lake Winnipegosis. This more than ten-fold difference was reported as statistically significant.

Total PCB concentrations and TCDD-eq were significantly higher in Spider Island eggs (7.8 mg/kg and 138 pg/g, respectively) than in Lake Winnipegosis eggs (1.0 mg/kg and 19 pg/g, respectively). However, *within* the Spider Island colony, neither PCBs nor TCDD-eq were significantly correlated with hatching success or the incidence of deformities among nestlings.

Depredation of seabird nests by gulls following disturbance by observers is a potential problem at many seabird colonies. In general, the greater the disturbance, the greater the opportunity for gulls to depredate eggs. To minimize this effect, the investigators visited the Spider Island colony only after dark (a time when gulls are less active). Nevertheless, the Spider Island colony was visited more frequently (13 visits in 1989) than the Lake Winnipegosis colonies (4 visits in 1989), and the success of the attempt to minimize nest predation by nocturnal visits was not evaluated. Thus, the contribution of observer disturbance to the observed differences in hatching success cannot be determined.

In addition to the above three studies, another study evaluated double-crested cormorant reproduction in Green Bay only (i.e., no reference site data were collected).

Custer et al., in press. During 1994 and 1995, the investigators in this study examined relationships between PCB, DDE, and dieldrin concentrations and hatching success, chick deformity rates, eggshell thickness, and biomarker activity in cormorant eggs from Cat Island in Green Bay. No reference colonies were sampled. Single pipping eggs were removed from each of the study nests. A subset of these eggs were analyzed for chemical contaminants; previous measurements had shown that approximately 85% of the total egg variability in contaminant concentrations in cormorant eggs in Green Bay was between-clutch variation (USFWS, 1993). Measurements made on these eggs included PCB, DDE, and dieldrin concentrations, eggshell thickness, and EROD activity in embryo livers. The fate of the eggs remaining in the study nests was monitored, as was that of the chicks that hatched. Study nests were divided into four groups on the basis of their success: nests that contained eggs with one or more dead embryos, nests that contained one or more infertile eggs, nests in which all the eggs hatched successfully, and nests that contained eggs with deformed embryos.

Table 5-6 shows the total PCB and DDE concentrations among the four nest categories. Two-way ANOVAs performed by the study investigators determined that there were no significant differences among the PCB concentrations in the four nest categories ($p = 0.05$). However, DDE concentrations did differ significantly among the four categories ($p = 0.03$). Mean concentrations of DDE in sample eggs were significantly higher in nests that contained dead embryos than in nests in which all the eggs hatched or contained deformed embryos. Total PCBs in the sample eggs were significantly correlated with EROD activity. The overall egg hatching success on Cat Island in 1994 and 1995 was 68% (1,067 of 1,570 eggs).

Table 5-6
Geometric Mean Total PCB and DDE Concentrations (mg/kg, wet weight) in Four Categories of Cormorant Nests on Cat Island, Green Bay in 1994 and 1995

Nest Category	Number of Nests	DDE	PCBs
Dead embryos	39	3.9	11.4
Infertile eggs	5	2.8	13.6
All eggs hatched	30	2.8	12.1
Deformed embryos	6	2.2	10.2
Source: Custer et al., in press.			

The study authors conducted a series of logistic regressions to evaluate whether DDE, PCBs, dieldrin, or eggshell thickness was associated with differences in percent hatching success of the cormorant eggs. Only DDE and eggshell thickness were found to have significant associations ($p < 0.002$) and < 0.008, respectively). Neither total PCBs nor dieldrin was found to have significant associations with hatching success ($p < 0.84$ and < 0.29, respectively). However, egg PCB concentrations were significantly negatively correlated with both egg volume ($r = -0.39$, $p < 0.001$) and embryo weight ($r = -0.28$, $p < 0.04$). DDE was not significantly correlated with egg volume and egg weight.

Although hatching success was significantly correlated with eggshell thickness and significantly negatively correlated with DDE concentration, these two variables explained relatively little of the total variability in hatching success (2.2% and 13%, respectively). The study authors concluded that DDE may be reducing the hatching success of only the most highly contaminated eggs, and that other factors may be responsible for the majority of the egg failure observed at the colony.

Physical Deformations

A number of studies have reported on physical deformations among Green Bay double-crested cormorants.

Langenberg (1990). In this companion study to Larson et al. (1996), the author examined 183 late-term cormorant eggs from Spider Island and 125 from Lake Winnipegosis: 95% of the Spider Island eggs contained embryos, of which 18.3% were dead, and 98% of the Lake Winnipegosis eggs contained embryos, of which 11.4% were dead. Our analysis of these data found that the differences were not statistically significant ($\chi^2 = 2.7$, 1df, $p > 0.25$). Two examples of cross bills were found, both in Lake Winnipegosis embryos. On Spider Island, 10.9% of the embryos examined had edema, similar to the 16.3% of Lake Winnipegosis embryos. Seven of the Spider Island embryos had petechial hemorrhages; none were found in the Lake Winnipegosis embryos. However, because hemorrhaging in several cases was noted only after handling, Langenberg concluded that these were a result of the examination process rather than toxicological action.

Fox et al. (1991). In this analysis of deformity rates among cormorant nestlings throughout the Great Lakes between 1979 and 1987, 31,168 chicks from 42 colonies were examined. The overall rate of head and bill deformities was 0.22%. However, the local rate varied, and the highest rate was found in Green Bay (60 of 11,520 chicks, 0.52%). The rates found in other areas were 0.03% (Lake Ontario), 0.05% (Lake Superior), 0.006% (Alberta and Saskatchewan), and 0.02% (Lake of the Woods and Lake Nipigon). The Green Bay deformity rate was significantly higher than the rates at the Lake of the Woods and Lake Nipigon, and at prairie colonies in Alberta and Saskatchewan. Head and bill defects were found in 8 of 11 (73%) Green Bay colonies, but in only 6% of reference colonies.

Yamashita et al. (1993). In this study carried out in 1988, the investigators collected late-term, incubated cormorant eggs from Little Gull Island in Green Bay and elsewhere in the Great Lakes (including from colonies not exposed to point source releases of PCBs). Eggs were examined and separated into four categories: live normal, infertile, containing a deformed embryo, and not incubated. Of the 41 Green Bay eggs examined, 26 were fertile; 78% of these contained normal young and 31% contained deformed embryos (compared with about 90% and 6%, respectively, in eggs collected elsewhere). The total PCB concentrations in Green Bay eggs were 7.3 mg/kg wet weight (live normal), 7.3 mg/kg wet weight (infertile), and 6.6 mg/kg wet weight (deformed). Total PCB concentrations in eggs from Lake Superior and the North Channel (colonies unlikely to be affected by point source releases of PCBs) varied from 3.6 to 7.3.

Larson et al. (1996). The incidences of bill deformities in cormorant nestlings were compared between Spider Island (1988, 1989, and 1990) and Lake Winnipegosis (1989, 1990). At Spider Island, 5,759 chicks were examined, and approximately 24,736 were examined at Coffee Island, Coffee Island Reef, Bachelor’s Island, Sugar Island Reef, and Hay Island Reef in Lake Winnipegosis. Bill defects were significantly more frequent ($p < 0.001$) at Spider Island (0.7%) than at Lake Winnipegosis (0.06%).

*Ludwig et al. (1996).*This study was based on measurements taken in several Great Lakes cormorant colonies between 1987 and 1991. In Green Bay, 24.8% of 660 dead eggs (eggs that were dead in the nest) contained deformed embryos. Of these, 65% had subcutaneous edema and hemorrhaging, and 18.9% had bill defects. These data are reported in Table 1 of Ludwig et al. (1996). Our statistical tests on the raw data in this table indicated significant differences in deformity rates between colonies ($\chi^2 = 31.8$, 6 df, $p < 0.001$). Our tests also showed that the embryo deformity rate in dead eggs in Green Bay (24.8%) was significantly higher than that found in colonies in the three areas that were least likely to be exposed to point source releases of PCBs: southeast Lake Superior (11.3%, $\chi^2 = 23.0$, 1 df, $p < 0.001$); Georgian Bay on northern Lake Huron (15.5%, $\chi^2 = 8.5$, 1 df, $p < 0.02$); and the North Channel of Lake Huron (18.7%, $\chi^2 = 6.6$, 1 df, $p < 0.05$). Of 315 live eggs from Green Bay, 14.3% contained deformed embryos, compared with 4% at Lake Winnipegosis. The proportions of the various types of embryo deformities found in the Green Bay live eggs were not reported. Of 7,975 Green Bay cormorant nestlings, 0.6% were deformed. These had mainly crossed bills (53% of deformities) and dwarfed appendages (20.4% of deformities).

Ryckman et al. (in press). In this study of organochlorine contamination and bill defects among cormorants nesting in the Canadian Great Lakes, no significant associations were found between regional rates of dill deformities and total PCB concentrations in eggs.

Custer et al. (in press). In addition to the results presented previously, this study also investigated relationships between deformity rates in chicks of double-crested cormorants and organochlorine residues. Eggs from nests in which one or more embryos were deformed did not have significantly higher concentrations of either PCBs or DDE than eggs from nests in which no deformed young were found. Custer et al. also reported that the frequency of bill deformities among nestlings at Cat Island in 1994-1995 (0%; 0 of 632) was generally lower than those reported from cormorant colonies in northern Green Bay during the period 1979-1990 (0.6%-0.7%). This is in spite of the fact that Cat Island cormorants had higher egg PCB concentrations [Cat Island 1994-1995: mean of 13.6 mg/kg, wet weight; Spider Island 1988 and 1989: 5.3 and 7.7 mg/kg wet weight, respectively (Tillitt et al., 1992; Larson et al., 1996)]. However, the deformity rates and egg PCB concentrations in northern Green Bay cormorant colonies in the years in which the Custer et al. study was performed at Cat Island are not known.

5.2.4 Data Evaluation

Evidence of Adverse Effects in Green Bay Double-Crested Cormorants

Reproductive malfunctions. The evidence that Green Bay cormorants have suffered adverse reproductive effects is strong. Two independent studies (Tillitt et al., 1992; Larson et al., 1996) demonstrated that hatching success rates are significantly lower in Green Bay nests than in control areas. One other study (Ludwig and Ludwig, undated report b) attempted to compare nesting success of cormorants in Green Bay with reference sites. Analyses of the data presented in that study showed no significant differences between hatching success in Green Bay and in other sites.

In addition to the field studies described above, one study, Powell et al. (1997), attempted to reproduce the impaired hatching success seen in the field by injecting cormorant eggs in the laboratory with a PCB congener. They injected cormorant eggs collected from Lake Winnipegosis (a site where cormorants are not exposed to point source releases of PCBs) with doses of PCB 126 and an extract derived from Green Bay cormorant eggs. The authors found that injections of PCB 126 significantly reduced hatching success of Lake Winnipegosis eggs but only at doses an order of magnitude greater than the highest concentrations of PCB 126 that have been found in Green Bay cormorant eggs. However, it should be noted that Powell et al. did not inject the Lake Winnipegosis eggs with the mixture of congeners found in Green Bay eggs. Also, there is uncertainty regarding how closely eggs injected with contaminants or extracts mirror the "natural" uptake and effects of contaminants in the field. Overall, however, the Powell et al. study does not support PCBs as the cause of the reduced hatching success observed among assessment area double-crested cormorants.

Physical deformations. The strength of the evidence that Green Bay cormorants have deformity rates that are elevated with respect to background is strong, depending on which deformity is addressed and what background level is assumed.

Studies have shown that crossed bills have occurred among Green Bay cormorant chicks (Fox et al., 1991; Larson et al., 1996; Ludwig et al., 1996). The rate of bill deformities found in Green Bay is substantially higher (by a factor of about 10) than that observed at reference sites. The background rate of bill deformities that is typically observed is usually less than 0.1% (Fox et al., 1991; Ryckman et al., in press). However, the evidence that such a low rate is representative of all appropriate reference colonies is not entirely unambiguous. Ross and Weseloh (1988) measured a large degree of spatial variability in bill deformity rates among Lake Winnipegosis colonies. The Sugar Island colony in 1988 had a rate of 3.9%, which is much higher than the highest deformity rate ever measured in Green Bay, or any other location. It should be noted, however, that the Ross and Weseloh study is the only study, thus far, that has found such high rates of deformities in cormorants from areas not affected by point sources of contaminants. Indeed, in their research on Lake Winnipegosis, Larson et al. (1996) included Sugar Island among their sampling locations and still found that the overall Lake Winnipegosis deformity rate was less than 0.1%. The relevance of the Ross and Weseloh study is, therefore, uncertain. All studies that have assessed bill deformity rates in both Green Bay and reference colonies have found higher rates in Green Bay.

Increased incidences of edema of the head and neck (which constitutes the majority of the deformities reported in double-crested cormorants) and hemorrhaging are less certain than crossed bills. Ludwig et al. (1996) found that 16.2% of dead eggs from Green Bay had embryos with edema (mainly of the head and neck), and only 6% of live nestlings showed hemorrhaging. These data indicate that the deformity rate among live chicks may underestimate the true population rate (since many deformed embryos may die before hatching). However, Langenberg (1990) examined live eggs from Green Bay and was unable to find abnormal occurrences of edema. She concluded that any hemorrhaging that she recorded was an artifact of her handling the embryos.

In conclusion, bill deformity rates among cormorant embryos and nestlings in the assessment area have exceeded background rates. However, the occurrence of other types of deformities is not as conclusive.

Evidence of PCB Effects

This section evaluates the evidence that PCBs are responsible for the observed adverse effects on reproduction and bill deformity rates. Chapter 4 presented data showing that total PCB and PCB congener concentrations measured in Green Bay cormorant eggs are at or above concentrations shown to cause avian toxicity in literature studies. However, because PCB concentrations can be correlated with concentrations of other lipophilic compounds (Bosveld, 1995), the likelihood of the adverse effects being caused by other contaminants must also be evaluated.

Reproductive malfunctions. The four main groups of candidate contaminants that potentially could cause the effects seen in the Green Bay cormorants are PCDDs, polychlorinated dibenzofurans (PCDFs), DDE, and PCBs.

Using the data presented in Yamashita et al. (1993), it is possible to evaluate the relative contributions of PCDD and PCDF to total dioxin-like toxicity. The analysis in Table 5-7 shows that the contributions to total toxicity by TCDD and TCDF are less than 5%. A similar result was obtained in the Kubiak et al. (1989) study of Forster's terns on Green Bay, where PCDDs accounted for less than 10% of total toxicity (no PCDFs were found in the tern eggs) and PCBs for more than 90%. Also, Tillitt et al. (1992) eliminated PCDD and PCDF residues when preparing Green Bay cormorant egg samples for H4IIE analysis and found significant correlations between the responses elicited by the extract and hatching success at Great Lakes colonies. These results indicate that PCDDs and PCDFs are unlikely to be important contributors to the adverse effects reported in the Green Bay cormorants.

Table 5-7
Percent Contributions to Total TCDD-EQ by PCB, TCDD, and TCDF Congeners in Green Bay Double-Crested Cormorant Eggs

	Percent Contribution[b]							
TEFs Used to Estimate Percent Contribution	**PCB 77**	**PCB 105**	**PCB 118**	**PCB 126**	**PCB 156**	**2378 TCDD**	**12378 TCDD**	**2378 TCDF**
WHO Avian TEFs[a]	47.2	2.6	<1	44.1	<1	2.4	<1	<1

a. Van den Berg et al., 1998.
b. Original data from Yamashita et al., 1993.

Concentrations of PCBs and DDE in Green Bay cormorant eggs are correlated (Custer et al., in press, $r = 0.53$, 73 df, $p < 0.001$). Also, although little research has been carried out on the effects of DDE on avian embryos at levels of exposure below those known to result in eggshell thinning and breakage, there is evidence from a field study of common terns that DDE concentrations of between about 3 and 7 mg/kg wet weight may result in embryo mortality (Fox, 1976). Thus, DDE may exert effects on embryo mortality other than those associated with eggshell breakage that are similar to those that may be caused by exposure to PCBs. Possible mechanisms for this are changes in shell microstructure that are associated with comparatively low levels of thinning, with consequent disruptions of gaseous transfer, or with direct embryotoxic effects (Fox, 1976).

The three studies that have attempted to rigorously address the potential effects of contaminants on cormorant hatching success in the field are Tillitt et al. (1992), Larson et al. (1996), and Custer et al. (in press). Tillitt et al. concluded that PCBs explained much of the observed variability in hatching success between Great Lakes cormorant colonies and were responsible for the reduced hatching success seen in Green Bay. This conclusion was based on the relationships between mean PCBs and H4IIE results and hatching success *between colonies*. In contrast, both the Larson et al. and the Custer et al. studies suggested that PCBs did not explain differences in hatching success among Green Bay cormorants. The conclusions were based on the lack of significant correlations when PCB concentrations were compared with individual nest reproductive success *within a colony*. In addition, Powell et al. (1997) was unable to reproduce embryo mortality among cormorants in the laboratory when injecting eggs with doses of PCB 126 comparable to those seen in the assessment area.

The discrepancy between the results of these studies may be at least partly a function of the different study approaches. Both Larson et al. (1996) and Custer et al. (in press) used the sample egg technique, in which the reproductive success of individual nests within a colony was measured and compared with the contaminant concentrations in an egg removed from the same nest. Because many factors other than contaminants affect the reproductive success of individual nests, such as nest abandonment, predation, accidental egg breakage, and parental experience, individual nests within a colony have a high degree of variability that is not expected to be explained by contaminant concentrations. Indeed, Custer et al. (in press) found that DDE explained only 13% of the variability in individual nest success. Therefore, in this approach the power to detect effects of contaminants on the inherently variable success of individual nests is low. In contrast, Tillitt et al. (1992) compared the mean reproductive success across colonies with the mean contaminant concentrations in eggs taken from the colony. Comparing mean colony success with mean colony contaminant concentrations across different colonies reduces the variability in the reproductive success data and allows for a greater ability to detect the effects of contaminants on reproductive success. However, it should be pointed out that Tillitt et al. (1992) did not compare DDE concentrations and reproductive success between colonies. Therefore, because of the different study approaches, the findings of Larson et al. (1996) and Custer et al. (in press) that PCBs are not correlated with reduced hatching success within the Green Bay colony are not inconsistent with the finding of Tillitt et al. (1992) that PCBs and H4IIE are correlated with mean hatching success across colonies and that the Green Bay colony had the highest PCBs and H4IIE activity and lowest mean reproductive success.

Custer et al. also indicated that in Green Bay cormorants DDE concentrations appeared to explain a greater percentage of the variability in hatching success than PCBs. However, such a small component of the total variance in hatching success is apparently explained by contaminants that detecting the effects of individual contaminants would be difficult. The fact that a significant effect on hatching success was found by Custer et al. for DDE, but not for PCBs, might only reflect this difficulty rather than the likelihood that only DDE was affecting the cormorants. Also, it cannot be definitely concluded from the data in Custer et al. that PCBs had no effect on hatching success, since the only measure of PCB contamination that was analyzed was total PCBs. Using TCDD-eq in the analysis may also have strengthened the relationship between PCBs and hatching success, as was found in the Tillitt et al. (1992) study.

Furthermore, Custer et al. (in press) found significant negative correlations between egg PCB concentrations and egg volume and embryo weight, indicating that PCBs were exerting some effects on the breeding biology of Cat Island cormorants in 1994-1995. Although they did not find a negative relationship between PCBs and hatching success that was statistically significant at $p < 0.05$, the probability of the correlation that they did find being due to chance was 0.13.

Overall, the evidence shows that exposure to PCBs may have resulted in reduced hatching success among Green Bay cormorants. However, the Custer et al. (in press) study shows that the effects observed in the assessment area are unlikely to be due to PCBs alone and that DDE has contributed to the adverse effects.

Physical deformations. PCBs have been shown in controlled experiments to cause deformations in avian embryos. These have included deformations of the head and bill and legs. However, DDE is not known to cause such deformities in avian embryos. The other candidate contaminants that could cause such deformities (PCDD and PCDF) do not occur at concentrations that could contribute significantly to the deformity rates observed among Green Bay cormorants (see previous discussion).

Summary and Conclusions

The data reviewed in this report indicate that exposure to elevated concentrations of PCBs has most likely resulted in adverse effects to double-crested cormorants in Green Bay, including reduced reproductive success and embryonic deformations. However, the evidence for this is not as conclusive as that for Forster’s terns. Also, it is likely that other contaminants complicate the attribution of effects.

5.3 BLACK-CROWNED NIGHT HERON

Black crowned night-herons are opportunistic feeders. Their diet often consists mainly of fish and other aquatic organisms, although they also eat terrestrial invertebrates and the nestlings of other colonial birds (Cramp and Simmons, 1977).

Three studies summarized in Chapter 4 of this report showed that black-crowned night herons in the assessment area have been exposed to PCBs: Heinz et al. (1985), Rattner et al. (1993), and Custer and Custer (1995).

Two studies investigated adverse effects in Green Bay night herons.

Hoffman et al. (1993). In this study, five pipping eggs were collected from the colony on the CDF in 1984, and the morphologies of their chicks were compared with others from a captive control colony at Patuxent Wildlife Research Center (PWRC) in Maryland. The two groups did not differ in egg or embryo weights. The Green Bay chicks had 36% larger livers than the PWRC chicks, but this difference was not significant. However, Green Bay chicks had significantly higher liver to body weight ratios than the PWRC chicks.

This study also investigated biomarker activity in the livers of the two groups of embryos. It found that AHH activity was significantly higher in the livers of the Green Bay chicks by a factor of three. No PCB concentration measurements were carried out to determine if the morphological and biomarker differences between the two groups of chicks were associated with differences in contaminant loads.

Rattner et al. (1993). In this study, PCB concentrations and biomarker activity were measured in black-crowned night heron chicks from Cat Island, a reference site in Virginia, and two islands in San Francisco Bay. The Green Bay chicks had the highest levels of biomarker activity (AHH, EROD, BROD, ECOD, CYP1A, and CYP2B) and the highest PCB concentrations (9.32 mg/kg wet weight, with a range of 2.4-53 mg/kg wet weight). The Green Bay PCB concentrations were significantly greater than those found in all of the other colonies. No morphological abnormalities were reported.

These studies show that black-crowned night herons in Green Bay have been exposed to PCBs at levels that exceed background concentrations. One study (Hoffman et al., 1993) also suggests that Green Bay black-crowned night herons may have been injured (enlarged livers).

5.4 TREE SWALLOW

Tree swallows are insectivorous birds that feed on the emerging adult life stages of aquatic insects. Thus, because of their diet, tree swallows nesting close to the Lower Fox River and Green

Bay might be expected to be exposed to PCBs. Only one study of contaminants and breeding success has been performed on this species in the assessment area (Custer et al., 1998). This study showed that pipping hatchlings and nestlings of tree swallows nesting close to the Lower Fox River and inner Green Bay had significantly higher PCB concentrations than pipping hatchlings and nestlings from reference areas.

The breeding success of the Green Bay and Lower Fox River tree swallows was not significantly different from that of tree swallows at the reference areas. Nor were any embryo or nestling differences in weight or body condition found. No deformities were observed. These data suggest that the PCBs measured in the tree swallow hatchlings and nestlings in the assessment area were not causing adverse effects.

5.5 RED-BREASTED MERGANSER

White and Cromartie (1977) and Haseltine et al. (1981) showed that in the 1970s PCB concentrations in red-breasted merganser eggs on islands off the Door Peninsula were high. Heinz et al. (1983), a companion study to Haseltine et al. (1981), found, however, that high PCB residues in 1977 and 1978 were not correlated with rates of nest desertion, hatching success, or duckling production. Also, Heinz et al. (1994) found no significant difference between merganser hatching success in 1977-1978 and 1990, despite egg PCB concentrations having decreased by 60%. Thus, the available data do not indicate that the elevated PCB concentrations in Green Bay red-breasted merganser eggs in the late 1970s were affecting reproduction in this species.

5.6 BALD EAGLES

5.6.1 Status and Ecology in the Lower Fox River and Green Bay

Bald eagle population trends in the Great Lakes can be divided into two phases: mid-century declines and post-1960s resurgence.

Midcentury population declines. In the middle of this century, bald eagle populations throughout the contiguous United States and much of Canada underwent drastic reductions. A chronology of these population declines was reported by Nisbet (1989):

> Reproductive impairment in the bald eagle was first reported in Florida in 1947 (Broley, 1958) and became widespread during the 1950s and 1960s (Sprunt, 1963; Sprunt and Ligas, 1966; Stickel et al., 1966; Postupalsky, 1971; Grier, 1972; Wiemeyer et al., 1972, 1984; Sprunt et al., 1973). By 1970, a number of local populations in the lower 48 states of the USA and in southern Canada had been markedly reduced or extirpated (Broley, 1958; Howell, 1963; Postupalsky, 1971;

> Grier, 1972; Sprunt et al., 1973; USDI, 1974; Kiff, 1980); populations in Alaska and parts of western and northern Canada were generally unaffected. . . . Several studies have shown the inter-relationships between eggshell-thinning, reproductive impairment, populations declines, and levels of contamination with DDE and other organochlorines (Postupalsky, 1971; Wiemeyer et al., 1972, 1984; Sprunt et al., 1973).

In the Great Lakes, bald eagles were extirpated from coastal areas and anadromous runs of Lakes Huron, Michigan (including Green Bay), Ontario, and Superior and nearly extirpated from Lake Erie by the late 1960s (Bowerman, 1993).

Post-1960s population resurgence. Since the mid-1970s, when the use of DDT, PCBs, and other organochlorine compounds was banned in North America, bald eagles have increased in number. The lessening of the eggshell-thinning effects of DDT's metabolite, p,p'-DDE, has been a major reason for the current resurgence of bald eagle populations in temperate North America (Grier, 1982; Postupalsky, 1985; Colborn, 1991; Best et al., 1994; Bowerman et al., 1995). The number of bald eagle breeding pairs within 8.0 km of the Great Lakes coasts increased from 26 in 1977 to 134 in 1993. Furthermore, the reproductive productivity of these birds increased from 0.23 young per occupied nest in 1977 to 0.87 in 1993 (Bowerman, 1993). Bald eagles breeding within 8.0 km of the Lake Michigan coast or along streams open to Great Lakes fish runs also increased over this period, from 2 pairs in 1977 to 28 pairs in 1993. The productivity of these birds increased from 0.0 young per occupied nest to a high of 0.89 in 1987, but was only 0.46 in 1993 (Bowerman, 1993).

Annual monitoring data collected by staff of Wisconsin DNR and by S. Postupalsky and W. Bowerman for the State of Michigan (M. Meyer, Wisconsin DNR; D. Best, USFWS, personal communication, March 1999) show that between 1974 and 1986 bald eagle nesting numbers on Green Bay and the eastern side of the Door Peninsula were stable at between one and two pairs (Figure 5-8). A rapid increase in nesting numbers began in 1987, and by 1997 there were 14 nesting pairs. The number of breeding pairs of eagles nesting along the Lower Fox River went from one in 1986 to three in 1994 to two since 1995. The distribution of the Green Bay and Lower Fox River nest sites is shown in Figure 5-9.

Bald eagles arrive back on their nesting territories in the assessment area in February, and the young fledge between early June and July. Depending on ice conditions, bald eagles remain in the assessment area during the winter; up to 12 have been recorded in December on the Lower Fox River (Howe et al., 1993). Thus, breeding bald eagles spend a substantial part of the year in the assessment area.

In August 1989, bald eagles were listed as threatened by the State of Wisconsin. This designation was removed in August 1997. They are currently listed as threatened by the Service.

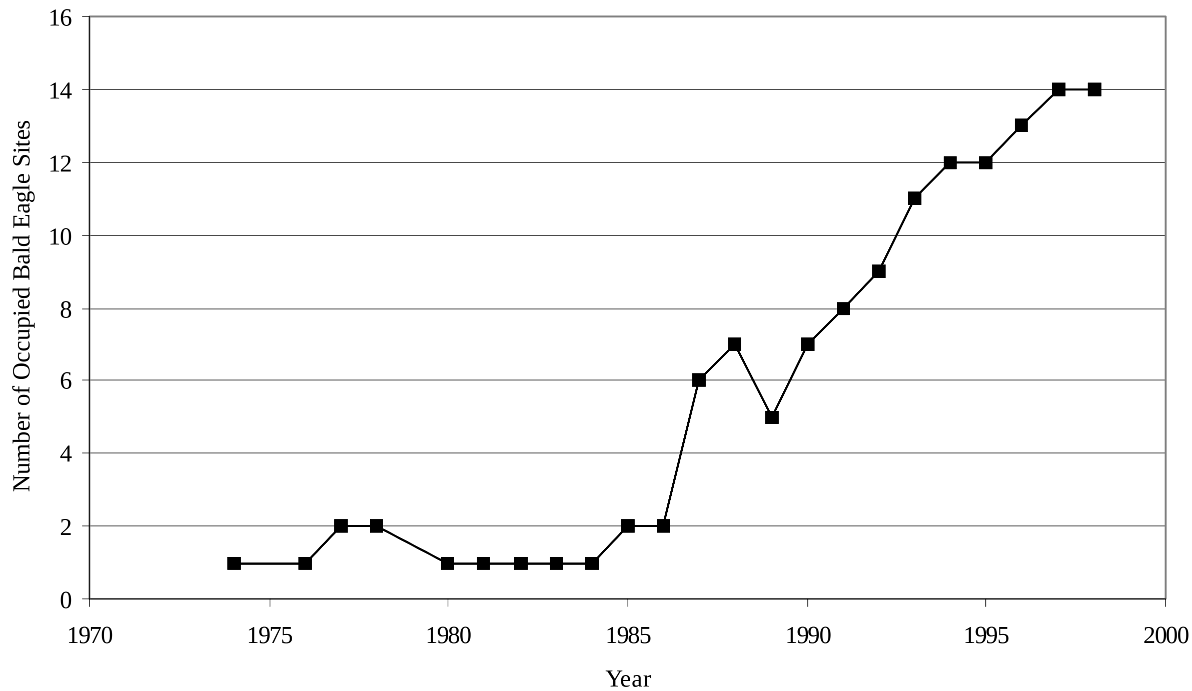


Figure 5-8. Numbers of occupied bald eagle nesting sites on Green Bay.

5.6.2 Diet and Foraging Areas around Green Bay

Diet. There are two studies that describe bald eagle diet in the assessment area. Bath (1991) quantified prey class percentages at a nest at Kaukauna on the Lower Fox River during the pre-hatching period. Dykstra and Meyer (1996) collected prey data from the entire nestling period from nests at Toft Point and Little Tail Point. The results of these studies were combined by W. Bowerman (personal communication, Lake Superior State University, April 1998) and are presented in Table 5-8. Data in Dykstra and Meyer (1996) from a nest at Blueberry Island were not used in this analysis since the nest was located along the Menominee River and might not be representative of eagles foraging around Green Bay. Also excluded were data collected by Dykstra and Meyer at Moss Lake since prey data were collected there only during the final 6 weeks of the nestling period, and prey species use changes over the nestling period (Dunstan and Harper, 1975). Prey items that were not identified in these studies were assigned identities based on the proportion of prey items that were identified by either class or species. Based on these observational and prey remains data, bald eagle prey composition on a frequency basis at Green Bay nests comprises approximately 74% fish, 23% avian prey, and 2% mammals (Table 5-8), which is similar to the diet composition of bald eagles elsewhere in the Great Lakes (Bowerman, 1993).

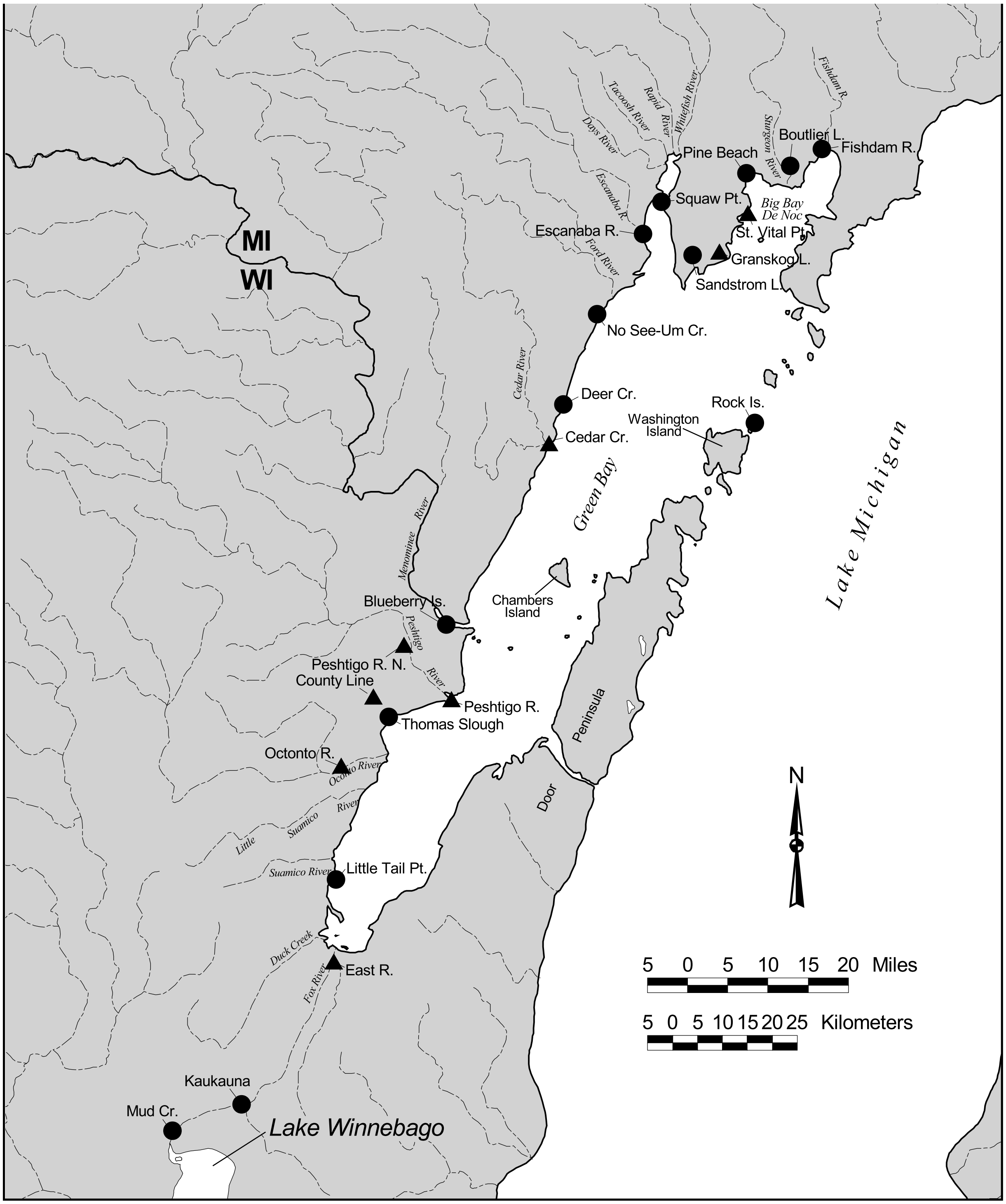


Figure 5-9. Distribution of bald eagle nest sites in Green Bay and the Lower Fox River. Circles are sites occupied in 1998. Triangles are sites not occupied in 1998 but occupied in previous years.

Table 5-8
Prey of Bald Eagles Nesting on Green Bay and the Lower Fox River Based on Prey Remains for the Breeding Period, and Observations during the Pre-Hatch Period

Class/Species	N	Percent of Total
Fish		
Sucker	23	13.8
Bullhead	30	18.0
Northern pike	28	16.8
Bass	3	1.8
Other centrarchids	6	3.6
Walleye	2	1.2
Bowfin	11	6.6
Carp	14	8.4
Freshwater drum	2	1.2
Alewife	1	0.6
Gizzard shad	4	2.4
Subtotal	**124**	**74.4**
Birds		
Herring and ring-billed gulls	15	9.0
Mergansers	2	1.2
Other ducks	4	2.4
Double-crested cormorant	1	0.6
Common raven	1	0.6
American crow	2	1.2
Unknown heron	1	0.6
Other birds	12	7.2
Subtotal	**38**	**22.8**
Mammals		
Muskrat	2	1.2
White-tailed deer	1	0.6
Red fox	1	0.6
Subtotal	**4**	**2.4**
Reptiles		
Unknown turtle	1	0.6
Subtotal	**1**	**0.6**
Total	**167**	**100.2**

Sources: Analysis of data in Bath, 1991 and Dykstra and Meyer, 1996, by W. Bowerman, Lake Superior State University, personal communication, April 1998.

Foraging areas. Observations of bald eagles nesting at Kaukauna on the Lower Fox River showed that during February through May 1991 the adults foraged along the Fox River and generally within 0.5 km of the nest, but ranged up to 3.0 km (Bath, 1991). No data exist that allow the determination of foraging ranges at Green Bay nests; however, most previous studies of bald eagle foraging assumed a radius of 8.0 km from the nest as the likely foraging area (Bowerman et al., 1995).

5.6.3 Ecological Traits that Could Affect PCB Exposure of Bald Eagles

Bald eagles nesting around Green Bay and along the Lower Fox River have a high potential for exposure to PCBs. First, they are likely to be either year-round residents in the assessment area or present for a substantial part of the year. Second, birds nesting on the Green Bay or Lower Fox River shorelines are likely to obtain much of their food from the contaminated aquatic systems, and even those birds nesting farther inland (up to about 8 km) are also likely to be dietarily exposed to assessment area contaminants. Lastly, bald eagles are tertiary predators that include high trophic level predatory birds and fish within their diet (Table 5-8). Because of these characteristics, bald eagles are potentially liable to be exposed to high levels of lipophilic compounds that bioaccumulate through trophic levels, such as PCBs.

5.6.4 Bald Eagle Exposure Pathways

The main exposure route through which bald eagles that nest on Green Bay and the Lower Fox River are exposed to PCBs is the dietary pathway. In this section, the following questions are addressed: Do the prey species that constitute the diet of the bald eagle in the assessment area have elevated PCB concentrations, and do bald eagle tissue analyses indicate that eagles are exposed to PCBs?

PCBs in Bald Eagle Prey

Many of the fish and bird species known to be eaten by bald eagles nesting in Green Bay are contaminated with PCBs (Table 5-9). Data on PCB concentrations in alewife, gizzard shad, and rainbow smelt described in Section 5.1 show that these species, also, are contaminated with PCBs in the assessment area. These data show that bald eagles in the assessment area are exposed to PCBs in their diets.

PCBs in Bald Eagle Tissues

Table 5-10 shows the total PCB concentrations in bald eagle eggs from nests around Green Bay from 1986 (when the earliest sample was collected) until 1997 (data from Wisconsin DNR and USFWS contaminants databases provided by M. Meyer, Wisconsin DNR, and D. Best, USFWS).

Table 5-9
PCB Concentrations in Potential Bald Eagle Prey from the Assessment Area

Prey Species	Date	Locality	Tissue	Sample Size	PCB Concentrations (mg/kg wet weight) Ranges (where known) in Parentheses	Reference
Mallard	1985-1986	Lower Fox River	Muscle, skin, and fat	55	0.4[a] (0-1.5)	Amundson, undated report
Double-crested cormorant	1987-1988	Green Bay	Whole body	6	84.8[a]	USFWS, 1993
Sucker	1979	Green Bay	Whole body	4	2.6[a] (1.7-4.4)	Wisconsin DNR, 1971-1995
Bullhead	1979	Green Bay	Whole body	1	2.1	Wisconsin DNR, 1971-1995
Northern pike	1979	Green Bay	Whole body	1	10.5	Wisconsin DNR, 1971-1995
Carp	1979-1989	Green Bay	Whole body	116	4.0[a] (0.04-10.5)	Wisconsin DNR, 1971-1995; Connolly et al., 1992

a. Mean of measurements.

These data show that bald eagles nesting in the assessment area have been exposed to PCBs. DDE concentrations in bald eagles are also shown in Table 5-10 and will be discussed in Section 5.6.6.

Figure 5-10 compares the Green Bay 1986-1997 egg PCB and DDE concentrations with concentrations in eggs from inland Michigan and inland Wisconsin. PCB and DDE concentrations are significantly higher in the Green Bay eggs than in eggs from nests in inland Michigan ($t = 5.9$, $p < 0.001$, and $t = 4.9$, $p < 0.001$, respectively) and Wisconsin ($t = 6.12$, $p < 0.001$, and $t = 4.4$, $p < 0.001$, respectively).

Table 5-11 shows the total PCB concentrations in bald eagle nestling blood plasma from the assessment area from 1987 to 1995 (Dykstra and Meyer, 1996) and from inland Michigan. Although no statistical tests were carried out by Dykstra and Meyer, the plasma levels in assessment area chicks exceed those in chicks from inland Michigan.

The above data confirm that assessment area bald eagles are likely to forage in areas that contain contaminated fish and wildlife, and that their prey has highly elevated PCB concentrations. They

Table 5-10
PCB and DDE Concentrations (mg/kg fresh wet weight) in Bald Eagle Eggs from Green Bay and the Lower Fox River

Breeding Area/State/Number	Year	Total PCBs	DDE
Green Bay			
Peshtigo River/WI/MT-07	1987	13.0	2.4
Boutlier Lake/MI/DE-15	1986	55.3[a]	30.2[a]
Fishdam River/MI/DE-17	1990	26.6[a]	10.1[a]
Fishdam River/MI/DE-17	1991	27.2	7.4
Peshtigo River/WI/MT-07	1991	56.5[a]	12.0[a]
Peshtigo River/WI/MT-16	1992	66.6[a]	14.7[a]
Peshtigo River/WI	1995	120.0	21.0
Fishdam River/MI/DE-17	1992	28.5[a]	11[a]
Squaw Point/MI/DE-18	1992	28.7	12.3
Squaw Point/MI/DE-18	1993	42.3[b]	12.9[b]
Moss Lake/MI/DE-09	1994	24.3	4.3
St. Vital's Point/MI/DE-20	1997	22.4	8.3
Oconto/WI	1997	88.0	16.0
Fox River			
Kaukauna Lower Fox River/OU-1	1990	36.0	1.1
Mean of Green Bay eggs[c]	n = 13	46.1	12.5

a. Mean of two eggs.
b. Mean of three eggs.
c. Multiple eggs from the same breeding area in a given year averaged prior to determining mean.

Sources: Dykstra and Meyer, 1996; Wisconsin and USFWS contaminants monitoring databases.

also show that Green Bay bald eagle eggs and plasma are contaminated with PCBs. Furthermore, the PCB concentrations in Green Bay bald eagle tissues significantly exceed those from inland control populations.

The contaminant concentrations in the Fox River pair of bald eagles are less clearly characterized. Only one egg has been analyzed; however, egg and nestling plasma data indicate that the Fox River birds are exposed to elevated concentrations of PCBs.

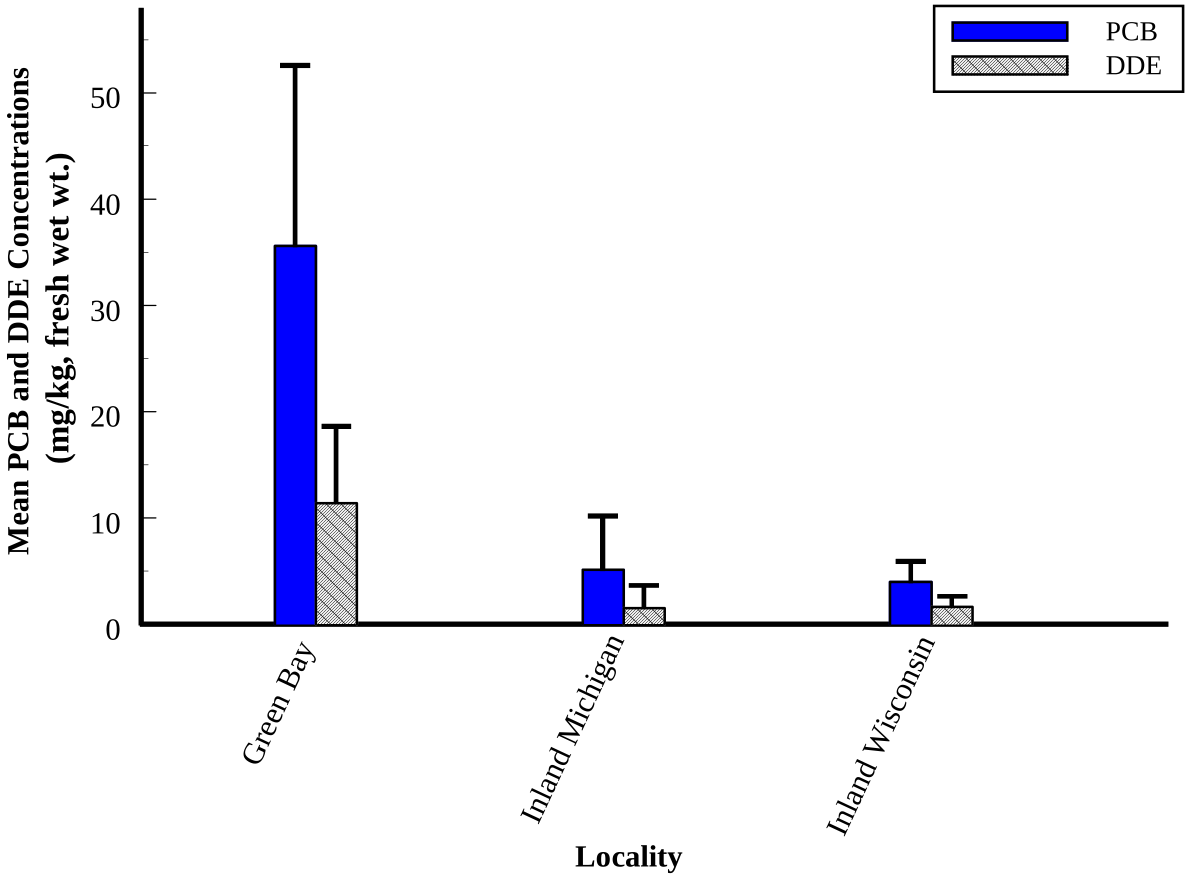


Figure 5-10. Mean PCB and DDE concentrations in bald eagle eggs from Green Bay, inland Michigan, and inland Wisconsin. Vertical lines represent one standard deviation.

5.6.5 Injuries to Assessment Area Bald Eagles

This section evaluates current evidence that assessment area bald eagles have been injured, focusing on reproductive malfunctions.[1] We then present an analysis of causality in which the main question addressed is whether observed injuries have been caused by exposure to PCBs.

Malfunctions in Green Bay Bald Eagle Reproduction

Figure 5-11 shows productivity histories of individual nests of bald eagles nesting in inland Michigan, inland Wisconsin, and Green Bay. These data show that there is much variability in

---

[1] Bowerman et al. (1994b) reported six instances of bill deformities among Great Lakes bald eagle nestlings. No such abnormalities have been reported among assessment area birds. As a result, this effect is not considered further for bald eagles.

Table 5-11
PCB Concentrations (μg/kg wet weight) in Plasma of Nestling Bald Eagles from Green Bay and the Lower Fox River

Breeding Area/State/Number	Year	Total PCBs[a]
Green Bay		
Granskog Lake/MI/DE-13	1987	229
Boutlier Lake/MI/DE-15	1987	319
Peshtigo River N/WI/MT-16	1992	901
Toft Point/WI/DO-01	1994	121
Oconto River-Thome/WI/OC-04	1994	393
Toft Point/WI/DO-01	1995	150
Blueberry Island	1994	83
Blueberry Island	1995	87
Fox River		
Kaukauna/WI/OU-01	1991	120
Kaukauna/WI/OU-01	1992	318
Kaukauna/WI/OU-01	1993	226
Kaukauna/WI/OU-01	1994	547
Kaukauna/WI/OU-01	1995	290
Mean Green Bay	n = 8	285.4
Mean Fox River	n = 5	300.2
Mean Inland Michigan[b]	n = 79	24

a. Data from Dykstra and Meyer, 1996.
b. Bowerman et al., 1994a.

inter-year productivity at individual nests. They also show, however, that the pattern for Green Bay nests is different from that in the two inland areas in that the Green Bay nests fail to produce young on a more consistent basis.

Figure 5-12 shows the mean annual productivity (number of large young produced) of Green Bay bald eagles compared with that of birds nesting in inland Michigan and inland Wisconsin between 1974 and 1998 (data provided by M. Meyer of Wisconsin DNR and D. Best of the USFWS). Mean annual productivity among inland Michigan and Wisconsin birds has consistently approximated or exceeded 1.0 young/nest, the productivity rate needed to maintain a healthy population (Kubiak and Best, 1991). Green Bay eagles had zero productivity during the period from 1974 until 1979. Green Bay nest productivity averaged at least 1.0 young per nest from 1980 to 1982 and from 1985 through 1987. After each of these three-year periods, productivity declined dramatically, reaching 0.0 within 1 or 2 years. However, during these periods there was only one or two pairs of eagles nesting in the assessment area (Figure 5-8). Productivity among Green Bay bald eagles has been at or near 1.0 young/year for 1995 through 1998. The

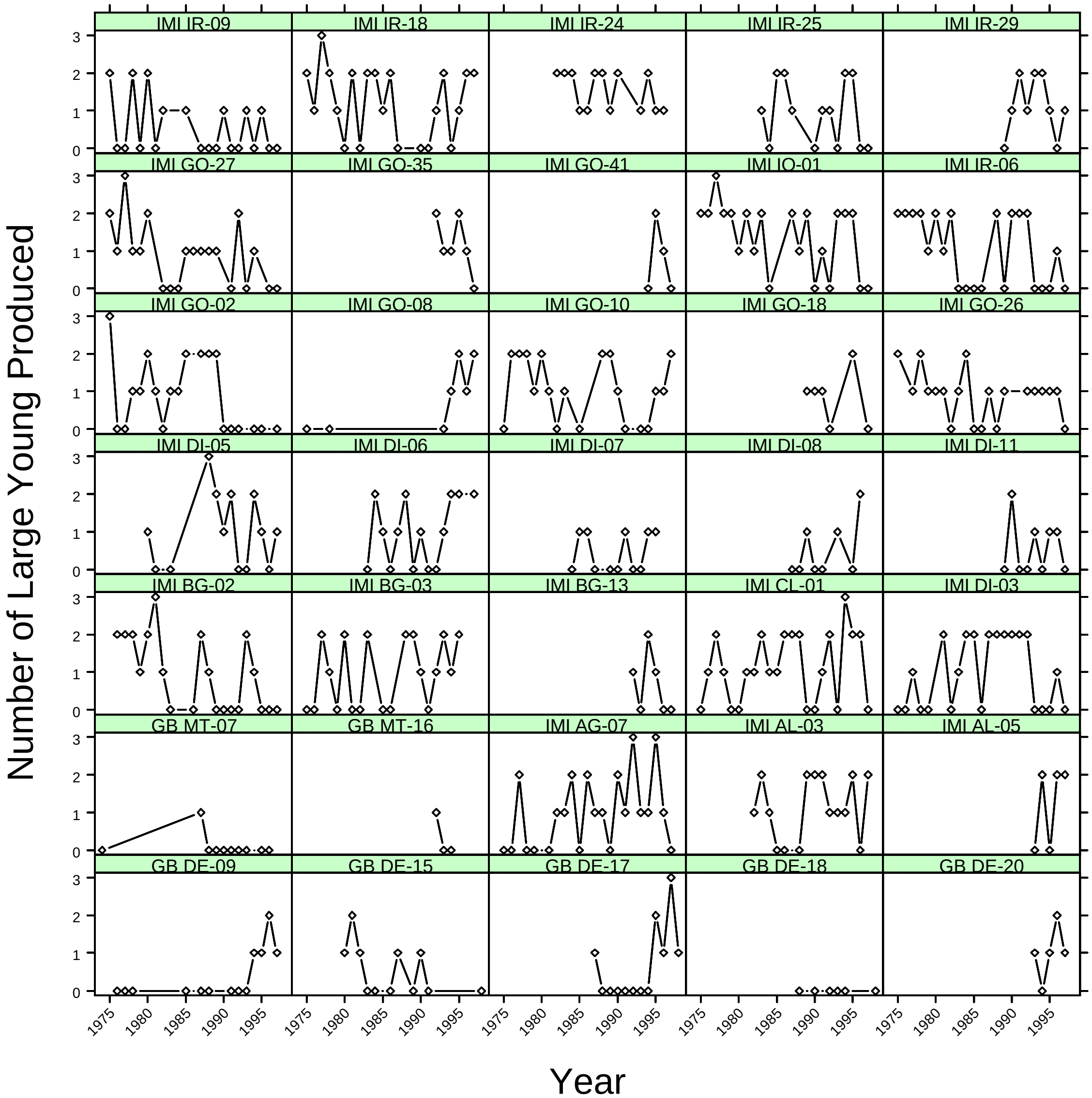


Figure 5-11. Productivity histories of individual bald eagle nests in inland Michigan (IMI), inland Wisconsin (IWI), and Green Bay (GB). Only nests for which there are both egg contaminants and productivity data are shown.

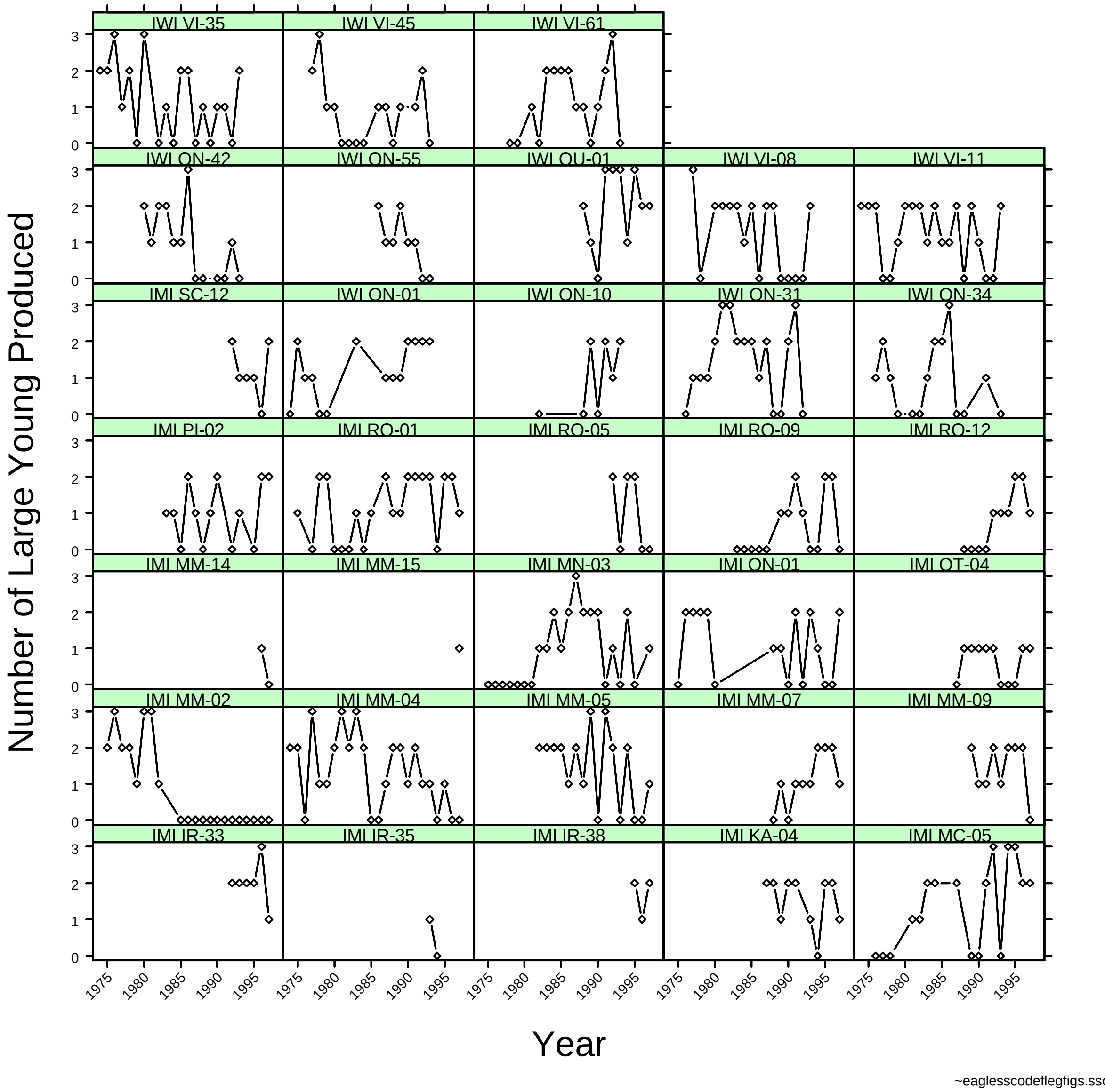


Figure 5-11 (cont.). Productivity histories of individual bald eagle nests in inland Michigan (IMI), inland Wisconsin (IWI), and Green Bay (GB). Only nests for which there are both egg contaminants and productivity data are shown.

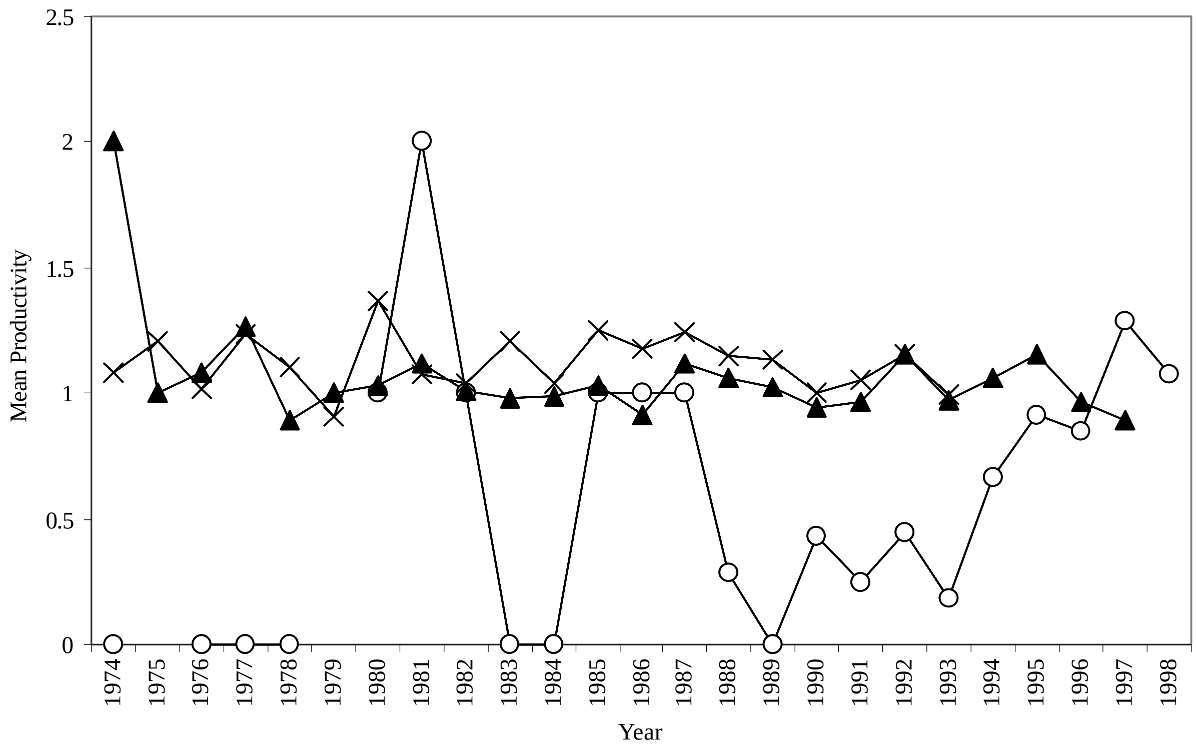


Figure 5-12. Mean annual productivity of bald eagles nesting on Green Bay (open circles), inland Michigan (triangles), and inland Wisconsin (crosses).

overall productivity rate of Green Bay bald eagles from 1974 through 1998 is significantly lower than for bald eagles in inland Wisconsin ($\chi^2 = 29.5$, 1 df, $p < 0.001$) and inland Michigan ($\chi^2 = 22.9$, 1 df, $p < 0.001$).

Table 5-12 presents the results of an analysis of the proportions of nests in Green Bay, inland Michigan, and inland Wisconsin that produced no, one, two, or three chicks during the period from 1974 to 1988. These data show that a higher proportion of bald eagle nest attempts in Green Bay resulted in no chicks being reared (0.54) than in either of the inland areas (0.36 and 0.34). Conversely, more inland nesting attempts resulted in one or more chicks being reared (0.63 and 0.66) than in Green Bay (0.46). These data support the conclusions of our previous mean productivity analysis by confirming that productivity is reduced in the assessment area.

Table 5-13 shows that the productivity of bald eagles nesting on the Fox River during the period from 1988 to 1998 was higher than in Green Bay. From 1988 to 1994 (when productivity among Green Bay eagles was low), the Fox River nests produced an average of 1.7 young/active nest. Since 1995, this productivity has been 2.4 young/active nest. The contaminants data from these sites suggest that the ratio of PCB to DDE in eggs and plasma may also be different from that for

Table 5-12
Proportions of Breeding Outcomes (0, 1, 2, or 3 chicks reared) among Green Bay, Inland Wisconsin, and Inland Michigan Bald Eagles

Area	Number of Nests and (nest years)	0 Chicks	1 Chick	2 Chicks	3 Chicks
Green Bay	23 (137)	0.54[a] (0.53)[b]	0.25 (0.20)	0.18 (0.24)	0.03 (0.03)
Inland Wisconsin	172 (1700)	0.34 (0.37)	0.25 (0.25)	0.36 (0.34)	0.05 (0.04)
Inland Michigan	251 (2664)	0.36 (0.37)	0.29 (0.31)	0.31 (0.30)	0.03 (0.02)

a. Proportions calculated for all nest/years in region without distinguishing between nests.
b. Average proportions calculated for each nest then combined in regional averages.

Table 5-13
Productivity (large young raised per active nest) of Fox River Bald Eagles from 1988 to 1998

Nest Name	88	89	90	91	92	93	94	95	96	97	98
Kaukauna, WI	2	1	0	3	3	3	1	3	2	2	3
Mud Creek, WI							2	3	1	2	3
East River, WI							0				
Productivity Summary, All Nests											
Number of active nests	1	1	1	1	1	1	3	2	2	2	2
Number of young reared	2	1	0	3	3	3	3	6	3	4	6
Young/active nest	2	1	0	3	3	3	1	3	1.5	2	3

Note: a blank cell indicates that the nesting territory was unoccupied in that year.

Source: USFWS and Wisconsin DNR bald eagle productivity databases.

the Green Bay eagles (Tables 5-10 and 5-11). However, the relatively few data that are available also suggest that the toxicity of the PCBs measured in the Fox River egg may be less than that measured in Green Bay eggs. Using the H4IIE bioassay method, two eggs from a Peshtigo Marsh nest in 1988 averaged 147.5 pg/g TCDD-EQ, while one egg from the Kaukauna nest on the Fox River in 1990 had only 34 pg/g TCDD-EQ (D. Tillitt, USFWS, unpublished data). Thus, although the Fox River eagles may have total PCB concentrations in their eggs that are similar to those in the eggs of Green Bay birds, their toxicity may be less.

Also, bald eagles nesting on Green Bay may have less opportunity to forage in uncontaminated areas than the Kaukauna and Mud River birds, which are close to Lake Winnebago (Figure 5-9) where uncontaminated prey can be obtained. This complicates the analysis of what may be causing the increased productivity of the Fox River birds. Overall, given the small sample sizes that are available for the Fox River birds, the reason that they have higher productivity than Green Bay birds is uncertain.

The data presented above confirm that, like other Great Lakes coastal populations of bald eagles (e.g., Kubiak and Best, 1991; Best et al., 1994; Bowerman et al., 1995), eagles nesting around the Green Bay coastline have suffered decreased reproductive rates. The reduced productivity in the assessment area began in 1974, when the area was first recolonized, and continued up until at least the mid-1990s.

5.6.6 Green Bay Bald Eagle PCB and DDE Tissue Residues and Toxicity Thresholds

Kubiak and Best (1991), Wiemeyer et al. (1993), and Nisbet and Risebrough (1994) have used relationships between geospatial differences in PCB and DDE concentrations and productivity to postulate toxicity thresholds for each contaminant. The results are shown in Table 5-14. From these data, egg toxicity thresholds (concentrations at which adverse impacts on productivity become likely) may be >3.0 mg/kg wet weight for PCBs and >3.6 mg/kg wet weight for DDE. Major impacts on productivity (reductions of 50% or greater) are suggested at PCB and DDE concentrations of 13-23 mg/kg wet weight and 3.6-6.3 mg/kg wet weight, respectively.

Studies of the closely related white-tailed sea eagle in Scandinavia have also attempted to determine the contributions of PCBs and DDE to reduced hatching success (Helander et al., 1982; Helander et al., 1998; Olsson et al., 1998). These studies have not been entirely successful in determining contributions (because of the high correlation between the two contaminants in eggs). However, Olsson et al. (1998) suggested a total PCB embryo mortality LOEL of 300 mg/kg wet weight. The relevance of these studies to bald eagles is not yet clear.

All 13 of the Green Bay bald eagle eggs analyzed (Table 5-10) either equaled or exceeded 13 mg/kg wet weight PCBs, and 12 of these eggs are within or exceed the 3.6-6.3 mg/kg wet weight DDE range. Based on the thresholds in Table 5-14, the PCB concentrations in all of the Green Bay eggs are sufficient to result in major reproductive failure. The same is true for DDE for most of the eggs. Thus, based on the above thresholds, both PCBs and DDE could have been responsible for the reduced productivity observed in Green Bay bald eagles before the mid-1990s.

Table 5-14
Bald Eagle Egg Toxicity Levels Identified from Comparisons of Regional Productivities and Contaminant Concentrations

Productivity Response	Egg PCB Toxic Level (mg/kg fresh wet weight)	Egg DDE Toxic Level (mg/kg fresh wet weight)	Reference
"Normal" productivity	<3.0	<3.6	Wiemeyer et al., 1993
10% productivity reduction	3.0-5.6		Wiemeyer et al., 1993
30% productivity reduction	5.6-13.0		Wiemeyer et al., 1993
50% productivity reduction	13-23	3.6-6.3	Wiemeyer et al., 1993
70% productivity reduction	>23		Wiemeyer et al., 1993
75% productivity reduction		>6.3	Wiemeyer et al., 1993
"Healthy" reproduction	<1.7	<6.0	Kubiak and Best, 1991
No productivity reduction		<2.5	Nisbet and Risebrough, 1994
Productivity approximately halved		>5.0	Nisbet and Risebrough, 1994

To investigate potential relationships between productivity and PCBs in bald eagle eggs, the productivity data in Figure 5-11 were converted to probabilities that bald eagles in the assessment area and in the two inland reference areas will raise either no young or one or more young, and were assessed in relation to egg PCB concentrations. Productivity observations for individual nest years were omitted if they were separated by more than two years from years in which PCB concentration data were available for that nest. In cases where multiple PCB records were available for the same nest, but were separated by more than four years, independent productivity probabilities were calculated for the two or more periods. The series of productivity records that were retained by this procedure were used to calculate the relative frequency of producing a particular number of chicks, which was used to represent probabilities. These probabilities are presented in relation to the PCB concentrations measured in eggs from those nests (Figure 5-13). Figure 5-13 shows that the probability that bald eagle nests will rear no young rises steeply after egg PCB concentrations exceed 20 mg/kg fresh wet weight. Conversely, the probability that birds will raise one or more young falls after that concentration. All but one of the Green Bay bald eagle eggs that have been analyzed (Table 5-10) had PCB concentrations that exceed this threshold. This indicates that, based on the 20 mg/kg threshold, PCB concentrations in Green Bay bald eagle eggs are sufficient to result in reduced productivity.

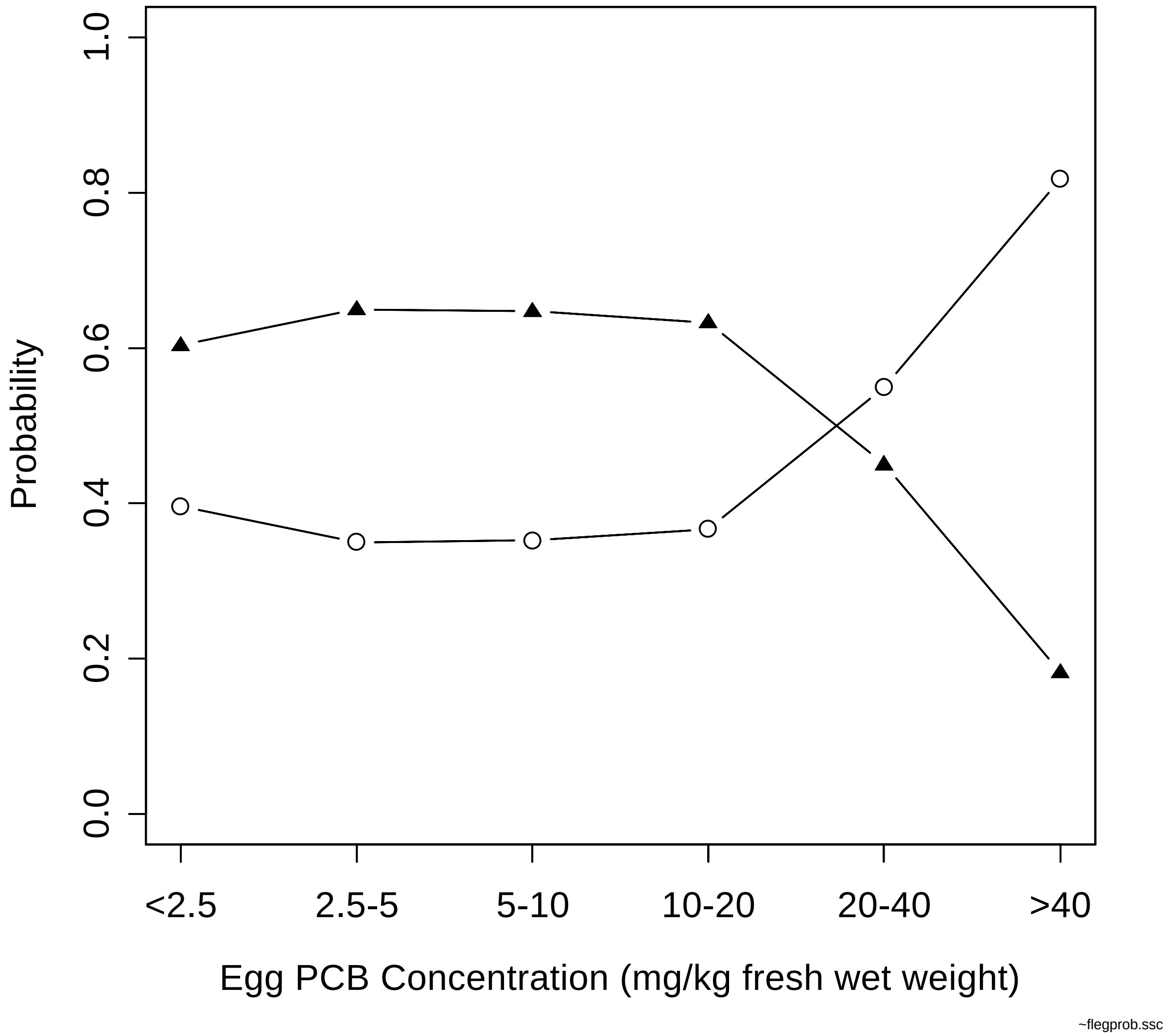


Figure 5-13. Probability of bald eagles in inland Michigan and Wisconsin and Green Bay producing no young (open circles) or one or more young (triangles) in relation to egg PCB concentrations.

5.6.7 PCBs and DDE and Reduced Reproductive Success among Green Bay Bald Eagles

The data presented in Sections 5.6.4 through 5.6.6 show that Green Bay bald eagles have elevated egg and plasma PCB and DDE concentrations. They also show that Green Bay bald eagles, during the period from 1987 until the mid-1990s, had significantly lower reproductive success than inland Wisconsin or Michigan birds and that, based on toxicity thresholds, the reduced reproduction could be attributable to the elevated PCB and DDE concentrations.

Previous studies of Great Lakes bald eagles [Kubiak and Best (1991), Bowerman (1993), Bowerman et al. (1994a), and Bowerman et al. (1995)] found that productivity among Great Lakes bald eagles was negatively correlated with both PCB and DDE concentrations in eggs and attributed the reduced reproductive success to these contaminants. Dykstra and Meyer (1996) evaluated the causes of the low productivity in Green Bay bald eagles and found that the low productivity was not attributable to either food availability (indices of food availability were similar to inland Wisconsin nests) or disturbance (adult attendance patterns at the nests were also similar to inland birds). Dykstra and Meyer (1996) concluded that the reduced productivity among Green Bay bald eagles was caused by PCBs and/or DDE. PCB and DDE concentrations are typically correlated in bald eagle eggs [Wiemeyer et al. (1993): $r = 0.76$; analysis of data in Dykstra and Meyer (1996): $r = 0.65$; analysis of mean PCB and DDE concentrations in bald eagle eggs from seven Great Lakes regions in Kubiak and Best (1991): $r = 0.9$; Clark et al. (1998): $r = 0.91$]. Figure 5-14 shows the relationship between PCB and DDE concentrations in bald eagle eggs from Green Bay. These data are from the USFWS and Wisconsin DNR contaminants monitoring databases. PCBs are positively correlated with DDE ($r = 0.67$, $p < 0.05$). Figure 5-15 shows a similar analysis but using all of the egg concentration data from Green Bay, inland Michigan, and inland Wisconsin. PCBs are again significantly correlated with DDE ($r = 0.8$, $p < 0.001$). This correlation between contaminants has proven to be a difficulty in previous attempts to quantify their relative contributions to reduced productivity in bald eagles (Wiemeyer et al., 1993; Dykstra and Meyer, 1996).

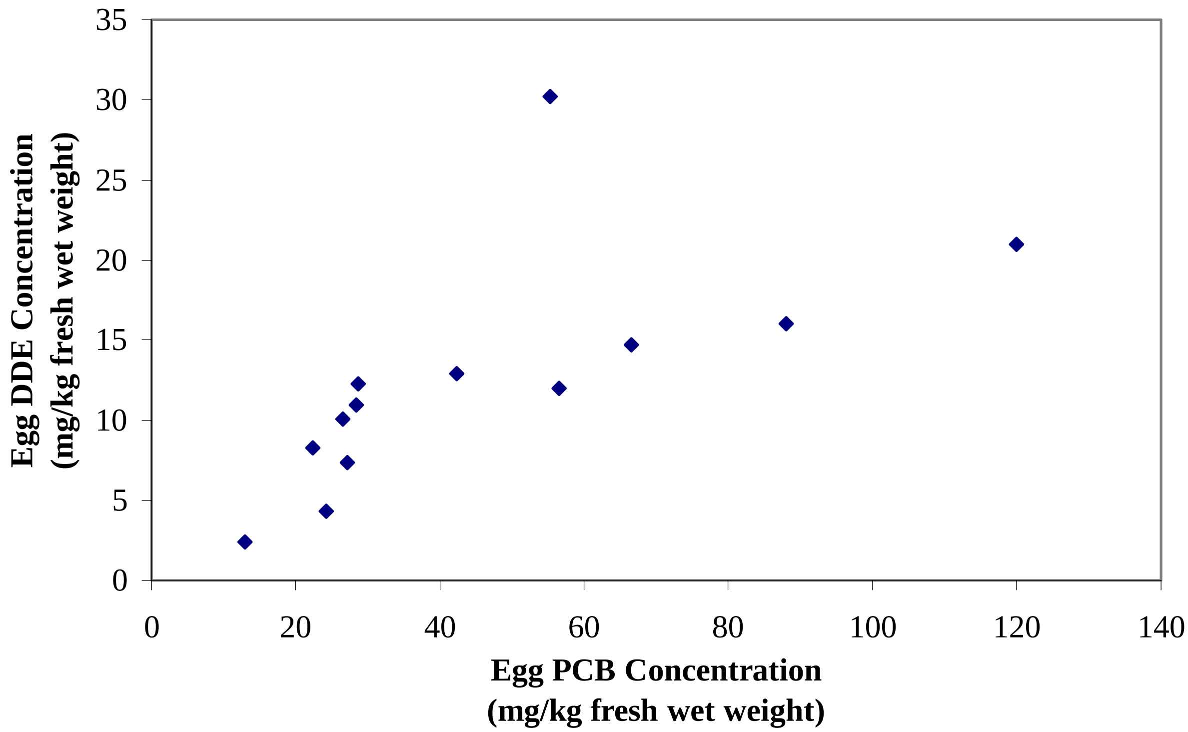


Figure 5-14. Relationship between PCB and DDE concentrations in Green Bay bald eagle eggs.

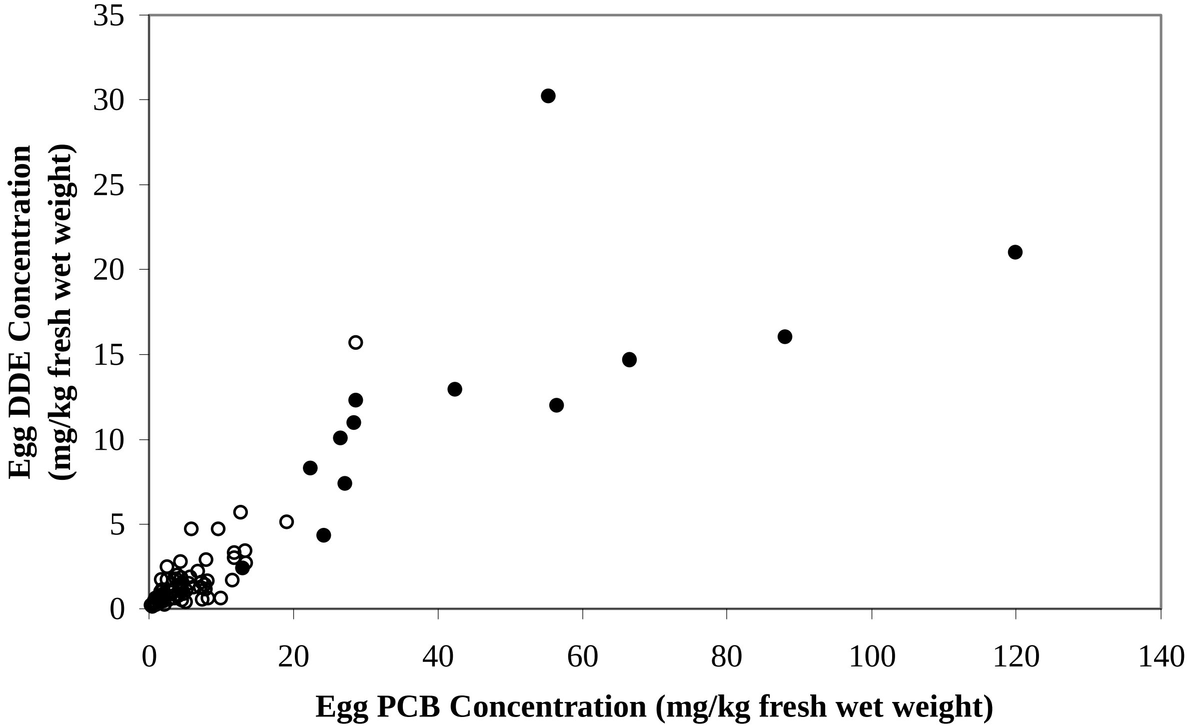


Figure 5-15. Relationship between PCB and DDE concentrations in bald eagle eggs from Green Bay (solid circles) and inland Michigan and Wisconsin (open circles).

Figures 5-16 through 5-19 show the relationships between PCB and DDE concentrations in eagle eggs from Green Bay, inland Wisconsin, and inland Michigan and two measures of productivity: the mean number of young reared at the site from which an egg was taken for chemistry analysis during the year of egg collection and the year preceding and subsequent to that event (3-year productivity), and the mean number of young reared at the site from which an egg was taken for chemistry analysis during the year of egg collection and the two years preceding and subsequent to that event (5 year productivity). These chemistry and productivity data were obtained from the USFWS and Wisconsin DNR monitoring data sets supplied by D. Best (USFWS) and M. Meyer (Wisconsin DNR).

Figures 5-16 through 5-19 show negative relationships between both PCB and DDE egg concentrations and productivity. These negative correlations are statistically significant for PCBs and 3 year productivity ($r = -0.4$, $p < 0.001$), DDE and 3 year productivity ($r = -0.36$, $p < 0.01$), PCBs and 5 year productivity ($r = -0.4$, $p < 0.001$), and DDE and 5 year productivity ($r = -0.3$, $p < 0.001$). Productivity in reference areas normally averages about 1.1 young/nest (Figure 5-11); thus, Figures 5-16 through 5-19 show that increases in egg PCB and DDE concentrations are associated with markedly reduced productivity. In contrast, in a recent study, Donaldson et al. (1999) found no significant relationships between productivity and either PCBs or DDE in eggs or

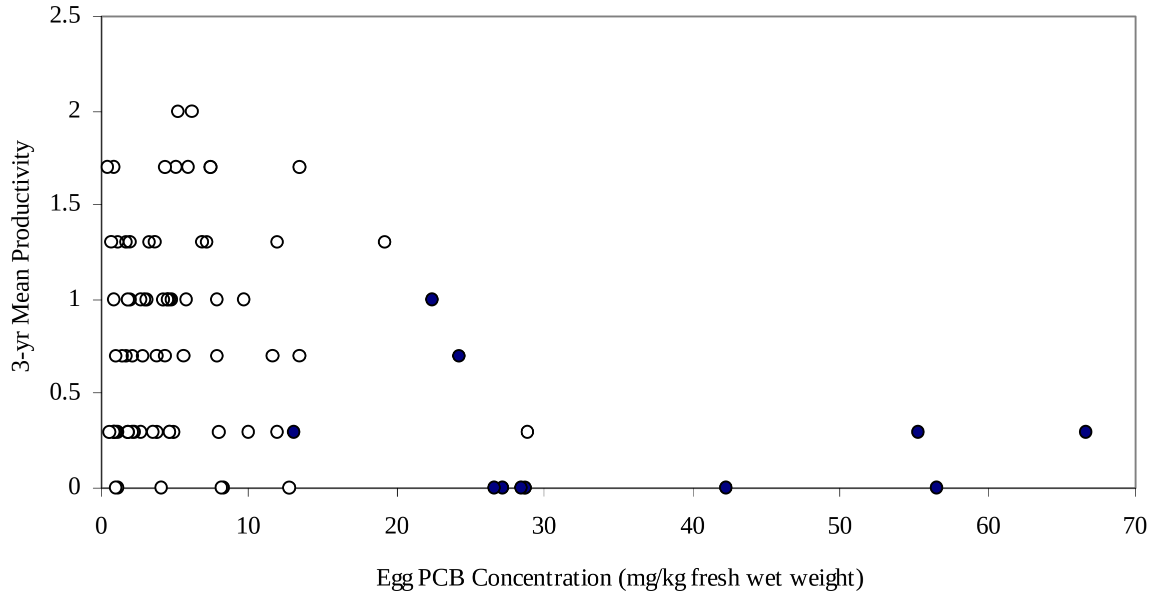


Figure 5-16. Egg PCB concentrations and 3-year mean productivity at bald eagle nests on Green Bay (solid circles) and in inland Michigan and Wisconsin (open circles).

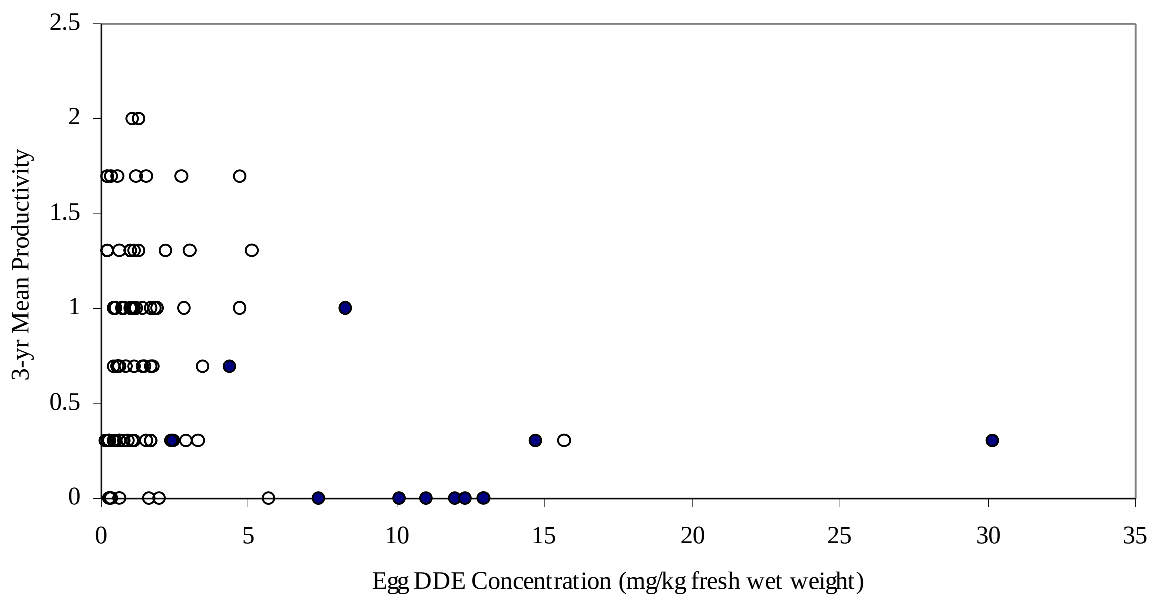


Figure 5-17. Egg DDE concentrations and mean 3-year productivity at bald eagle nests in Green Bay (solid circles) and in inland Michigan and Wisconsin (open circles).

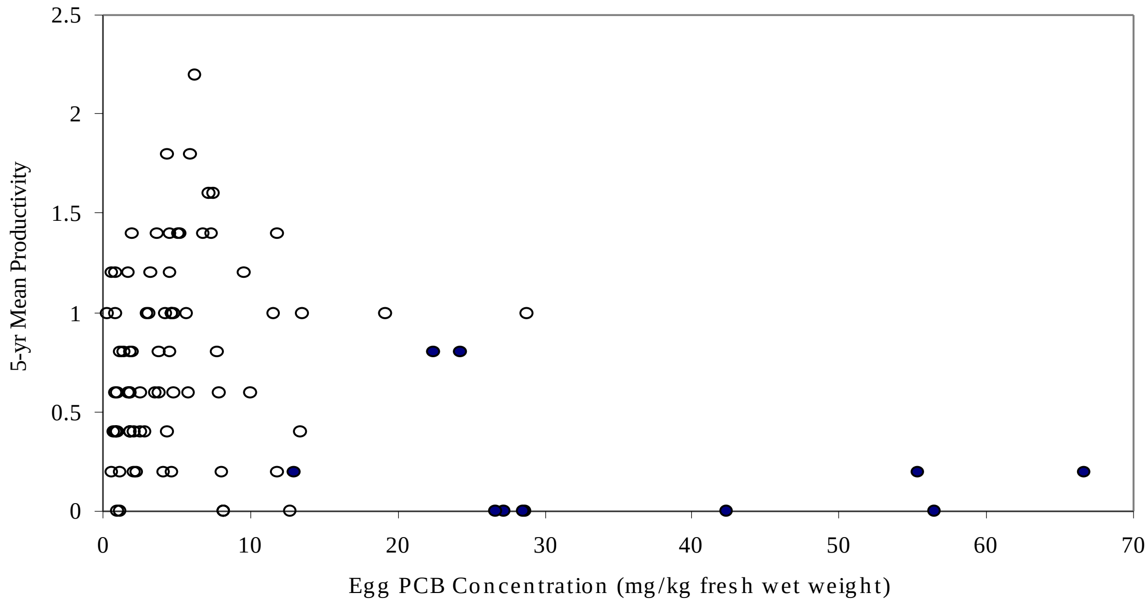


Figure 5-18. Egg PCB concentrations and mean 5-year productivity at bald eagle nests on Green Bay (solid circles) and in inland Michigan and Wisconsin (open circles).

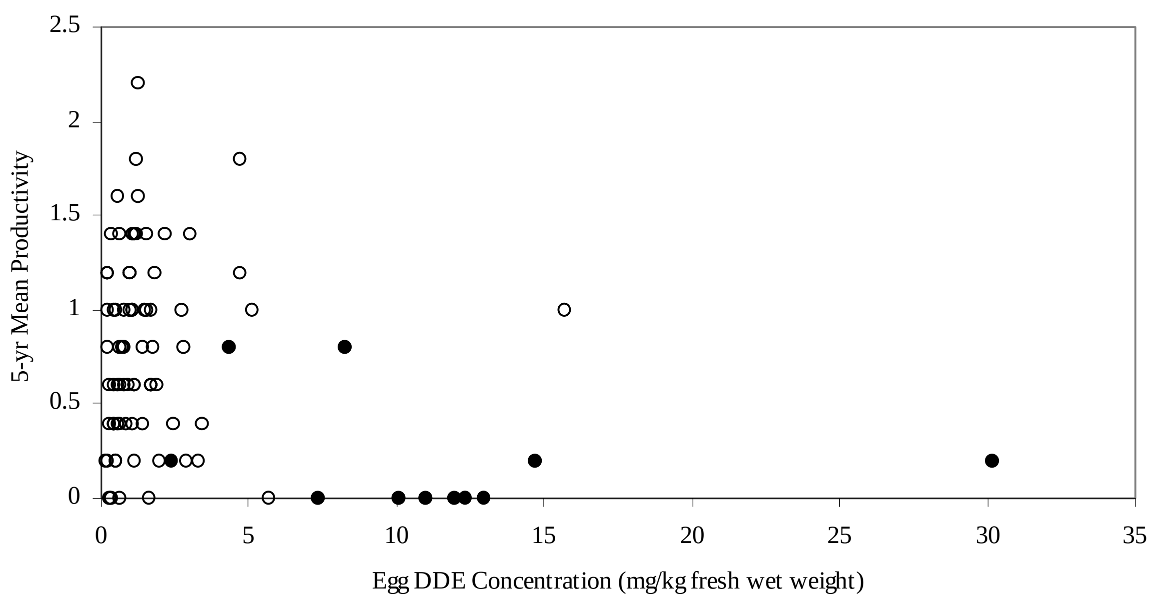


Figure 5-19. Egg DDE concentrations and mean 5-year productivity at bald eagle nests on Green Bay (solid circles) and in inland Michigan and Wisconsin (open circles).

nestling plasma from the Canadian Great Lakes. The reasons for the differences between the results of our analysis and those of Donaldson et al. are unclear.

However, the data reported in Donaldson et al. show that the period over which productivity was measured (1980-1996) was largely subsequent to a period in which PCB and DDE concentrations in Lake Erie bald eagle eggs had undergone substantial declines (1974-the early to mid 1980s). Thus, the productivities of the nests were measured after contaminants had declined (by approximately factors of 4).

5.6.8 The Relative Contributions of PCBs and DDE to Reduced Reproductive Success in Green Bay Bald Eagles

In this section we evaluate the relative contributions of PCBs and DDE to the reduced reproductive success among Green Bay bald eagles. We concentrate on PCBs and DDE because: these are the only contaminants that have been found in Great Lakes bald eagle tissues in high enough concentrations to result in adverse effects (Bowerman et al., 1995); they are the contaminants that have been most closely correlated with bald eagle reproductive success in the Great Lakes and elsewhere (Wiemeyer et al., 1984; Kubiak and Best 1991; Nisbet and Risebrough, 1994; Bowerman et al., 1995; Clark et al., 1998); and they are known to result in the types of adverse effect (embryo mortality and reduced reproductive success) observed in assessment area bald eagles.

To evaluate whether PCB effects on productivity in Green Bay can be differentiated from those of DDE, we performed partial correlation analyses. In these analyses, we partialled out DDE [making the conservative assumption that DDE is having a significant effect on productivity] to evaluate whether PCBs explain a significant amount of the residual variation. The results of these tests (Pearson and Spearman) are shown in Table 5-15. Egg PCB concentrations did not explain a significant amount of the residual variation.

Table 5-15
Partial Correlation Coefficients Obtained in Pearson and Spearman Analyses of Egg PCB Concentrations and Productivity

Test	3-Yr Productivity	5-Yr Productivity
Pearson	-0.01 (0.91)	0.00 (0.93)
Spearman	0.03 (0.77)	0.05 (0.66)

Note that p values are given in parentheses.

In a complementary analysis, we then partialled out the effects of PCBs to evaluate whether DDE explains a significant amount of the residual variation. The results of these tests (Pearson and Spearman) are shown in Table 5-16. Egg DDE concentrations did not explain a significant amount of the residual variation.

Table 5-16
Partial Correlation Coefficients Obtained in Pearson and Spearman Analyses of Egg DDE Concentrations and Productivity

Test	3-Yr Productivity	5-Yr Productivity
Pearson	-0.15 (0.17)	-0.16 (0.15)
Spearman	-0.13 (0.24)	-0.11 (0.32)
Note that p values are given in parentheses.		

The results of the analyses described above are not sufficient to allow us to determine the relative contributions of PCBs and DDE to reductions in productivity in Green Bay bald eagles. The exceedences of the thresholds developed by Nisbet (1989), Kubiak and Best (1991), Wiemeyer et al. (1993), Nisbet and Risebrough (1994), and Stratus Consulting (Section 5.6.6), and the correlations shown above, suggest that both contaminants may be affecting productivity and that separating their effects, given the degree of correlation, is not feasible.

5.6.9 Summary

The data and analyses on bald eagles described in this section show the following:

- Green Bay bald eagles have been exposed to PCBs through their diet.
- PCB concentrations in bald eagle eggs and chick plasma in Green Bay are significantly higher than those in reference areas.
- Productivity among Green Bay bald eagles was significantly reduced relative to reference area eagles from 1974 until at least the mid-1990s.
- Exceedences of the Kubiak and Best (1991), Wiemeyer et al. (1993), and Nisbet and Risebrough (1994) thresholds and thresholds developed by Stratus Consulting, together with the negative correlations between PCB and DDE egg concentrations and productivity, indicate that PCBs and/or DDE have contributed to the reduced productivity of Green Bay bald eagles.

- Given the limitations of the chemistry and productivity data sets and of correlation among contaminants, it is not possible to determine the relative contributions of PCB and DDE to the reduced productivity of Green Bay bald eagles.

CHAPTER 6
INJURIES TO WATERFOWL: CONSUMPTION ADVISORIES

Previous chapters discussed toxicological injuries to birds caused by PCBs. In this chapter we evaluate injuries to waterfowl (ducks and geese) in Green Bay associated with PCB accumulation in bird tissue in excess of federal or state action, tolerance, or consumption advisory levels. In addition to the toxicological injuries described in previous chapters, the Departmental NRDA regulations specify that injury has occurred when concentrations of hazardous substances are sufficient to "exceed action or tolerance levels established under section 402 of the Food, Drug and Cosmetic Act" [43 CFR § 11.62(f)(1)(ii)] or "exceed levels for which an appropriate State health agency has issued directives to limit or ban consumption" [43 CFR § 11.62(f)(1)(iii)].

6.1 STATUS AND ECOLOGY OF WATERFOWL

Waterfowl are both breeding summer residents and passage migrants in the assessment area (see Chapter 2). During the summer months, surface feeding ducks and geese such as mallard, teal, gadwall, and Canada geese nest in the marshes adjacent to Green Bay, whereas red-breasted mergansers nest on many of the islands that are adjacent to the Door Peninsula (White and Cromartie, 1977; Heinz et al., 1983). In the fall, the breeding populations are augmented by large numbers of migratory ducks and geese (Robbins, 1991). These migrants, including scaup, bufflehead, goldeneye, redheads, and canvasbacks, feed in the bay until they are forced by the onset of winter to move to more southerly wintering areas. During the fall influx, the waterfowl in Green Bay and its surrounding wetlands are intensively hunted and comprise an important recreational resource (K. Stromborg, U.S. Fish and Wildlife Service, personal communication, 1998).

6.2 PATHWAY AND EXPOSURE ANALYSIS

No data have been reported on the diets of waterfowl species in Green Bay. However, based on what is known about the diets of waterfowl in general (Ehrlich et al., 1988), many of the species that inhabit the bay (e.g., mallard, teal, gadwall) are primarily herbivorous, consuming aquatic and marsh vegetation. Others (e.g., goldeneye, canvasbacks, and buffleheads) are likely to consume mainly benthivorous organisms such as molluscs, while mergansers are mainly piscivores and prey on small forage fish (Ehrlich et al., 1988).

PCB contamination in assessment area forage fish is described in Section 5.1.2. These data demonstrated that forage fish contain elevated concentrations of PCBs and serve as a component of the dietary pathway to higher trophic levels.

No data have been reported on PCBs in the diets of herbivorous or benthivorous waterfowl in the assessment area. However, given that water, sediment, phytoplankton, and zooplankton in the assessment area are contaminated with PCBs, it is likely that aquatic plants consumed by waterfowl are contaminated with PCBs. In addition, Beyer et al. (1997, 1998) found that up to 18% of the diet of swans, geese, and ducks can be incidentally ingested sediment. Thus, herbivorous and benthivorous waterfowl species are likely to be exposed to PCBs in the assessment area through ingestion of food items and incidental uptake of sediments.

Tissue analysis of various waterfowl species confirms that individuals from Green Bay have been exposed to PCBs. These data are summarized in Table 6-1.

Table 6-1
PCB Concentrations Measured in the Tissues of Waterfowl from Green Bay

Species	Diet[a]	Tissue	Site	Year	Mean PCB Conc. (mg/kg wet weight)	Reference
Red-breasted merganser	fish	eggs	Door Cty.	1975	44.7	White and Cromartie, 1977
Red-breasted merganser	fish	eggs	Door Cty.	1977/1978	20	Haseltine et al., 1981
Red-breasted merganser	fish	eggs	Door Cty.	1990	11.1	Williams et al., 1995b
Common merganser	fish	eggs	Door Cty.	1975	79.4	White and Cromartie, 1977
Mallard	plants	eggs	Door Cty.	1977/1978	2	Haseltine et al., 1981
Mallard	plants	muscle, skin, and fat	Lower Fox River	1985/1986	0.4	Amundson, undated
Mallard	plants	muscle, skin, and fat	Lower Fox River and inner Green Bay	1987	0.37	Wisconsin DNR wildlife contaminants database (supplied by K. Patnode, B. Hill of WDNR)
Mallard	plants	muscle and skin	Green Bay	1997	0.45	USFWS, unpublished data
Scaup	benthos	muscle and skin	Southern Green Bay	1997	2.0	USFWS, unpublished data

a. Assumed based on species description in Ehrlich et al., 1988.

Except for Amundson (undated), none of these studies reported PCB concentrations in waterfowl from reference areas. Amundson reported that the mean PCB concentration among 55 mallard from the assessment area in 1985-1986 (0.43 mg/kg wet weight) was significantly greater than that reported among mallard from inland areas of Wisconsin (0.19 mg/kg wet weight). Amundson also found that only 20% of inland Wisconsin mallard exceeded the PCB detection limit, compared with 64% of mallard from the assessment area.

Overall, these data confirm that waterfowl in Green Bay have been exposed to PCBs and that, at least for mallard, they have PCB body burdens that exceed those from reference areas.

6.3 INJURIES TO WATERFOWL IN GREEN BAY

In this section we show that waterfowl in the assessment area have been injured by their exposure to PCBs. These injuries comprise exceedences of "action or tolerance levels established under section 402 of the Food, Drug and Cosmetic Act" [43 CFR § 11.62(f)(1)(ii)] and exceedences of "levels for which an appropriate State health agency has issued directives to limit or ban consumption" [43 CFR § 11.62(f)(1)(iii)]. We first discuss the procedural bases for the federal tolerance level and the state advisories. We then present data that show that waterfowl in the assessment area have PCB tissue concentrations that exceed federal and state action or tolerance levels. Lastly we describe the waterfowl consumption advisory imposed in the assessment area by the State of Wisconsin in response to the measured PCB concentrations in waterfowl tissue.

6.3.1 Basis of the Federal Tolerance Level

The Food, Drug, and Cosmetic Act (21 U.S.C. 301 et seq.) authorizes the federal Food and Drug Administration (FDA) to protect the public health by regulating food shipped in interstate commerce. Sections 402 and 406 of the Act prohibit food from interstate commerce if the food contains any added poisonous or deleterious substance that is unsafe, unless the presence of the poisonous or deleterious substance cannot be avoided. Section 406 authorizes the FDA to limit the quantities of such substances by using formal rulemaking to set legal limits called tolerances. The tolerances are set at the level necessary to protect public health, taking into account the extent to which the substance is unavoidable and the ways that a consumer may be affected by the same or other deleterious substances (44 Fed. Reg. 38,330).

No tolerances have been established for waterfowl per se, but in 1972, the FDA proposed tolerances for PCBs in poultry (37 Fed. Reg. 5,705). The FDA acknowledged that there was limited knowledge of the toxicological effects of PCBs, but that PCBs appeared to be of moderate acute toxicity. The proposed temporary tolerance for poultry was 5.0 mg/kg wet weight on a fat basis. In 1973, the FDA issued regulations setting temporary tolerances for PCBs in poultry

(38 Fed. Reg. 18,096). The FDA called the PCB tolerances "temporary" because "new data may justify a further downward revision of the tolerances" (42 Fed. Reg. 17,493).

In 1977, the FDA proposed reducing the poultry tolerance from 5.0 mg/kg wet weight (fat basis) to 3.0 mg/kg wet weight (fat basis). In proposing this reduction, the FDA stated that it needed to balance protecting public health with avoiding excessive losses of food (42 Fed. Reg. 17,487). The proposal to reduce the tolerance to 3.0 mg/kg wet weight (fat basis) contained an extensive discussion of the basis for the decision based on the contaminant having become more avoidable and on new toxicity data on PCBs.

On June 29, 1979, the FDA issued a final rule reducing tolerances for PCBs (44 Fed. Reg. 38,330). The FDA also removed the designation "temporary" from the tolerances because the word was deemed not to have legal significance under Section 406 of the Food, Drug, and Cosmetic Act.

6.3.2 Basis for State Waterfowl Consumption Advisories

In 1984 Wisconsin initiated its wildlife contaminant monitoring program (Amundson, undated; Miller, 1987). This program was initiated for two reasons: first the state was cognizant that it had a responsibility to assure that game harvested by sportsmen was "healthy, wholesome, and free of contamination" (Miller, 1987); second the state wanted to monitor contaminant levels in wildlife species (Miller, 1987).

The results of the monitoring program showed that the majority of game over most of the state was relatively free of contamination. However, for certain species in certain regions, contaminants such as PCBs were elevated (Amundson, undated; Miller, 1987). Wisconsin then developed procedures for issuing consumption advisories for waterfowl (Miller, 1987). These procedures indicate that an advisory will be issued once a year, that the mechanism for public notification will be through a news release, that preparation and cooking recommendations will form part of the advisory notice, and that the advisory will be specific about areas and species covered. The threshold level that was adopted by the state, and that triggered this PCB advisory, was the federal tolerance level for poultry of 3 mg/kg wet weight PCBs on a fat basis.

6.3.3 Exceedences of FDA and State Tolerance Levels

There are two sources of data on PCB contamination of waterfowl species in the assessment area. The first is the Wisconsin DNR wildlife contaminants database (Amundson, undated; unpublished data supplied by K. Patnode and B. Hill of Wisconsin DNR). These data form the basis for the consumption advisories issued by the Wisconsin DNR and printed in the yearly hunting regulations guide. The second source of data is a study undertaken by the Service during the summer and fall of 1997 to update, replicate, and extend the findings of the Wisconsin DNR monitoring (USFWS, unpublished data). In this study, a variety of waterfowl species were collected and edible portions were analyzed for total PCBs and lipids. Collections were made on several different dates in 1997 and at several locations. Details regarding the sample collection and analysis procedures are provided in Appendix B.

The Wisconsin DNR data show that in 1985-1986 the mean PCB concentration of 55 mallard collected from the Lower Fox River was 6.05 mg/kg fat (Amundson, undated) (Figure 6-1). In 1987 it was 9.02 mg/kg fat in 33 mallard collected from the Lower Fox River and inner Green Bay (Wisconsin DNR wildlife contaminants database) (Figure 6-1).

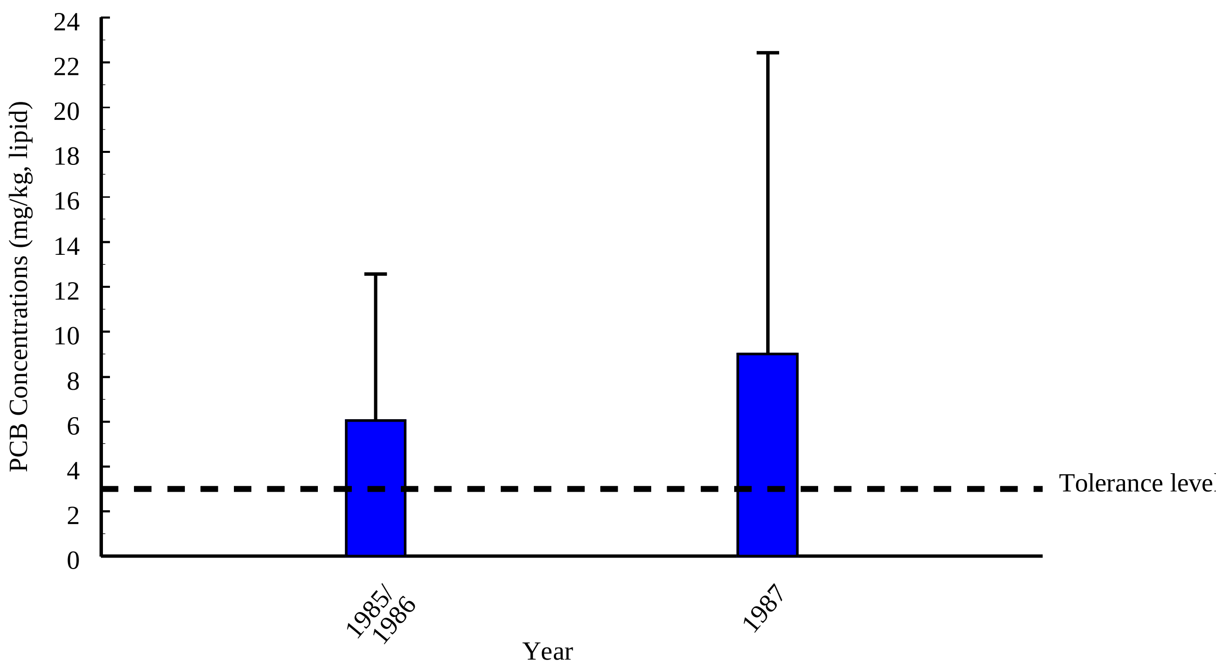


Figure 6-1. Mean PCB concentrations in mallard from the assessment area. Dashed line is the FDA and State of Wisconsin tolerance level. Vertical line represents one standard deviation.

The first USFWS collection was of 10 mallard from Lower Green Bay along the shoreline from the mouth of the Fox River eastward to the vicinity of Point au Sable in June 1997. These birds were probably summer residents. In addition to the mallard, one lesser scaup was collected. This bird apparently had not migrated to its normal breeding grounds. Eight of 10 of the mallards and the scaup exceeded the federal and the Wisconsin tolerance levels in skin plus attached muscle fillets (Table 6-2). Many of these birds also exceeded the tolerance levels in muscle tissue alone. The scaup exceeded the tolerance levels for both tissue types.

Table 6-2
PCB Concentrations in Lesser Scaup and Mallards Collected by USFWS in Southern Green Bay on June 12, 1997

Species	Sex	Age	PCBs (mg/kg wet weight lipid)[a] Muscle	PCBs (mg/kg wet weight lipid)[a] Muscle and Skin
Lesser scaup	male	adult	16.3[b]	23.3[b]
Mallard	male	adult	nd	2.0
Mallard	female	adult	nd	2.2
Mallard	female	adult	nd	8.0[b]
Mallard	male	adult	nd	4.5[b]
Mallard	male	adult	15[b]	19.5[b]
Mallard	male	adult	6.6[b]	15.4[b]
Mallard	male	adult	13.9[b]	17.4[b]
Mallard	female	adult	2.9	9.6[b]
Mallard	male	adult	nd	5.6[b]
Mallard	male	adult	5.9[b]	16.9[b]

a. The level of detection was 0.02 mg/kg wet weight, which is equivalent to 0.4 mg/kg wet weight on a lipid basis for a tissue sample of 5% lipid.
b. Exceeds federal and State of Wisconsin tolerance levels of 3 mg/kg wet weight, fat basis.

nd = not detected.

Another sample of waterfowl was collected by the USFWS (unpublished data) near Point au Sable during the peak of the influx of migratory ducks from northern areas in late October and November 1997. A variety of species was collected (Tables 6-3 and 6-4), spanning the range of diving ducks normally encountered by hunters in this area. Two of these ducks had PCB residues in their tissues that exceeded the federal and the Wisconsin tolerance levels.

A third set of samples was collected in September 1997 in northern Door County, Green Bay, and adjacent Lake Michigan (Table 6-5). All of these birds were diving ducks, which feed on a diet of animal rather than plant material. Of the 14 birds sampled, 13 had PCB residues that exceeded the federal and the Wisconsin tolerance levels.

The data in the Wisconsin DNR database and in Tables 6-2 through 6-5 show that many of the waterfowl collected on the Lower Fox River and Green Bay have body burdens of PCBs that

Table 6-3
PCB Concentrations in Skin and Breast Muscle of Waterfowl Collected from Point au Sable, Southern Green Bay on October 27, 1997

Species	Sex	Age	PCBs (mg/kg wet weight lipid)
Greater scaup	male	immature	1.6
Greater scaup	female	immature	3.1[a]
Greater scaup	male	immature	0.8
Greater scaup	female	immature	0.3
Greater scaup	female	immature	2.2
Greater scaup	male	immature	0.2
Lesser scaup	female	adult	2.7
Lesser scaup	male	immature	0.6
Common goldeneye	male	adult	14.1[a]

a. Exceeds federal and Wisconsin tolerance levels of 3 mg/kg wet weight, fat basis.

Table 6-4
PCB Concentrations in Skin and Breast Muscle of Waterfowl Collected from Point au Sable, Southern Green Bay on November 12-13, 1997

Species	Sex	Age	PCBs (mg/kg wet weight lipid)
Greater scaup	male	adult	3.3[a]
Lesser scaup	male	adult	2.4
Lesser scaup	male	immature	1.2
Lesser scaup	male	adult	5.1[a]
Lesser scaup	male	immature	2.5
Lesser scaup	female	immature	1.4
Lesser scaup	male	immature	0.4
Lesser scaup	male	immature	0.8
Lesser scaup	male	immature	0.9
Bufflehead	female	immature	0.4
Bufflehead	male	adult	1.4
Common goldeneye	female	immature	0.1
Common goldeneye	female	immature	1.5
Common goldeneye	male	immature	0.1

a. Exceeds federal and Wisconsin tolerance levels of 3 mg/kg wet weight, fat basis.

Table 6-5
PCB Concentrations in Skin and Breast Muscle of Waterfowl Collected from the Door Passage to Bailey's Harbor, Lake Michigan on September 16-17 and September 22 and 26, 1997

Species	Sex	Age	PCBs (mg/kg wet weight lipid)[a]
Common goldeneye	male	adult	3.5[b]
Ruddy duck	female	adult	4.6[b]
Common merganser	male	immature	8.3[b]
Common merganser	female	immature	373.9[b]
Common merganser	female	immature	36.3[b]
Common merganser	female	adult	27.4[b]
Common merganser	female	immature	25.7[b]
Common merganser	female	adult	30.3[b]
Common merganser	male	immature	10.8[b]
Common merganser	male	immature	16.8[b]
Red-breasted merganser	female	adult	36.9[b]
Red-breasted merganser	female	adult	11.4[b]
Red-breasted merganser	female	adult	nd
Red-breasted merganser	female	immature	25.3[b]

a. The lower limit of detection was 0.01 mg/kg wet weight, which is equivalent to 0.2 mg/kg wet weight lipid for a tissue sample of 5% lipid.
b. Exceeds federal and Wisconsin tolerance levels.

nd = not detected.

exceed the federal tolerance levels and the Wisconsin advisory level. The data also indicate that the PCB body burdens in waterfowl are determined by the residence time that the individual has spent in the system (summer resident mallards had higher PCB concentrations than migrant birds that had most likely recently arrived in the assessment area) and by their diet (individuals whose diet comprises fish generally had higher PCB concentrations than nonpiscivores).

6.3.4 The State of Wisconsin Waterfowl Consumption Advisory

In response to the PCB tissue concentrations measured in Green Bay waterfowl, the Wisconsin DNR and the Division of Health issued a waterfowl consumption advisory in 1987 (Wisconsin DNR, 1987). The advisory was for mallards taken in the "Lower Fox River from Lake Winnebago at Neenah and Menasha downstream, including Little Lake Butte des Morts, to the northeast city limits of Kaukauna," and the "Lower Fox River from the DePere Dam to the river's mouth at Green Bay, and lower Green Bay south of a line from Point au Sable west to the west shore of Green Bay" (health advisory recommendations in annual Wisconsin DNR hunting pamphlets). The areas covered by the advisory are shown in Figure 6-2. The advisory advises hunters to "remove all skin and visible fat before cooking mallard ducks using these waters. Discard drippings or stuffings because they may retain fat that contains PCBs" (health advisory recommendations in annual Wisconsin DNR hunting pamphlets).

Since the first advisory was issued in 1987, the advisory has remained in place every year. The advisories are issued each year in the annual hunting guide distributed by the Wisconsin DNR.

The text for the advisory specifies that the advisory is being issued because of PCB contamination.

6.4 SUMMARY

The data reported in this chapter show that waterfowl from the Lower Fox River and from Green Bay are contaminated by PCBs. Resident species of waterfowl and species that feed relatively high in the food chain show the greatest body burdens. Apparently, migratory species newly arrived in the assessment area have relatively low levels of contamination. It is likely that these levels increase with the duration of their residence time in the Lower Fox River and Green Bay. PCB concentrations measured in waterfowl have and continue to exceed federal tolerance levels for poultry.

The Wisconsin DNR and the Division of Health issued a consumption advisory for mallards from the Lower Fox River and inner Green Bay in 1987 because of their elevated levels of PCBs. This advisory is still in force. The data reviewed in this chapter show that the Wisconsin DNR and Division of Health imposition of the consumption advisory on mallards is justified by the elevated concentrations of PCBs in the tissues of that species.

The elevated PCB tissue concentrations have resulted in waterfowl in the assessment area being injured based on the injury definitions at 43 CFR § 11.62(f)(1)(ii) (concentrations of hazardous substances sufficient to " exceed action or tolerance levels established under section 402 of the Food, Drug and Cosmetic Act" or "exceed levels for which an appropriate State health agency has issued directives to limit or ban consumption").

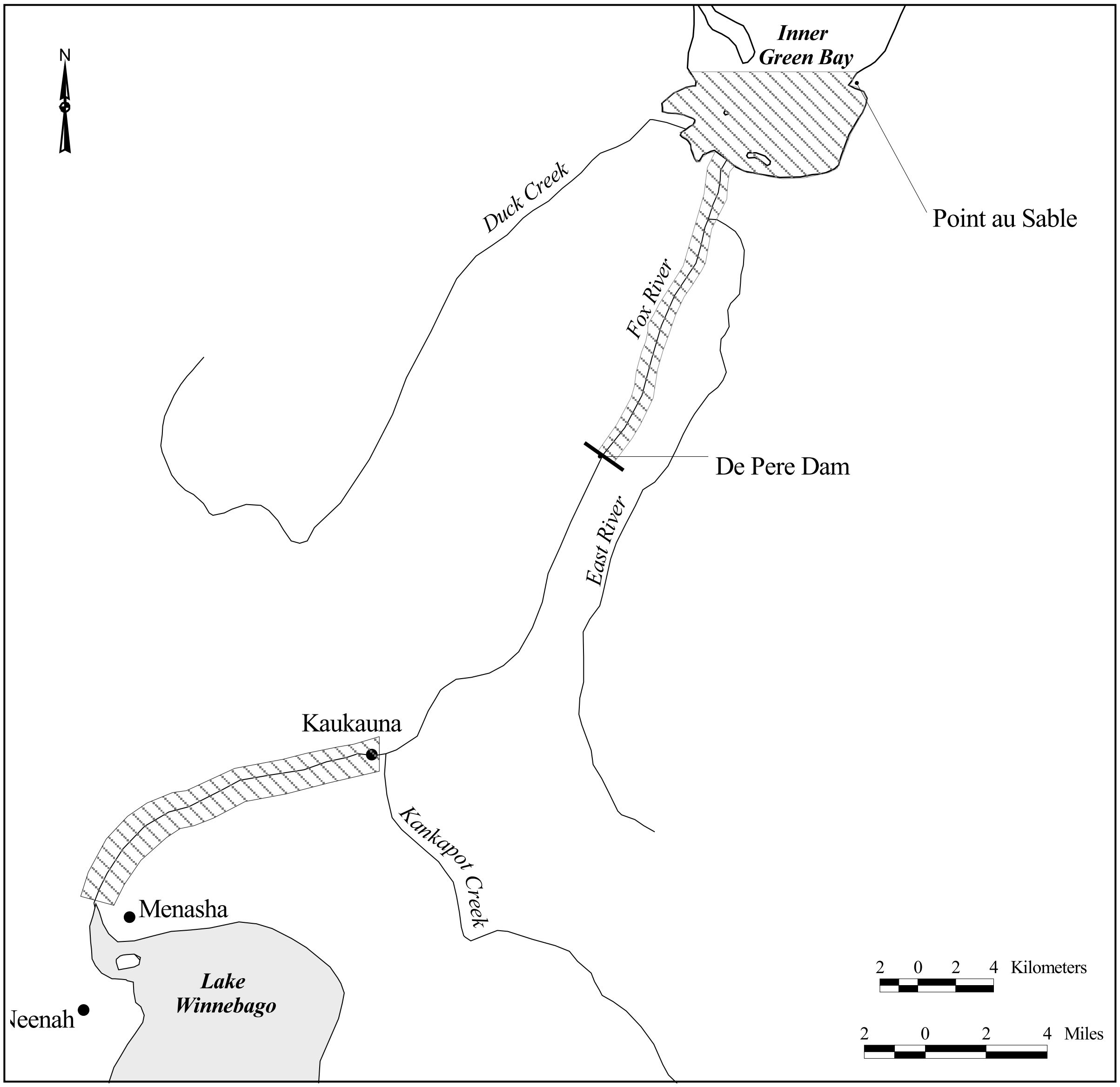


Figure 6-2. Areas covered by the Wisconsin waterfowl consumption advisory (hatched).

CHAPTER 7
WEIGHT OF EVIDENCE EVALUATION

7.1 INTRODUCTION

Chapter 5 of this report described scientific evidence that adverse effects have occurred among birds in the assessment area. As noted in that chapter, there is uncertainty regarding the extent and causes of these effects. In this chapter we carry out a weight of evidence evaluation of the scientific data to address and answer two questions:

1. Is it more likely than not that adverse effects that are consistent with the definitions of biological injury in the Departmental regulations [43 CFR § 11.62 (f)(1)(i)] have occurred among assessment area birds?

We address this question by categorizing the evidence provided by each study to determine whether the case for the occurrence of each reported adverse effect is either:

- ***Highly likely.*** There is little or no doubt that the evidence reported in the study supports the conclusion that birds experienced adverse effects.
- ***Likely.*** While there may be some uncertainty associated with the evidence presented in the study, the evidence suggests that it is more likely than not that birds experienced adverse effects.
- ***Unlikely.*** The evidence indicates that it is unlikely that the reported adverse effects occurred.
- ***Indeterminate.*** Although the data in the report may indicate that adverse effects have occurred, the data do not allow an unequivocal determination. This categorization does not necessarily indicate that the adverse effects have *not* occurred.

We considered the following issues when evaluating studies of adverse effects:

- Did the study include appropriate reference areas or controls?
- How adequate were the field/laboratory methods reported in the study?
- Were sample sizes large enough to provide adequate statistical power?

- Were statistically significant differences measured between the assessment and reference/control conditions?
- Were the effects demonstrated both in the field and under controlled conditions in the laboratory?
- Do any uncertainties in the study cast doubt on the conclusions that were drawn?

2. Is it more likely than not that the adverse effects were caused by exposure to PCBs?

For those effects determined to be either highly likely or likely in the above evaluation, we evaluate causation as follows:

- ***Likely.*** While there may be some uncertainty associated with the causality, it is more likely than not that PCBs were at least a contributing factor to the adverse effect.
- ***Unlikely.*** The evidence indicates that it is not likely that PCBs caused or contributed to the adverse effect.
- ***Indeterminate.*** The data reported in the study do not allow an unequivocal determination of whether or not the adverse effects were caused by PCBs. This categorization does not necessarily indicate that the adverse effects were *not* caused by PCBs.

We considered the following when evaluating causation:

- To what extent were study results consistent with laboratory studies of the toxicology of PCBs?
- Was consistency of effects observed across studies?
- Were dose-response relationships observed?
- Were the effects consistent with definitions of injury in the Departmental NRDA regulations?
- Is there evidence that supports an alternative cause?

Lastly, we evaluated the scientific evidence from all species studied in the assessment area to determine whether there are cross-species consistencies in adverse effects that could further clarify our understanding of causation.

7.2 EVALUATION OF EVIDENCE THAT ADVERSE EFFECTS HAVE OCCURRED

7.2.1 Terns

Forster's Terns

In this analysis, we evaluate the evidence provided in two studies: Kubiak et al. (1989) and Hoffman et al. (1987). The evidence from these studies (summarized in Table 7-1 at the end of Section 7.2) supports the conclusion that it is ***highly likely*** that two of the adverse effects reported for Forster's terns in the assessment area, reduced hatching success and embryonic deformities, occurred.

The reduced hatching success was demonstrated by Kubiak et al. (1989) both in the field and under controlled conditions in the laboratory. The results of these studies leave little room for doubt that the reduced hatching success occurred. Hoffman et al. (1987) demonstrated, also under controlled conditions in the laboratory, that physical deformations also occurred in the Green Bay Forster's tern hatchlings. Statistically significant differences were found between Green Bay and reference area hatchlings for two of these deformations (femur length and liver to body weight ratios); three instances of obvious skeletal deformities were found in Green Bay chicks, but none in reference area chicks. These results provide evidence that it is highly likely that assessment area Forster's tern chicks suffered physical deformations.

The evidence for the occurrence of behavioral abnormalities in Forster's terns is deemed ***likely*** in this evaluation. The time it took Green Bay Forster's tern eggs to hatch was significantly longer than reference area Forster's tern eggs. It is very likely that this effect was caused by reduced incubation attentiveness in Green Bay adult terns. However, the incubation schedules of the terns were not measured directly and, as a result, there is uncertainty associated with this conclusion.

Common Terns

The evidence provided by Hoffman et al. (1993) was evaluated. The data supports the conclusion that it is ***likely*** that the two adverse effects reported for common terns in the assessment area, reduced hatching success and embryonic deformities, occurred (summarized in Table 7-1 at the end of Section 7.2).

Under controlled conditions in laboratory incubators Hoffman et al. (1993) found significantly lower hatching success in Green Bay eggs compared with eggs from one of the reference colonies. Hoffman et al. (1993) also demonstrated under controlled conditions in the laboratory that physical deformations occurred in the Green Bay common tern hatchlings. Statistically significant differences in femur length were found between Green Bay hatchlings and hatchlings from one of the reference colonies (the same colony that had significantly higher hatching success). The results in Hoffman et al. provide good evidence that the Green Bay common terns had lower hatching success than eggs from the reference area. They also show that it is ***likely*** that a higher rate of deformities occurred in the Green Bay hatchlings. The only uncertainty is introduced by the fact that significant differences were found (in both metrics) for only one of the reference colonies (Cut River), but not the other (Point aux Chenes). Nevertheless, the deformity rate from the Point aux Chenes reference colony was lower (0% of hatchlings) than from Green Bay (11% of hatchlings). However, the sample size from the Point aux Chenes colony was smaller than that from Green Bay or the other reference colony (20, 35, and 35, respectively), and it is possible that this may have contributed to the lack of statistical significance.

Caspian Tern

The weight of evidence evaluation that adverse effects have occurred among Caspian terns (summarized in Table 7-1 at the end of Section 7.2) is based on data in Ludwig et al. (1996), Ludwig and Ludwig (undated report b), and Mora et al. (1993). Ludwig et al. (1996) showed that physical deformations occurred at a greater rate among Green Bay Caspian tern embryos than in embryos from reference areas. Some uncertainty is introduced into this determination by the fact that Ludwig and Ludwig (undated report b) did not find any deformities in Green Bay Caspian tern embryos. However, Ludwig and Ludwig (undated report b) focused on hatched chicks, which were shown in Ludwig et al. (1996) to have low rates of deformity. Thus, the probability of detecting effects in hatched chicks is lower. Although less conclusive than for Forster's terns, it is deemed ***likely*** that unhatched Caspian terns in the assessment area have suffered greater incidences of deformities than terns elsewhere.

The evidence that Caspian terns in the assessment area have exhibited behavioral abnormalities (Mora et al., 1993) cannot be substantiated using the data available. Although reduced site fidelity could reflect a behavioral abnormality caused by PCBs, the major effect that is measured, reduced fidelity to the nesting colony, could be a function of disturbance caused by the method used by Mora et al. (1993) to trap birds, cannon netting. Moreover, the other major method employed, analysis of band recoveries, has many potential biases, including band loss and wear, likelihood of recovery, and search effort. These biases were not adequately addressed in the study. As a result, it is concluded that the existence of behavioral effects in assessment area Caspian terns is ***indeterminate***.

7.2.2 Double-Crested Cormorants

This weight of evidence evaluation that adverse effects have occurred in double-crested cormorants in the assessment area is based on consideration of Ross and Weseloh (1988), Fox et al. (1991), Tillitt et al. (1992), Larson et al. (1996), and Ludwig et al. (1996). The results are summarized in Table 7-1 at the end of Section 7.2. These studies support the conclusion that it is ***highly likely*** that the reported adverse effects occurred in the assessment area. Of the two major studies that independently compared hatching success among Green Bay cormorants with reference areas, both found that Green Bay hatching success was significantly lower. All three major studies that independently compared embryo/chick deformity rates in Green Bay with reference areas, found that the deformity rate in Green Bay was significantly higher. The only uncertainty concerns the "background" rate of deformities. Ross and Weseloh (1988) found anomalously high rates of head and bill defects from one small subcolony at Lake Winnipegosis. This, however, was an isolated finding, and other studies have shown that, in comparison to Green Bay, the rate of deformities at Lake Winnipegosis is typically low.

7.2.3 Bald Eagles

The weight of evidence evaluation that adverse effects have occurred in bald eagles in the assessment area is based on data in Dykstra and Meyer (1996) and in Chapter 5 of this report, and is summarized in Table 7-1 at the end of Section 7.2. Data confirm that bald eagles in the assessment area have consistently had breeding productivity that is significantly reduced compared to that in reference areas that are not exposed to point source releases of PCBs. In this evaluation, therefore, we consider it ***highly likely*** that this adverse effect occurred.

7.2.4 Black-Crowned Night Herons

The weight of evidence evaluation that adverse effects have occurred in black-crowned night herons in the assessment area is based on two studies, Hoffman et al. (1993) and Rattner et al. (1993) and is summarized in Table 7-1 at the end of Section 7.2. Only one of the studies (Hoffman et al., 1993) demonstrated physical deformations (liver to body weight ratios). A limitation of the Hoffman et al. study is that the sample size was small (n = 5). The sample size in the Rattner et al. study (in which no deformities were found) was larger (n = 18). On balance, the evidence in these studies does not provide compelling support for the conclusion that adverse effects have occurred among Green Bay black-crowned night herons and we consider the evidence ***indeterminate***.

7.2.5 Tree Swallow and Red-Breasted Merganser

No evidence that adverse effects have occurred among Green Bay tree swallows or red-breasted mergansers has been reported in the literature. These species are not considered further in this evaluation.

7.2.6 Summary

Table 7-1 summarizes the results of the weight of evidence evaluation of adverse effects. It is highly likely that adverse effects have occurred among assessment area Forster's terns, double-crested cormorants, and bald eagles. These effects comprise reproductive malfunctions (reduced hatching success), and physical deformations (head and bill deformaties). It is likely that adverse effects have occurred among assessment area common terns (reduced hatching success and physical deformations) and Caspian terns (physical deformations in unhatched chicks). It was concluded that adverse effects among black-crowned night herons, tree swallows, or red-breasted mergansers could not be substantiated using the available data.

7.3 WEIGHT OF EVIDENCE EVALUATION OF CAUSATION

In this section we evaluate whether it is more likely than not that those adverse effects identified in Section 7.2 as being highly likely or likely were caused by PCBs. The results of these evaluations are summarized in Table 7-2 at the end of Section 7.3.

7.3.1 Forster's, Common, and Caspian Terns

The data evaluated support the conclusion that it is ***likely*** that most of the adverse effects observed among assessment area Forster's and common terns have been caused, at least in part, by exposure to PCBs. This conclusion is based on the following:

- The types of effects that were observed in both species (reproductive malfunctions, deformities) are consistent with PCB toxicosis.

- Both species are likely sensitive to PCBs as discussed in Chapter 5 of this report.

- The concentrations of PCBs found in Green Bay Forster's and common tern eggs exceeded the 5-10 mg/kg toxicity range for sensitive species.

- There was a dose-response relationship established in the Kubiak et al. (1989) study.

- The study performed by Harris et al. (1993) failed to find reduced hatching success. However, at the time of this study PCB concentrations in Green Bay Forester's tern eggs were more than 50% lower than in 1983, when the Kubiak et al. (1989) study was performed. In 1988, none of the adverse effects observed in 1983 were found. Therefore, the Harris et al. study is not considered to present confounding data. Indeed, it may be suggestive of an exposure-response relationship.

Table 7-1
Weight of Evidence Evaluation that Adverse Effects Have Occurred among Assessment Area Birds

Species	Adverse Effect	Study	Support for Adverse Effect Having Occurred	Comments	Result of Evaluation
Forster's tern	Reduced hatching success	Kubiak et al., 1989	Hatching success in Green Bay significantly lower than reference colony. Hatching success of Green Bay eggs significantly lower in laboratory than reference colony eggs.	Comprehensive, rigorous study design. Reference and Green Bay eggs incubated under identical conditions in laboratory. Appropriate reference area used. Appropriate statistical tests performed.	Highly likely
	Physical deformations	Hoffman et al., 1987	Significantly shorter femurs in Green Bay hatchlings than reference colony hatchlings. Significantly greater liver to body weight ratios in Green Bay hatchlings than reference colony hatchlings. Three instances of skeletal deformities in Green Bay hatchlings but none in reference colony hatchlings.	Assessments carried out under controlled laboratory conditions. Appropriate statistical tests used to compare samples. Appropriate reference area used.	Highly likely
	Behavioral abnormalities	Kubiak et al., 1989	Significantly extended incubation periods of Green Bay clutches compared with reference area. Lake Poygan eggs incubated by Green Bay adults had low hatching success, suggesting reduced incubation attentiveness in Green Bay terns.	Appropriate reference area used. Appropriate statistical tests used to compare samples. Extended incubation periods were probably due to reduced incubation attentiveness in adult Green Bay terns. However, this is uncertain since incubation schedules were not measured.	Likely

Table 7-1 (cont.)
Weight of Evidence Evaluation that Adverse Effects Have Occurred among Assessment Area Birds

Species	Adverse Effect	Study	Support for Adverse Effect Having Occurred	Comments	Result of Evaluation
Common tern	Reduced hatching success	Hoffman et al., 1993	Hatching success of Green Bay eggs in incubators significantly lower than eggs from one of two reference colonies.	Reference and Green Bay eggs incubated under identical conditions. Assessments carried out under controlled laboratory conditions. Appropriate statistical tests performed. Uncertainty because of significant difference found for only one of the two reference colonies.	Likely
	Physical deformations	Hoffman et al., 1993	Green Bay chicks had significantly shorter femurs than chicks from one of the two reference colonies (but not the other). Four of the Green Bay neonates were deformed compared with none of the reference birds.	Assessments carried out under controlled laboratory conditions. Appropriate statistical tests performed. Uncertainty because of significant difference found for only one of the two reference colonies.	Likely/ indeterminate
Caspian tern	Physical deformations	Ludwig et al., 1996	Greater incidence of deformities found in Green Bay embryos compared with colonies in the Great Lakes not exposed to point source releases of PCBs. Deformities consistent with those observed in other species in assessment area. Age-related incidences of deformations consistent with deformations being associated with egg mortality.	More subtle deformations may be difficult to detect or correctly classify under field conditions. No abnormalities reported in other studies (Ludwig and Ludwig, undated report b).	Likely

Table 7-1 (cont.)
Weight of Evidence Evaluation that Adverse Effects Have Occurred among Assessment Area Birds

Species	Adverse Effect	Study	Support for Adverse Effect Having Occurred	Comments	Result of Evaluation
Caspian tern (cont.)	Behavioral abnormalities	Mora et al., 1993	Apparently lower site fidelity among assessment area terns than reference areas.	Study provides some evidence that Green Bay Caspian terns may have suffered behavioral abnormalities. However, two limitations on interpretation: method of capture (cannon netting) very intrusive and could, potentially, cause terns to not return to colony in future years; conclusion based on analysis of band recoveries and failed to address biases inherent in this procedures.	Indeterminate
Double-crested cormorant	Reduced hatching success	Tillitt et al., 1992 Larson et al., 1996	Hatching success at Spider Island lowest of Great Lakes colonies evaluated. Hatching success at Spider island significantly lower than at Lake Winnipegosis.	Study compared a wide range of sites, not just one reference site. Study used appropriate reference site. Used appropriate field and statistical methods.	Highly likely

Table 7-1 (cont.)
Weight of Evidence Evaluation that Adverse Effects Have Occurred among Assessment Area Birds

Species	Adverse Effect	Study	Support for Adverse Effect Having Occurred	Comments	Result of Evaluation
Double-crested cormorant (cont.)	Physical deformities	Fox et al., 1991	Highest rate of head and bill deformities found in 42 Great Lakes and other colonies was in assessment area. Rate of head and bill deformities in assessment area significantly higher than at most other colonies.	Used large number of potential reference colonies. Statistical tests appropriate. Field methods appropriate.	Highly likely
		Larson et al., 1996	Bill deformities significantly more frequent at Spider Island than Lake Winnipegosis.	Statistical tests appropriate. Field methods appropriate.	
		Ludwig et al., 1996	Deformity rate in Green Bay significantly higher than at reference colonies.	More than one reference colony evaluated. Appropriate statistical comparisons performed for this report. Some uncertainties in determination of background deformity rates (Ross and Weseloh, 1988).	

Table 7-1 (cont.)
Weight of Evidence Evaluation that Adverse Effects Have Occurred among Assessment Area Birds

Species	Adverse Effect	Study	Support for Adverse Effect Having Occurred	Comments	Result of Evaluation
Bald eagle	Reduced productivity	Dykstra and Meyer, 1996	Productivity of Green Bay bald eagles significantly lower than inland Wisconsin bald eagles.	Reference data adequate since study compared Green Bay sites with a large number of inland Wisconsin reference sites. Appropriate statistical methods used. Appropriate field methods used.	Highly likely
		This study	Productivity at Green Bay sites significantly lower than at sites in inland Wisconsin and inland Michigan.	Study compared Green Bay sites with a large number of inland Wisconsin and Michigan reference sites. Appropriate statistical methods used. Appropriate field methods used.	
Black-crowned night heron	Physical deformities	Hoffman et al., 1993	Significantly higher liver to body weight ratios in Green Bay than in reference site chicks.	Appropriate statistical methods used. Appropriate laboratory used.	Indeterminate
		Rattner at al., 1993	No deformities reported in Green Bay chicks.	Small sample size in Hoffman et al. study.	

- No alternative contaminants are likely to have caused the effects (DDE is not known to cause deformations, and PCDDs and PCDFs do not contribute substantially to the dioxin-like toxicity in the assessment area).

- Since many of the adverse effects were recorded under controlled laboratory conditions, other anthropogenic or ecological factors do not plausibly explain the results.

Although the types of deformities found in Green Bay Caspian terns by Ludwig et al. (1996) are consistent with PCB toxicosis and may have resulted from releases of PCBs into the assessment area, the evidence is equivocal, and thus we categorize it as ***indeterminate***. No relationship between PCBs and deformity rates was found in the assessment area by Yamashita et al. (1993). Also, one of the major findings of the Ludwig et al. study was that embryo deformations were likely to be associated with mortality. However, in a study that included many Caspian tern nesting sites across the Great Lakes, Ewins et al. (1994) found no relationship between egg PCB concentrations and embryo survival. Given these contradictory results from different studies, it cannot be concluded that the adverse effects observed in assessment area Caspian terns were caused by exposure to PCBs.

7.3.2 Double-Crested Cormorant

Section 7-2 concluded that it is highly likely that adverse effects (reduced hatching success and physical deformations) have occurred in assessment area double-crested cormorants. The weight of evidence evaluation in this section leads to the conclusion that it is ***likely*** that PCBs have caused, or were a significant contributing cause, of the reduced reproductive success. This conclusion is based largely on the Tillitt et al. (1992) study, which showed the following:

- Total PCB concentrations in cormorant eggs from a number of Great Lakes sites were significantly negatively correlated with hatching success.

- The correlation between contaminants in eggs and hatching success improved when H4IIE results were used as the determinate variable. This result could not have been obtained if the cause of the variation in hatching success were anything other than a dioxin-like contaminant.

- The laboratory method of sample preparation screened out PCDDs and PCDFs. Thus, these contaminants could not have contributed to the observed relationships.

- It is unlikely that any ecological or genetic factor or disease (e.g., Newcastle disease) could explain the pattern of variability in hatching success that was observed among the colonies.

The Powell et al. (1997) study failed to elicit significantly elevated embryo mortality in the laboratory by injecting Lake Winnipegosis cormorants eggs with PCB 126 at doses that exceed those observed in Green Bay. While this result is interesting and suggests that further studies need to be carried out, we consider that it does not show that PCBs do not affect hatching success in the assessment area for two reasons:

- PCB 126 is only one of the congeners that may be important in Green Bay. The contributions to toxicity by other potentially important congeners (e.g., PCB 81) were not evaluated.

- The relevance of egg injection studies to maternal transfer conditions in the field is uncertain.

Recent work by Custer et al. (in press) also demonstrates that DDE is currently affecting the hatching success of cormorants in the assessment area. However, we conclude that the Custer et al. (in press) study does not demonstrate that PCBs are *not* affecting hatching success, or did not do so in the past. In fact, the Custer et al. (in press) study did find significant relationships between egg PCB concentrations and two cormorant reproductive parameters, egg size and hatchling weight, and while the correlation between egg PCB concentrations and hatching success was not significant at $p < 0.05$, it did approach that level ($p = 0.13$). We, therefore, conclude that it is more likely than not that PCBs (possibly together with DDE) have been contributing to the reduced hatching success observed in Green Bay cormorants.

The evidence that PCBs have caused the physical deformities observed in assessment area cormorants is less certain. Although head and bill deformities are consistent with the results of laboratory studies of the effects of PCBs on birds, there are a number of other factors that could potentially cause such effects, including founder effect (reduced genetic variability due to a small colonist population with little subsequent immigration), or nutritional deficiencies. No study of deformities in Green Bay has adequately evaluated these alternative causal factors. Therefore we consider the causality to be ***indeterminate***.

7.3.3 Bald Eagles

The weight of evidence evaluation that the low productivity among bald eagles in the assessment area has been caused by PCBs is summarized in Table 7-2 at the end of Section 7.3. It is deemed ***likely*** that PCBs have contributed to the reduced productivity for the following reasons:

- The concentrations of PCBs in assessment area bald eagle eggs consistently greatly exceed the estimated toxicity range for sensitive species.

- The effect observed (embryo mortality or infertility) is consistent with PCB toxicosis.

- Human disturbance and food shortage are not contributing factors to reproductive failure in the assessment area (Dykstra and Meyer, 1996).

- There is no evidence that any other ecological factor (e.g., disease) has caused the effect, nor is it likely to do so over such a long period.

- Two alternative contaminants (PCDDs and PCDFs) are not important contributors to dioxin-like toxicity in the assessment area.

However, it cannot be concluded unequivocally that PCBs have caused the reduced productivity in assessment area bald eagles because of the potential confounding effect of DDE. While the Dykstra and Meyer (1996) study and this study have conclusively demonstrated significant negative relationships between egg PCB and DDE concentrations and productivity, neither study was able to identify the relative contributions of each. This is because PCB and DDE concentrations in Great Lakes bald eagle eggs are usually correlated. Our assessment concludes, therefore, that, based on the type of effect and the egg contaminant concentrations relative to toxicity thresholds, it is likely that both PCBs and DDE are contributing to the adverse effect, but it is not possible to identify the relative contributions.

7.4 INTER-SPECIES CONSISTENCY

The previous analyses in this chapter used a species-by-species approach to evaluate the evidence that PCB-induced adverse effects have occurred among assessment area birds. This approach, while valid, might fail to identify between-species consistencies that can further improve our understanding of the effects of contaminants in the assessment area. In this section we compare the adverse effects that have been observed among all of the species in the assessment area to determine if similarities and/or dissimilarities contribute to our understanding of causality.

The determinations presented in Table 7-3 clearly show that adverse effects caused in the laboratory and the field by PCBs were observed in every species that has been studied in detail in the assessment area. The strength of the evidence that PCBs caused these effects in birds varies from likely to indeterminate, depending on the species. However, although some studies provide only indeterminate evidence that some species may have been injured by PCBs, the consistency in effects across studies and species warns against regarding these studies as demonstrating that PCB-induced effects have *not* occurred. They only show that PCB-induced effects have not been conclusively determined.

Table 7-2
Weight of Evidence Evaluation that the Adverse Effects among Assessment Area Birds Were Caused by PCBs

Species	Adverse Effect	Study	Support for Adverse Effect Being Caused by PCBs	Result of Evaluation
Forster's tern	Reduced hatching success, physical deformations	Kubiak et al., 1989 Hoffman et al., 1987 Harris et al., 1993	Type of effects observed consistent with PCB toxicosis). Deformities not consistent with DDE toxicosis. PCB dose-response relationship established. Forster's terns likely to be sensitive to PCBs. PCB concentrations in eggs exceeded 5-10 mg/kg wet weight toxicity range for sensitive species. Low DDE concentrations in eggs (similar to concentrations in successfully reproducing Forster's terns at other sites [King et al., 1991]). Effects unlikely to be due to PCDDs or PCDFs because of their small contributions to TCDD-EQ. Harris et al. (1993) showed that reduction in PCBs in eggs associated with lack of adverse effect.	Likely
Common tern	Reduced hatching success	Hoffman et al., 1993	Type of effects observed consistent with PCB toxicosis. PCB dose-response relationship established. Common terns likely to be sensitive to PCBs.	Likely
	Physical deformations		PCB concentrations in eggs exceeded 5-10 mg/kg toxicity range for sensitive species. No significant difference in DDE concentrations in Green Bay and reference eggs. Effects unlikely to be due to PCDDs or PCDFs because of their small contributions to TCDD-EQ.	Likely/ indeterminate

Table 7-2 (cont.)
Weight of Evidence Evaluation that the Adverse Effects among Assessment Area Birds Were Caused by PCBs

Species	Adverse Effect	Study	Support for Adverse Effect Being Caused by PCBs	Result of Evaluation
Caspian tern	Physical deformations	Yamashita et al., 1993 Ludwig et al., 1996 Ewins et al., 1994	Deformities observed by Ludwig et al. consistent with PCB toxicosis, but not DDE. Effects unlikely to be due to PCDDs or PCDFs due to their small contributions to TCDD-EQ. However: No relationship between Green Bay deformities and PCB concentrations in eggs (Yamashita et al., 1993). No relationship between embryo survival and egg PCB concentrations across Great Lakes (Ewins et al., 1994).	Indeterminate

Table 7-2 (cont.)
Weight of Evidence Evaluation that the Adverse Effects among Assessment Area Birds Were Caused by PCBs

Species	Adverse Effect	Study	Support for Adverse Effect Being Caused by PCBs	Result of Evaluation
Double-crested cormorant	Reduced hatching success	Tillitt et al., 1992 Larson et al., 1996 Powell et al., 1997 Custer et al., in press	PCB dose-response relationship established. PCB concentrations in eggs exceeded 5-10 mg/kg toxicity range for sensitive species. Effects unlikely to be due to PCDDs or PCDFs. Effect shown in avian laboratory studies to be caused by PCBs but not DDE. Custer et al. (in press) found that PCBs significantly negatively correlated with egg size and hatchling weight. However: Custer et al. (in press) found significant negative correlation between DDE and hatching success but none between PCBs and hatching success (though p = 0.13). Powell et al. (1997) unable reproduce reduction in hatching success in laboratory by injecting cormorant eggs with concentrations of PCB 126 representative of egg concentrations in Green Bay.	Likely
	Physical deformations		Custer et al. (in press) showed that in 1994 and 1995 deformity rates higher among Spider Island chicks than Cat Island chicks. However, respective rates not known for any other years. Other "natural" potential causes not evaluated adequately.	Indeterminate

Table 7-2 (cont.)
Weight of Evidence Evaluation that the Adverse Effects among Assessment Area Birds Were Caused by PCBs

Species	Adverse Effect	Study	Support for Adverse Effect Being Caused by PCBs	Result of Evaluation
Bald eagle	Reduced productivity	Dykstra and Meyer, 1996 this study	PCB dose-response relationship established. PCB concentrations in eggs exceed 5-10 mg/kg toxicity range for sensitive species. PCB concentrations in eggs exceed thresholds established by Weimeyer et al. (1984) and Kubiak and Best (1991). Effects unlikely to be due to PCDDs or PCDFs due to their small contributions to TCDD-EQ. However: This and previous studies unable to separate the effects of PCBs and DDE due to their correlation in eggs.	Likely

7.5 SUMMARY

This weight of evidence evaluation of the data pertaining to the occurrence and causes of adverse effects in assessment area birds has demonstrated the following:

- Forster's, common, and Caspian terns have either suffered, or are likely to have suffered, adverse effects in the assessment area. These include low reproductive success, behavioral abnormalities, and physical deformations. The adverse effects in Forster's and common tern have more likely than not been caused by exposure to PCBs. It is uncertain whether PCBs caused the adverse effects observed in Caspian terns, though the effects are consistent with PCB toxicosis.

- Double-crested cormorants have suffered adverse effects in the assessment area. These comprise reduced hatching success and physical deformations. It is likely that PCBs have caused or contributed to the reduced reproductive success in assessment area double-crested cormorants, but the evidence linking head and bill deformities to PCBs is uncertain, although the effects are consistent with PCB toxicosis.

- Bald eagles have suffered reduced productivity in the assessment area. PCBs are likely to have caused or contributed to the reduced productivity in assessment area bald eagles. However, the relative contributions of PCBs and DDE are uncertain.

Table 7-3
Adverse Effects Documented in Assessment Area Birds and the Likelihood that They Were Caused by PCBs

Adverse Effect	Species	Evidence that Adverse Effect Occurred	Evidence that Adverse Effect Caused by PCBs
Reduced hatching success/ productivity	Forster's tern	Highly likely	Likely
	Common tern	Likely	Likely
	Double-crested cormorant	Highly likely	Likely
	Bald eagle	Highly likely	Likely
Physical deformations	Forster's tern	Highly likely	Likely
	Common tern	Likely	Likely/indeterminate
	Caspian tern	Likely	Indeterminate
	Double-crested cormorant	Highly likely	Indeterminate
	Black-crowned night heron	Indeterminate	Indeterminate
Behavioral abnormalities	Forster's tern	Likely	Likely
	Caspian tern	Indeterminate	Indeterminate

CHAPTER 8
INJURY DETERMINATION

8.1 OVERVIEW

The purpose of this chapter is to present a determination of injury for avian resources of the Lower Fox River/Green Bay assessment area. This injury determination is consistent with the components of the Departmental NRDA regulations at 43 CFR §§11.61-11.64 and is based on the data and information presented in the preceding chapters of this report. The injury determination contained herein is organized as follows:

- *Section 8.2* presents the relevant definitions of injury, as outlined in 43 CFR §11.62. These definitions of injury represent the adverse effects for which the injury determination has been conducted.

- *Section 8.3* presents the results of pathway determination, as outlined in 43 CFR §11.63. This section focuses on confirming the pathways by which assessment area birds have come to be exposed to PCBs. Separate pathway reports being prepared by the Trustees will present more detailed pathway data establishing those pathways by which PCBs have and continue to be transported throughout the environment of the assessment area.

- *Section 8.4* presents conclusions regarding the results of injury determination testing and sampling (43 CFR §11.64) and injury conclusions for the various injury definitions. Injuries to avian resources are determined in this report primarily through the use and interpretation of historical studies of birds in the assessment area. As noted previously in this report, PCB contamination in Green Bay birds was first detected in the early 1970s (Bishop et al., 1992). Since then, multiple studies have been conducted on the exposure to and accumulation of PCBs in Green Bay birds and on adverse effects resulting from this exposure. Most of these studies have been published in the peer-reviewed literature; the evaluation presented in this report is based primarily on peer-reviewed scientific papers. The previously available information was supplemented by the collection and chemical analysis of a limited number of tern eggs (12) from the Green Bay assessment area in 1996. This data collection effort, which was outlined in the NRDA Assessment Plan, is described in detail in Appendix B. Finally, the available information was evaluated using a weight-of-evidence approach (Chapter 7). The adverse effects determined to be "likely" to be caused by PCBs in Chapter 7 were considered to be injuries within the context of the injury determination.

8.2 INJURY DEFINITIONS

Chapter 3 described the types of adverse effects of PCBs on birds and discussed the relationship between these adverse effects and biological injury definitions at 43 CFR §11.62(f). Based on this information, relevant definitions of injury to avian resources of the assessment area include the following:

- ***Death.*** 43 CFR §11.62 (f)(4)(i). PCBs are known to cause embryo mortality, as manifested in reduced hatching success, reduced productivity, and embryo and chick mortality. This response is conceptually linked to the injury "reduced avian reproduction" described below.

- ***Physiological malfunctions/reduced avian reproduction.*** 43 CFR §11.62 (f)(4)(v)(B). PCBs have been found to cause reduced reproduction in various bird species. This reduced reproduction can be linked to death (through embryo or chick mortality), reduced hatching success, reduced egg fertility, reduced parental attentiveness, or other toxicological responses (see Chapter 3).

- ***Physical deformation.*** 43 CFR §11.62 (f)(4)(vi). PCBs can cause external deformations such as cross bills [43 CFR §11.62 (f)(4)(vi)(A)], skeletal deformities [43 CFR §11.62 (f)(4)(vi)(B)], and internal organ deformations [43 CFR §11.62 (f)(4)(vi)(C)].

- ***Tissue concentrations.*** 43 CFR §11.62 (f)(1)(ii-iii). Injury has occurred if concentrations of PCBs are sufficient to cause bird tissues to "exceed action or tolerance levels established under section 402 of the Food, Drug and Cosmetic Act, 21 U.S.C. 432, in edible portions of organisms" or "exceed levels for which an appropriate State health agency has issued directives to limit or ban consumption of such organism."

Injuries to birds in the assessment area are determined for each of these injury definitions in Section 8.4.

8.3 PATHWAY DETERMINATION

The purpose of the pathway determination phase is to identify the pathways by which avian resources come to be exposed to PCBs released into the assessment area.[1] As described in the Departmental regulations, pathways may be determined by demonstrating the presence of the hazardous substance in the pathway resources, or by using a model that demonstrates the routes

1. As noted previously, a separate report being prepared by the Trustees presents a complete evaluation of exposure pathways in the assessment area. The information contained in this chapter focuses on pathways by which birds are exposed to PCBs.

of exposure [43 CFR §11.63 (a)(2)]. Figure 8-1 presents the pathway diagram previously shown for avian resources in the assessment area. Table 8-1 demonstrates that PCBs have been detected at elevated concentrations in the various component pathway routes depicted in Figure 8-1, and that the spatial patterns of contamination are consistent with Fox River being the primary PCB source to the bay. Based on this information, it can be concluded that the presence of PCBs has been demonstrated in the various pathway resources that link PCB releases with avian resources.

The Green Bay Mass Balance Model also can be used to demonstrate PCB pathways to birds. The model is a multimillion dollar research effort to model the fate and transport of PCBs in the Fox River and Green Bay and their accumulation in the aquatic food chain (Connolly et al., 1992; DePinto et al., 1994). The model was constructed by numerous scientific and modeling experts from academia, government agencies, and private firms, and it has undergone extensive peer review. It provides a quantitative estimate of how PCBs move through the physical and biological compartments of Green Bay. It is based on scientific principles of PCB movement and accumulation, and was calibrated using extensive field-collected data. The model demonstrates that PCBs move through the system primarily as adherents to suspended sediment particles. Once in Green Bay, PCBs can enter the food chain through a variety of pathways, including biota ingestion of contaminated sediment and direct uptake from dissolved PCB phases in water.

Of the relevant pathway resources, the principal pathway of PCB exposure for assessment area birds is the dietary (biological) pathway. The food chain pathway is referred to as “indirect” exposure in the Departmental regulations [43 CFR §11.63(f)(2)]. Departmental regulations specify that “if indirect exposure to the biological resource has occurred . . . chemical analysis of free-ranging biological resources using one or more indicator species . . . may be performed” 43 CFR §11.63(f)(4)(ii). Thus, as demonstrated above, biological pathway determination is confirmed based both on chemical analysis of free-ranging biological resources, and on the use of a mass balance model that demonstrates the exposure routes.

In addition, Chapter 5 presented more detailed information that further confirms PCB dietary pathways to birds in the assessment area. This information included the following:

- As presented in Section 5.1.2, the diets of Forster’s, common, and Caspian terns were characterized based on known feeding behaviors and on examination of regurgitated pellets. Elevated concentrations of PCBs were measured in forage fish species that are consumed by terns. Monitoring of PCB uptake in tern chicks from hatching to fledging demonstrated that chicks reared on Kidney Island accumulated PCBs, demonstrating that chicks were being fed PCB-contaminated food and thus confirming the dietary pathway.

- As described in Section 5.2.2, the composition of cormorant diets is primarily forage fish. These prey items were shown to be contaminated with PCBs. Foraging areas were delineated, and cormorants were observed feeding in Green Bay in close proximity to their colonies. Cormorant stomachs were shown to contain fish prey that were contaminated

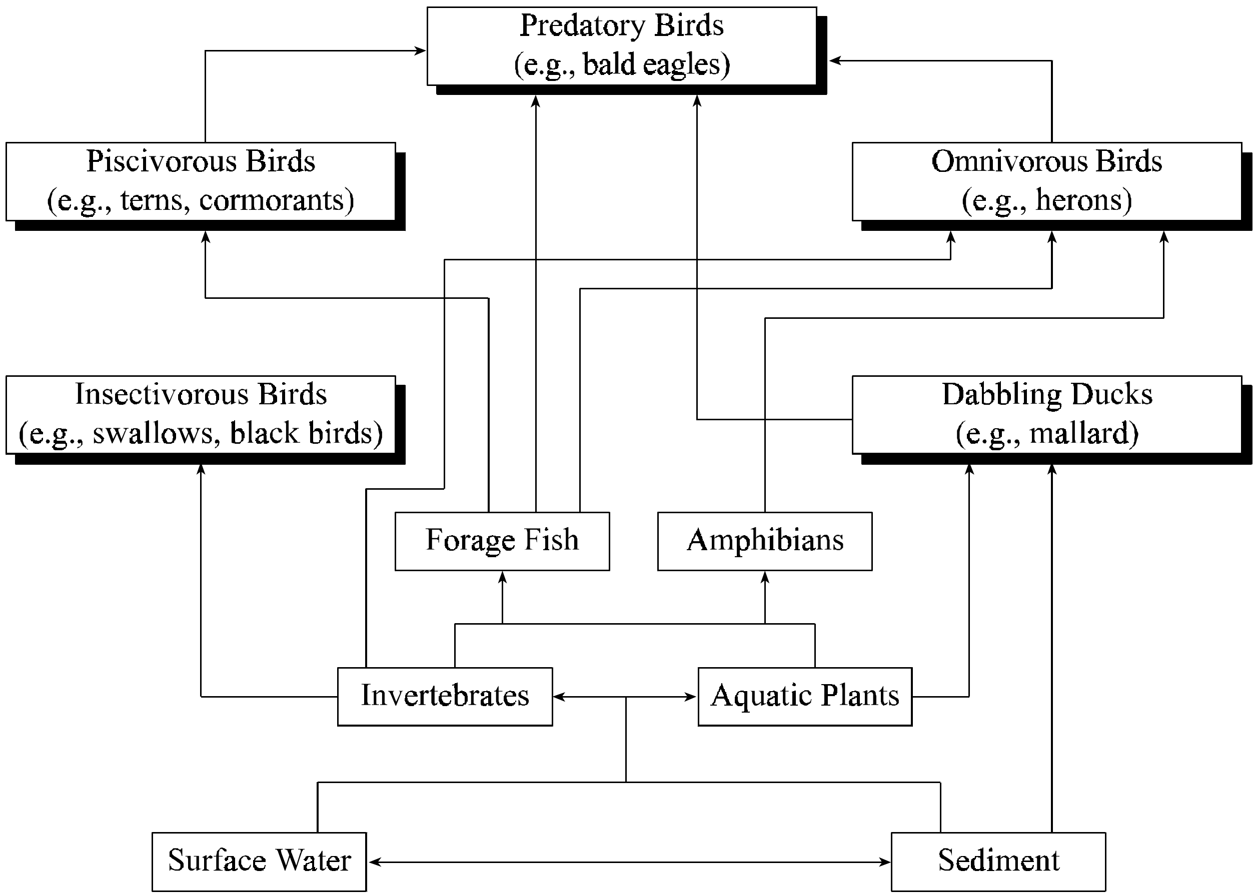


Figure 8-1. General PCB exposure pathways for assessment area birds.

> with PCBs. Also, cormorant PCB tissue residues were found to increase during the breeding season while the birds were nesting in Green Bay, confirming that they were exposed to PCBs in Green Bay.

The above information — including direct measurement of PCB exposures in bird tissue and in bird prey, detailed mass balance modeling, and site-specific biological observations — is concluded to have met the requirements for pathway determination.

8.4 CONCLUSIONS OF INJURY DETERMINATION TESTING AND SAMPLING

This section summarizes the conclusions derived from the weight of evidence evaluation of those studies that comprise the injury determination testing and sampling. The methods used to

Table 8-1
Examples of PCB Concentrations Measured in Assessment Area Pathway Resources

Pathway Resource	Location	PCB Concentration	Source
Sediment	Inner bay, east side	1,600 µg/kg dry weight (averaged over top 3 cm)	Manchester-Neesvig et al., 1996
Surface water	Fox River	21.7 ng/L dissolved (mean)	Connolly et al., 1992
	Inner bay, east side	8.5 ng/L dissolved (mean)	
	Inner bay, west side	4.2 ng/L dissolved (mean)	
	Middle bay, east side	1.7 ng/L dissolved (mean)	
	Middle bay, west side	1.8 ng/L dissolved (mean)	
	Outer bay	0.6 ng/L dissolved (mean)	
Phytoplankton (aquatic plants)	Bay-wide average	~4-12 µg/kg dry weight, depending on time of year	Connolly et al., 1992
Zooplankton (invertebrates)	Fox River	~600 µg/kg dry weight (mean)	Connolly et al., 1992
	Outer bay	~60 µg/kg dry weight (mean)	
Forage fish (alewife)	Fox River	2,100 µg/kg dry weight (mean)	Connolly et al., 1992
	Inner bay, east side	1,800 µg/kg dry weight (mean)	
	Inner bay, west side	1,400 µg/kg dry weight (mean)	
	Middle bay, east side	1,300 µg/kg dry weight (mean)	
	Middle bay, west side	680 µg/kg dry weight (mean)	
	Outer bay	520 µg/kg dry weight (mean)	

determine injuries to avian resources are consistent with those contained in the Departmental regulations for NRDA [43 CFR §11.64]. Specifically, the approach relies on the use of previously collected data as outlined in the assessment plan [43 CFR §11.64 (a)(2)] and therefore is cost-effective [43 CFR §11.64 (a)(3)(ii)]. Moreover, the various studies relied upon methods that were applied to “biological responses that have satisfied the acceptance criteria of Sec. 11.62(f)(2)” and applied approaches “that have been documented and are applicable to the biological response being tested” [43 CFR §11.64(f)(2)(I-ii)]. Most of the studies relied upon in the injury evaluation have been published in the peer-reviewed literature; therefore, the methods are appropriately documented and were deemed applicable.

The conclusions derived from the evaluation of the testing and sampling data indicate that avian resources of the Lower Fox River/Green Bay assessment area have been injured. Specifically, various fish-eating birds in the assessment area, including Forster’s terns, common terns, double-crested cormorants, and bald eagles have been injured as a result of exposure to PCBs. The injuries documented in the preceding chapters of this report include death [43 CFR §11.62

(f)(4)(I)] and reduced reproduction [43 CFR §11.62 (f)(4)(v)(B)],[2] as well as physical deformations [43 CFR §11.62 (f)(4)(vi)]. Waterfowl are also injured by exposure to PCBs in the assessment area. This injury comprises exceedences of tissue action or tolerance levels and waterfowl consumption advisories [43 CFR §11.62 (f)(1)(ii-iii)].

2. The injury categories "death" and "reduced avian reproduction" are effectively equivalent in this case. As noted in previous report chapters, available information suggests that mortality in assessment area birds is limited to bird embryos/chicks and this mortality contributes to reduced avian reproduction. Therefore, the two injury definitions are presented together.

CHAPTER 9
REFERENCES

Amundson, T.E. Undated. Environmental Contaminant Monitoring of Wisconsin Wild Game 1985-86. Report of Bureau of Wildlife Management, Wisconsin Department of Natural Resources, Madison.

Ankley, G.T., G.J. Niemi, K.B. Lodge, H.J. Harris, D.L. Beaver, D.E. Tillitt, T.R. Schwartz, J.P. Giesy, P.D. Jones, and C. Hagley. 1993. Uptake of planar polychlorinated biphenyls and 2,3,7,8-substituted polychlorinated dibenzofurans and dibenzo-*p*-dioxins by birds nesting in the lower Fox River and Green Bay, Wisconsin, USA. *Arch. Environ. Contam. Toxicol.* 24: 332-344.

Barron, M.G., H. Galbraith, and D. Beltman. 1995. Comparative reproductive and developmental toxicology of PCBs in birds. *Comp. Biochem. Physiol.* 112C: 1-14.

Barron, M.G., M. Anderson, D. Beltman, T. Podrabsky, W. Walsh, D. Cacela, J. Lipton, S.J. Teh, D.E. Hinton, J.T. Zelikoff, A.L. Dikkeboom, D.E. Tillitt, M. Holey, P. Bouchard, and N. Denslow. 1999. Association between PCBs, Liver Lesions, and Biomarker Responses in Adult Walleye (*Stizostedium vitreum*) Collected from Green Bay, Wisconsin. Prepared by Stratus Consulting Inc., Boulder, CO, for the U.S. Fish and Wildlife Service, Green Bay Field Office.

Bath, A.J. 1991. Kaukauna, Wisconsin Bald Eagle Nest Site — A Management Plan. Unpubl. Report. Stevens Point, WI.

Becker, P.H., S. Schuhmann, and C. Koepff. 1993. Hatching failure in common terns (*Sterna hirundo*) in relation to environmental chemicals. *Environ. Pollut.* 79: 207-213.

Belonger, B. 1983. Double-crested cormorant study results and recommendations. State of Wisconsin memorandum to C. Higgs dated December 23.

Best, D.A., W.W. Bowerman, T.J. Kubiak, S.R. Winterstein, S. Postupalsky, M.C. Shieldcastle, and J.P. Giesy. 1994. Reproductive impairment of bald eagles along the Great Lakes shorelines of Michigan and Ohio. In *Raptor Conservation Today*. B.U. Meyerburg and R.D. Chancellor (eds.), World Working Group on Birds of Prey and The Pica Press, East Sussex, Great Britain, pp. 697-701.

Beyer, W.N., L.J. Blus, C.J. Henny, and D. Audet. 1997. The role of sediment ingestion in exposing wood ducks to lead. *Ecotoxicology* 6: 181-186.

Beyer, W.N., D.J. Audet, A. Morton, J.K. Campbell, and L. Captain. 1998. Lead exposure of waterfowl ingesting Coeur d' Alene River Basin sediments. *J. Environ. Qual.* 27: 1533-1538.

Bird, D.M., P.H. Tucker, G.A. Fox, and P.C. Lague. 1983. Synergistic effects of Aroclor 1254 and mirex on the semen characteristics of American kestrels. *Arch. Environ. Contam. Toxicol.* 12: 633-640.

Bishop, C.A., D.V. Weseloh, N.M. Burgess, J. Struger, R.J. Norstrom, R. Turle, and K.A. Logan. 1992. *An Atlas of Contaminants in Eggs of Fish-Eating Colonial Birds of the Great Lakes (1970-1988).* Vols. I and II. Environment Canada. Canadian Wildlife Service. Technical Report Series No. 152.

Bosveld, A.T.C. 1995. Effects of Polyhalogenated Aromatic Hydrocarbons on Piscivorous Avian Wildlife. Doctoral thesis, University of Utrecht.

Bosveld, A.T.C. and M. Van den Berg. 1994. Effects of polychlorinated biphenyls (PCBs), dibenzo-p-dioxins (PCDDs), and dibenzofurans (PCDFs) on fish-eating birds. *Environ. Rev.* 2: 147-166.

Bosveld, A.T.C., J. Gardener, A.J. Murk, A. Brouwer, M. Van Kampen, E.H.G. Evers, and M. van den Berg. 1995. Effects of PCDDs, PCDFs and PCBs in common tern (*Sterna hirundo*) breeding in estuarine and coastal colonies in the Netherlands and Belgium. *Environ. Toxicol. Chem.* 14: 99-115.

Bowerman, W.W. 1993. Regulation of Bald Eagle (*Haliaeetus leucocephalus*) Productivity in the Great Lakes Basin: An Ecological and Toxicological Approach. PhD Dissertation, Michigan State University, East Lansing, 291 pp.

Bowerman, W.W., J.P. Giesy, D.A. Best, and V.J. Kramer. 1995. A review of factors affecting productivity of bald eagles in the Great Lakes region: Implications for recovery. *Environ. Health Perspect.* 103(Suppl 4): 51-59.

Bowerman, W.W., D.A. Best, T.J. Kubiak, M.S. Martell, S. Postupalsky, and J.P. Giesy, Jr. 1994a. The influence of environmental contaminants on bald eagle populations in the Laurentian Great Lakes, North America. In *Raptor Conservation Today,* B.U. Meyburg and R.D. Chancellor (eds.), World Working Group on Birds of Prey and The Pica Press, East Sussex, Great Britain, pp. 703-707.

Bowerman, W.W., T.J. Kubiak, J.B. Holt, D.E. Evans, R.J. Eckstein, and C.R. Sindelar. 1994b. Observed abnormalities in mandibles of nestling bald eagles. *Bull. Environ. Contam. Toxicol.* 53: 450-457.

Braune, B.M. and R.J. Norstrom. 1989. Dynamics of organochlorine compounds in herring gulls: III. Tissue distribution and bioaccumulation in Lake Ontario gulls. *Environ. Toxicol. Chem.* 8: 957-968.

Brewer, R., G.A. McPeak, and R.J. Adams. 1991. *The Atlas of Breeding Birds of Michigan.* Michigan State University Press, East Lansing.

Britton, W.M. and T.M. Huston. 1973. Influence of polychlorinated biphenyls in the laying hen. *Poult. Sci.* 32: 1620-1624.

Broley, C.L. 1958. The plight of the American bald eagle. *Audubon Mag.* 60: 162-163.

Brouwer, A. 1991. Role of biotransformation in PCB-induced alterations in vitamin A and thyroid hormone metabolism in laboratory and wildlife species. In *The Role of Enzymes in Regulating the Toxicity of Xenobiotics.* Reading University, Reading, PA, pp. 731-737.

Brunstrom, B. 1988. Sensitivity of embryos from duck, goose, herring gull, and various chicken breeds to 3,3',4',4'-tetrachlorobiphenyl. *Poultry Sci.* 67: 52-57.

Brunstrom, B. 1989. Toxicity of coplanar polychlorinated biphenyls in avian embryos. *Chemosphere* 19: 765-768.

Brunstrom, B. 1990. Mono-ortho-chlorinated chlorobiphenyls: Toxicity and induction of 7-ethoxyresorufin O-deethylase (EROD) activity in chick embryos. *Arch. Toxicol.* 64: 188-192.

Brunstrom, B. and L. Andersson. 1988. Toxicity and 7-ethoxyresorufin O-deethylase-inducing potency of coplanar polychlorinated biphenyls (PCBs) in chick embryos. *Arch. Toxicol.* 62: 263-266.

Brunstrom, B. and J. Lund. 1988. Differences between chick and turkey embryos in sensitivity to 3,3',4,4'-tetrachlorobiphenyl and in concentration/affinity of the hepatic receptor for 2,3,7,8-tetrachlorodibenzo-p-dioxin. *Comp. Biochem. Physiol.* 91C: 507-512.

Brunstrom, B. and L. Reutergårdh. 1986. Differences in sensitivity of some avian species to the embryotoxicity of a PCB, 3,3',4,4'-tetrachlorobiphenyl, injected into the eggs. *Environ. Pollut. (Series A)* 42: 37-45.

Bruzner, J. and W. DeVita. 1998. PCBs, DDE, and mercury in young-of-the-year littoral fishes from Green Bay, Lake Michigan. *J. Great Lakes Res.* 24(1): 83-92.

Burger, J. and M. Gochfeld. 1991. *The Common Tern: Its Breeding Biology and Social Behavior.* Columbia University Press. New York, NY.

Burness, G.P., R.D. Morris, and J.P. Bruce. 1994. Seasonal and annual variation in brood attendance, prey type delivered to chicks, and foraging patterns of male common terns (*Sterna hirundo*). *Can. J. Zool.* 72: 1243-1251.

Carlson, R.W. and R.T. Duby. 1973. Embryotoxic effects of three PCBs in the chicken. *Bull. Environ. Contam. Toxicol.* 9: 261-266.

Carr, R.A., R.L. Durfee, and A.G. McKay. 1977. PCBs Involvement in the Pulp and Paper Industry. Report prepared by Versar, Inc., Springfield, VA, for U.S. EPA, Washington, DC, February 25.

Chen, S.-W., P.J. Dziuk, and B.M. Francis. 1994. Effect of four environmental toxicants on plasma Ca and estradiol 17*B* and hepatic P450 in laying hens. *Environ. Toxicol. Chem.* 13: 789-796.

Choksi, N.Y., P.R.S. Kodavanti, and R.G. Booth. 1997. Effects of polychlorinated biphenyls (PCBs) on brain tyrosine hydroxylase activity and dopamine synthesis in rats. *Fundamen. Appl. Toxicol.* 39: 76-80.

Clark, K.E., L.J. Niles, and W. Stansley. 1998. Environmental contaminants associated with reproductive failure in bald eagle eggs in New Jersey. *Bull. Environ. Contam. Toxicol.* 61: 247-254.

Colborn, T. 1991. Epidemiology of Great Lakes bald eagles. *J. Toxicol. Environ. Health.* 33: 395-453.

Colborn, T., F.S. vom Saal, and A.M. Soto. 1993. Developmental effects of endocrine-disrupting chemicals in wildlife and humans. *Environ. Health Perspect.* 101: 378-384.

Connolly, J.F., T.F. Parkerton, J.D. Quadrine, S.T. Taylor, and A.J. Thumann. 1992. Development and Application of a Model of PCBs in the Green Bay, Lake Michigan Walleye and Brown Trout and their Food Webs: 200. Report prepared for the U.S. EPA Large Lakes Research Station, Grosse lle, MI. October 2.

Connor, K., K. Ramamoorthy, M. Moore, M. Mustain, I. Chen, S. Safe, T. Zacharewski, B. Gillesby, A. Joyeu, and P. Balaguer. 1997. Hydroxylated polychlorinated biphenyls (PCBs) as estrogens and antiestrogens: Structure-activity relationships. *Toxicol. Appl. Pharmacol.* 145: 111-123.

Courtney, P. and H. Blokpoel. 1980. Food and indicators of food availability for common terns on the Lower Great Lakes. *Can. J. Zool.* 58: 1318-1323.

Cramp, S. 1985. *Handbook of the Birds of Europe, the Middle East and North Africa. Birds of the Western Palearctic. Volume IV: Terns to Woodpeckers.* Oxford University Press. Oxford, U.K.

Cramp, S. and K.E.L. Simmons. 1977. *Handbook of the Birds of Europe, the Middle East and North Africa. Birds of the Western Palearctic. Vol I.* Oxford University Press, Oxford, U.K.

Curtis, J.T. 1959. *The Vegetation of Wisconsin.* University of Wisconsin Press, Madison.

Custer, C.M., T.W. Custer, P.D. Allen, K.L. Stromborg, and M.J. Melancon. 1998. Reproduction and environmental contamination in tree swallows nesting in the Fox River drainage and Green Bay, Wisconsin, USA. *Environ. Toxicol. Chem.* 17(9): 1786-1798.

Custer, T.W. and C. Bunck. 1992. Feeding flights of breeding double-crested cormorants at two Wisconsin colonies. *J. Field Ornithol.* 63: 303-311.

Custer, T.W. and C.M. Custer. 1995. Transfer and accumulation of organochlorines from black-crowned night heron eggs to chicks. *Environ. Toxicol. Chem.* 14: 533-536.

Custer, T.W., C.M. Custer, and K.L. Stromborg. 1997. Distribution of organochlorine contaminants in double-crested cormorant eggs and sibling embryos. *Environ. Toxicol. Chem.* 16: 1646-1649.

Custer, T.W., C.M. Custer, R.K. Hines, S. Gutreuter, K.L. Stromborg, P.D. Allen, and M.J. Melancon. In press. Organochlorine contaminant effects on double-crested cormorants nesting in Green Bay, Wisconsin. *Environ. Toxicol. Chem.*

Dahlgren, R.B. and R.L. Linder. 1971. Effects of polychlorinated biphenyls on pheasant reproduction, behavior, and survival. *J. Wildl. Manag.* 35: 315-319.

Dahlgren, R.B., R.L. Linder, and C.W. Carlson. 1972a. Polychlorinated biphenyls: Their effects on penned pheasants. *Environ. Health Perspect.* 1: 89-101.

Dahlgren, R.B., R.J. Bury, R.L. Linder, and R.F. Reidinger. 1972b. Residue levels and histopathology in pheasants given polychlorinated biphenyls. *J. Wildl. Manag.* 36: 524-533.

DePinto, J.V., R. Raghunathan, P. Sierzenga, X. Zhang, V.J. Bierman, P.W. Rodgers, and T.C. Young. 1994. Recalibration of GBTOX: An Integrated Exposure Model for Toxic Chemicals in Green Bay, Lake Michigan. Draft Final Report. Prepared for the U.S. EPA, Large Lakes and Rivers Research Branch, Grosse lle, MI. March 1.

Dolbeer, R.A. 1991. Migration patterns of double-crested cormorants east of the Rocky Mountains. *J. Field Ornithol.* 62: 83-93.

Donaldson, G.M., J.L. Shutt, and P. Hunter. 1999. Organochlorine contamination in bald eagle eggs and nestlings from the Canadian Great Lakes. *Arch. Environ. Contam. Toxicol.* 36: 70-80.

Dunstan, T.C. and J.F. Harper. 1975. Food habits of bald eagles in north-central Minnesota. *J. Wildl. Manag.* 39(1): 140-143.

Dykstra, C.J.R. and M.W. Meyer. 1996. Effects of Contaminants on Reproduction of Bald Eagles on Green Bay, Lake Michigan. Report to U.S. Fish and Wildlife Service and Wisconsin Department of Natural Resources.

Ehrlich, P.R., D.S. Dobben, and D. Wildeye. 1988. *The Birder's Handbook.* Simon and Schuster, New York.

Eisler, R. 1986. Polychlorinated Biphenyl Hazards to Fish, Wildlife, and Invertebrates: A Synoptic Review. Biological Report 85(1.7), Contaminant Hazard Reviews Report No. 7. U.S. Department of the Interior, Washington, DC.

Elliott, J.E., S.W. Kennedy, and A. Lorenzen. 1997. Comparative toxicity of polychlorinated biphenyls to Japanese quail (*Coturnix c. japonica)* and American kestrels *(Falco sparverius)*. *J. Toxicol. Environ. Health.* 51: 57-75.

Elliott, J.E., S.W. Kennedy, D.B. Peakall, and H. Won. 1990. Polychlorinated biphenyl (PCB) effects on hepatic mixed function oxidases and porphyria in birds. I. Japanese quail. *Comp. Biochem. Physiol.* 96C: 205-210.

Elliott, J.E., R.J. Norstrom, A. Lorenzen, L.E. Hart, H. Philibert, S.W. Kennedy, J.J. Stegeman, G.D. Bellward, and K.M. Cheng. 1996. Biological effects of polychlorinated dibenzo-p-dioxins, dibenzofurans, and biphenyls in bald eagle (*Haliaeetus leucocephalus*) chicks. *Environ. Toxicol. Chem.* 15: 782-793.

Environment Canada. 1991. Toxic Chemicals in the Great Lakes and Associated Effects. Minister of Supply and Services Canada.

Environment Canada. 1995. The rise of the double-crested cormorant on the Great Lakes: Winning the war against contaminants. Great Lakes Fact Sheet. Environment Canada, Canadian Wildlife Service, Burlington, Ontario.

Erdman, T.C. and J.P. Jacobs. 1991. Birds of Brown County. Richter Natural History Museum, Research Report No. 3. University of Wisconsin-Green Bay.

Erickson, M.D. 1997. *Analytical Chemistry of PCBs.* Lewis Publishers, New York.

Ewins, P.J., C.V. Weseloh, R.J. Norstrom, K. Legierse, H.J. Auman, and J.P. Ludwig. 1994. Caspian terns on the Great Lakes: Organochlorine contamination, reproduction, diet, and population changes, 1972-91. Canadian Wildlife Service Occasional Paper Number 85.

Fisher, L.J., R.F. Seegal, P.E. Ganey, I.N. Pessah, and P.R.S. Kodavanti. 1998. Symposium overview: Toxicity of non-coplanar PCBs. *Toxicol. Sci.* 41: 49-61.

Fowles, J.R., A. Fairbrother, K.A. Trust, and N.I. Kerkvliet. 1997. Effects of Aroclor 1254 on the thyroid gland, immune function, and hepatic cytochrome P450 activity in mallards. *Environ. Res.* 75: 119-129.

Fox, G.A. 1976. Eggshell quality: its ecological and physiological significance in a DDE-contaminated common tern population. *Wilson Bull.* 88: 459-477.

Fox, G.A., A.P. Gilman, D.B. Peakall, and F.W. Anderka. 1978. Behavioral abnormalities of nesting Lake Ontario herring gulls. *J. Wildl. Manag.* 42: 477-483.

Fox, G.A., S.W. Kennedy, R.J. Norstrom, and D.C. Wigfield. 1988. Porphyria in herring gulls: A biochemical response to chemical contamination of Great Lakes food chains. *Environ. Toxicol. Chem.* 7: 831-839.

Fox, G.A., B. Collins, E. Hayakawa, D.V. Weseloh, J.P. Ludwig, T.J. Kubiak, and T.C. Erdman. 1991. Reproductive outcomes in colonial fish-eating birds: A biomarker for developmental toxicants in the Great Lakes food chains. II. Spatial variation in the occurrence and prevalence of bill defects in young double-crested cormorants in the Great Lakes, 1979-1987. *J. Great Lakes Res.* 17: 158-167.

Fraser, G.S. 1994. Feeding and nesting behaviors of the Forster's tern on Lake Osakis, Minnesota. Masters Thesis to North Dakota State University, Fargo. April.

Friend, M. and D.O. Trainer. 1970. Polychlorinated biphenyl: Interaction with duck hepatic virus. *Science.* 170: 1314-1316.

Fry, D.M. 1995. Reproductive effects in birds exposed to pesticides and industrial chemicals. *Environ. Health Perspect.* 103(Suppl. 7): 165-171.

Garner, C.E., W.N. Jefferson, L.T. Burka, H.B. Matthews, and R.R. Newbold. 1999. In vitro estrogenicity of the catechol metabolites of selected polychlorinated biphenyls. *Toxicol. Appl. Pharmacol.* 154: 188-197.

Giesy, J.P., J.P. Ludwig, and D.E. Tillitt. 1994. Deformities in birds of the Great Lakes region: Assigning causality. *Environ. Sci. Technol.* 28: 128A-135A.

Gilbertson, M., T. Kubiak, J. Ludwig, and G. Fox. 1991. Great Lakes Embryo Mortality, Edema and Deformities Syndrome (GLEMEDS) in colonial fish-eating birds: similarity to chick-edema disease. *J. Toxicol. Environ. Health.* 33: 455-520.

Goldstein, J.A., J.D. McKinney, G.W. Lucier, P. Hickman, H. Bergman, and J.A. Moore. 1976. Toxicological assessment of hexa chloro bi phenyl isomers and 2,3,7,8 tetra chloro di benzo furan in chicks. Part 2: Effects of drug metabolism and porphyrin accumulation. *Toxicol. Appl. Pharmacol.* 36(1): 81-92.

Gould, J.C., K.R. Cooper, and C.G. Scanes. 1997. Effects of polychlorinated biphenyl mixtures and three specific congeners on growth and circulating growth-related hormones. *Gen. Compar. Endocrinol.* 106: 221-230.

Grau, C.R. 1996. Nutritional needs for egg formation in the shag (*Phalacrocorax aristotelis*). *Ibis.* 138: 756-764.

Grier, J.W. 1972. Current status of the bald and golden eagles in North America: A summary of information and opinions from field researchers. In Transactions of the 36th Federal-Provincial Wildlife Conference, pp. 76-78.

Grier, J.W. 1982. Ban of DDT and subsequent recovery of reproduction in bald eagles. *Science* 212: 1232-1235.

Guillette, L.J., Jr., D.A. Crain, A.A. Rooney, and D.B. Pickford. 1995. Organization versus activation: The role of endocrine disrupting contaminants (EDCs) during embryonic development in wildlife. *Environ. Health Perspect.* 103(Suppl. 7): 157-164.

Harris, H.J., T.C. Erdman, G.T. Ankley, and K.B. Lodge. 1993. Measures of reproductive success and polychlorinated biphenyl residues in eggs and chicks of Forster's terns on Green Bay, Lake Michigan, Wisconsin — 1988. *Arch. Environ. Contam. Toxicol.* 25: 304-314.

Hart, C.A., M.E. Hahn, I.C.T. Nisbet, M.J. Moore, S.W. Kennedy, and D.M. Fry. 1998. Feminization in common terns (*Sterna hirundo*): Relationship to dioxin equivalents and estrogenic compounds. *Mar. Environ. Res.* 46: 174-175.

Hart, L.E., K.M. Cheng, P.E. Whitehead, R.M. Shah, R.J. Lewis, S.R. Ruschkowski, R.W. Blair, D.C. Bennett, S.M. Bandiera, R.J. Norstrom, and G.D. Bellward. 1991. Dioxin contamination and growth and development in great blue heron embryos. *J. Toxicol. Environ. Health.* 32: 331-344.

Haseltine, S.D. and R.M. Prouty. 1980. Aroclor 1242 and reproductive success of adult mallards (*Anas platyrhynchos*). *Environ. Res.* 23: 29-34.

Haseltine, S.D., Z.G.H Heinz, W.L. Reichel, and J.F. Moore. 1981. Organochlorine and metal residues in eggs of waterfowl nesting on islands in Lake Michigan off Door County, Wisconsin, 1977-78. *Pest. Monitor. J.* 15: 90-97.

Hatch, J.J. 1995. Changing populations of double-crested cormorants. *Colon. Waterbirds.* 18: 8-24.

Heath, R.G., J.W. Span, E.F. Hill, and J.F. Kreitzer. 1972. Comparative Dietary Toxicities of Pesticides to Birds. Special Scientific Report Wildlife No. 152, U.S. Department of Interior, Washington, DC.

Heinz, G.H., E.F. Hill, and J.F. Contrera. 1980. Dopamine and norepinephrine depletion in ring doves fed DDE, dieldrin, and Aroclor 1254. *Toxic. Appl. Pharmacol.* 53: 75-82.

Heinz, G.H., T.C. Erdman, S.D. Haseltine, and C. Stafford. 1985. Contaminant levels in colonial waterbirds from Green Bay and Lake Michigan, 1975-80. *Environ. Monitor. Assess.* 5: 223-236.

Heinz, G.H., S.D. Haseltine, W.L. Reichel, and G.L. Hensler. 1983. Relationships of environmental contaminants to reproductive success in red-breasted mergansers (*Mergus serrator*) from Lake Michigan. *Environ. Pollut. Ser. A.* 32: 211-232.

Heinz, G.H., D.S. Miller, B.J. Ebert, and K.L. Stromborg. 1994. Declines in organochlorines in eggs of red-breasted mergansers from Lake Michigan, 1977-1978 versus 1990. *Environ. Monitor. Assess.* 33: 175-182.

Helander, B., M. Olsson, and L. Reutergårdh. 1982. Residue levels of organochlorine and mercury compounds in unhatched eggs and the relationships to breeding success in white-tailed sea eagles *Haliaeetus albicilla* in Sweden. *Holarct. Ecol.* 5: 349-366.

Helander, B., A. Olsson, A. Bignert, K. Litzén, L. Asplund, and A. Bergman. 1998. Abnormal dehydration in eggs of the white-tailed sea eagle (*Haliaeetus albicilla*) in Sweden — a persisting effect from high exposure to organochlorines? *Organohal. Comp.* 39: 423-426.

Henshel, D.S. 1998. Developmental neurotoxic effects on dioxin and dioxin-like compounds on domestic and wild avian species. *Environ. Toxicol. Chem.* 17: 88-98.

Henshel, D.S., J.W. Martin, R.J. Norstrom, J. Elliot, K.M. Cheng, and J. DeWitt. 1997. Morphometric brain abnormalities in double-crested comorant chicks exposed to polychlorinated dibenzo-p-dioxins, dibenzofurans, and biphenyls. *J. Great Lakes Res.* 23: 11-26.

Hobson, K.A., R.W. Knapton, and W. Lysack. 1989. Population, diet and reproductive success of double-crested cormorants breeding on Lake Winnipegosis, Manitoba, in 1987. *Colon. Waterbirds.* 12: 191-197.

Hoffman, D.J., C.P. Rice, and T.J. Kubiak. 1996a. PCBs and dioxins in birds. Chapter 7 in *Environmental Contaminants in Wildlife,* W.N. Beyer, G.H. Heinz, A.W. Redmon-Norwood, eds. Lewis Publishers, Boca Raton, FL.

Hoffman, D.J., G.J. Smith, and B.A. Rattner. 1993. Biomarkers of contaminant exposure in common terns and black-crowned night herons in the Great Lakes. *Environ. Toxicol. Chem.* 17: 1095-1103.

Hoffman, D.J., M.J. Melancon, J.D. Eisemann, and P.N. Klein. 1995. Comparative developmental toxicity of planar PCB congeners by egg injection. In *Proceedings, 2nd SETAC World Congress,* Vancouver, BC, Canada, November 5-9, p. 207.

Hoffman, D.J., M.J. Melancon, J.D. Eisemann, and P.N. Klein. 1998. Comparative developmental toxicity of planar PCB congeners in chickens, American kestrels, and common terns. *Environ. Toxicol. Chem.* 17: 747-757.

Hoffman, D.J., B.A. Rattner, L. Sileo, D. Docherty, and T. Kubiak. 1987. Embryotoxicity, teratogenicity, and aryl hydrocarbon hydroxylase activity in Forster's terns on Green Bay, Lake Michigan. *Environ. Res.* 42: 176-184.

Hoffman, D.J., B.A. Rattner, C.M. Bunck, A. Krynitsky, H.M. Ohlendorf, and R.W. Lowe. 1986. Association between PCBs and lower embryonic weight in black-crowned night herons in San Francisco Bay. *J. Toxicol. Environ. Health.* 19: 383-391.

Hoffman, D.J., M.J. Melancon, P.N. Klein, C.P. Rice, J.D. Eisemann, R.K. Hines, J.W. Spann, and G.W. Pendleton. 1996b. Developmental toxicity of PCB 126 (3,3',4,4',5-pentachlorobiphenyl) in nestling American kestrels (*Falco sparverius*). *Fund. Appl. Toxicol.* 34: 188-200.

Howe, R.W., A.T. Wolf, and T.C. Erdman. 1993. A Study of Wintering Bald Eagles on the Fox River of Northeast Wisconsin. Report to REPAP, Wisconsin, Inc.

Howell, J.C. 1963. The 1961 status of some bald eagle nest sites in east-central Florida. *The Auk* 79: 716-718.

Hughes, K.D., D.V. Weseloh, and B.M. Braune. 1998. The ratio of DDE to PCB concentrations in Great Lakes herring gull eggs and its use in interpreting contaminants data. *J. Great Lakes Res.* 24(1): 12-31.

Jansen, H.T., P.S. Cooke, J. Porcelli, T.-C. Liu, and L.G. Hansen. 1993. Estrogenic and antiestrogenic actions of PCBs in the female rat: In vitro and in vivo studies. *Repro. Toxicol.* 7: 237-248.

Janz, D.M. and G.D. Bellward. 1996. In ovo 2,3,7,8-tetrachloro-p-dioxin exposure in three avian species. 1. Effects on thyroid hormones and growth during the perinatal period. *Toxicol. Appl. Pharmacol.* 139: 281-291.

Jefferies, D.J. and J.L.F. Parslow. 1976. Thyroid changes in PCB-dosed guillemots and their indication of one of the mechanisms of action of these materials. *Environ. Pollut*. 10: 295-311.

Jones, P.D., J.P. Giesy, J.L. Newstead, D.A. Verbrugge, D.L. Beaver, G.T. Ankley, D.E. Tillitt, K.B. Lodge, and G.J. Niemi. 1993. 2,3,7,8-Tetrachlorodibenzo-*p*-dioxin equivalents in tissues of birds at Green Bay, Wisconsin, USA. *Arch. Environ. Contam. Toxicol.* 24: 345-354.

Kennedy, S.W., A. Lorenzen, and R.J. Norstrom. 1996a. Chicken embryo hepatocyte bioassay for measuring cytochrome P4501A-based 2,3,7,8-tetrachlorodibenzo-p-dioxin equivalent concentrations in environmental samples. *Environ. Sci. Technol.* 30: 706-715.

Kennedy, S.W., A. Lorenzen, S.P. Jones, M.E. Hahn, and J.J. Stegeman. 1996b. Cytochrome P4501A induction in avian hepatocyte cultures: A promising approach for predicting the sensitivity of avian species to toxic effects of halogenated aromatic hydrocarbons. *Toxicol. Appl. Pharmacol.* 141: 214-230.

Kiff, L.F. 1980. Historical changes in resident populations of California islands raptors. In *The California Islands: Proceedings of a Multidisciplinary Symposium,* D.M. Power, ed. Santa Barbara Natural History Museum, Santa Barbara, CA. pp. 651-673.

King, K.A., T.W. Custer, and J.S. Quinn. 1991. Effects of mercury, selenium, and organochlorine contaminants on reproduction of Forster's terns and black skimmers nesting in a contaminated Texas bay. *Arch. Environ. Contam. Toxicol.* 20: 32-40.

Korach, K.S., P. Sarver, K. Chae, J.A. McLachlan, and J.D. McKinney. 1987. Estrogen receptor-binding activity of polychlorinated hydrodroxybiphenyls: Conformationally restricted structural probes. *Mol. Pharmacol.* 33: 120-126.

Kubiak, T.J. and D.A. Best. 1991. Wildlife Risks Associated with Passage of Contaminated, Anadromous Fish at Federal Energy Regulatory Commission Licensed Dams in Michigan. U.S. Fish and Wildlife Service Report, East Lansing, MI.

Kubiak, T.J., H.J. Harris, L.M. Smith, T.R. Schwartz, D.L. Stalling, J.A. Trick, L. Sileo, D.E. Docherty, and T.C. Erdman. 1989. Microcontaminants and reproductive impairment of the Forster's tern on Green Bay, Lake Michigan. *Arch. Environ. Contam. Toxicol.* 18: 706-727.

Lamparski, L.L., T.J. Nestrick, and S.S. Cutie. 1990. The impact on the environment of airborne particulate matter from the eruption of Mount Saint Helens in May 1980. In *Emissions from Combustion Processes: Origin, Measurement, Control*. R. Clement and R. Kager (eds.). Lewis Publishers, Chelsea, MI.

Langenberg, J. 1990. Preliminary Summary of Pathology Findings for 1989. Report to U.S. Fish and Wildlife Service, Green Bay Field Office.

Langham, N.P.E. 1972. Chick survival in terns (*Sterna spp.*) with particular reference to the common tern. *J. Anim. Ecol.* 41: 385-395.

Larson, J.M., W.H. Karasov, L. Sileo, K.L. Stromborg, B.A. Hanbidge, J.P. Giesy, D.E. Tillitt, and D. Verbrugge. 1996. Reproductive success, developmental anomalies, and environmental contaminants in double-crested cormorants (*Phalacrocorax auritus*). *Environ. Toxicol. Chem.* 15: 553-559.

Lincer, J.L. and D.B. Peakall. 1970. Metabolic effects of polychlorinated biphenyls in the American kestrel. *Nature* 228: 783-784.

Lorenzen, A., J.L. Shutt, and S.W. Kennedy. 1997. Sensitivity of common tern (*Sterna hirundo*) embryo hepatocyte cultures to CYP1A induction and porphyrin accumulation by halogenated aromatic hydrocarbons and common tern egg extracts. *Arch. Environ. Comtam. Toxicol.* 32: 126-134.

Ludwig, J.P. 1965. Biology and structure of the Caspian tern (*Hydroprogne caspia*) population of the Great Lakes from 1896-1964. *Bird Banding* 36: 217-233.

Ludwig, J.P. 1984. Decline, resurgence and population dynamics of Michigan and Great Lakes double-crested cormorants. *Jack Pine Warbler* 62: 91-102.

Ludwig, J.P. and M.E. Ludwig. Undated report a. Food Habits of the Double-Crested Cormorant in Michigan, 1986-1988: Implications for the Acquisition of Toxic Contaminants from Fish by Breeding Females. Michigan Department of Natural Resources.

Ludwig, J.P. and M.P. Ludwig. Undated report b. Productivity and Deformities in Chicks of the Michigan Great Lakes' Colonies of Double-Crested Cormorants (*Phalacrocorax auritus*) and Caspian Terns (*Hydroprogne caspia*). Michigan Department of Natural Resources.

Ludwig, J.P., H.J Auman, D.V. Weseloh, G.A. Fox, J.P. Giesy, and M.E. Ludwig. 1995. Evaluation of the effects of toxic chemicals in Great Lakes cormorants: Has causality been established? *Colon. Waterbirds* 18: 60-69.

Ludwig, J.P., H. Kurita-Matsuba, H.J. Auman, M.E. Ludwig, C.L. Summer, J.P. Giesy, D.E. Tillitt, and P.D. Jones. 1996. Deformities, PCBs and TCDD-equivalents in double-crested cormorants (*Phalacrocorax auritus*) and Caspian terns (*Hydroprogne caspia*) of the upper Great Lakes 1986-1991: Testing a cause-effect hypothesis. *J. Great Lakes Res.* 22: 172-197.

Ludwig, J.P., J.P. Giesy, C.L. Summer, W. Bowerman, S. Heaton, R. Aulerich, S. Bursian, H.J. Auman, P.D. Jones, L.L. Williams, D.E. Tillitt, and M. Gilbertson. 1993. A comparison of water quality criteria in the Great Lakes basin based on human and wildlife health. *J. Great Lakes Res.* 19: 789-807.

Manchester-Neesvig, J.B., A.W. Andren, and D.W. Edgington. 1996. Patterns of mass sedimentation and of deposition of sediment contaminated by PCBs in Green Bay. *J. Great Lakes Res.* 22(2): 444-462.

Matteson, S.W. 1988. Wisconsin Common Tern Recovery Plan. Wisconsin Endangered Resources Report 41. Bureau of Endangered Resources, Wisconsin Department of Natural Resources, Madison.

Matteson, S.W. 1993. Wisconsin Caspian Tern Recovery Plan. Bureau of Endangered Resources, Wisconsin Department of Natural Resources. Madison.

McCarty, J.P. and A.L. Secord. 1999. Nest-building behavior in PCB-contaminated tree swallows. *The Auk* 116: 55-63.

McKinney, J.D., J. Fawkes, S. Jordan, K. Chae, S. Oatley, R.E. Coleman, and W. Briner. 1985. 2,3,7,8-Tetrachlorodibenzo-p-dioxin (TCDD) as a potent and persistent thyroxine agonist: A mechanistic model for toxicity based on molecular reactivity. *Environ. Health Perspect.* 61: 41-53.

Miller, S. 1987. Wildlife Management's procedures for issuing an advisory — wildlife consumption advisory. Memorandum from S. Miller (State of Wisconsin Bureau of Wildlife Management) to J.R. Huntoon. April 15.

Mora, M.A., H.J. Auman, J.P. Ludwig, J.P. Giesy, D.A. Verbrugge, and M.E. Ludwig. 1993. Polychlorinated biphenyls and chlorinated insecticides in plasma of Caspian terns: Relationships with age, productivity, and colony site tenacity in the Great Lakes. *Arch. Environ. Contam. Toxicol.* 24: 320-331.

Mossman, M.J. 1988. Wisconsin Forster's Tern Recovery Plan. Bureau of Endangered Resources, Wisconsin Department of Natural Resources. Madison.

Murk, A.J., A.T.C. Bosveld, M. van den Berg, and A. Brouwer. 1994. Effects of polyhalogenated aromatic hydrocarbons (PHAHs) on biochemical parameters in chicks of the common tern (*Sterna hirundo*). *Aquat. Toxicol.* 30: 91-115.

Neuman, J., D.L. Pearl, P.J. Ewins, R. Black, D.V. Weseloh, M. Pike, and K. Karwowski. 1997. Spatial and temporal variation in the diet of double-crested cormorants (*Phalacrocorax auritus*) breeding on the lower Great Lakes in the early 1990s. *Can. J. Fish. Aquat. Sci.* 54: 1569-1584.

Nikolaidis, E., B. Brunstrom, and L. Dencker. 1988. Effects of TCDD congeners 3,3',4,4'-tetrachlorobiphenyl and 3,3',4,4'-tetrachloroazoxybenzene on lymphoid development in the bursa fabricus of the chick embryo. *Toxicol. Appl. Pharmacol.* 92: 315-323.

Nisbet, I.C.T. 1989. Organochlorines, reproductive impairment and declines in bald eagle *Haliaeetus leucocephalus* populations: Mechanisms and dose-response mechanisms. In *Raptors in the Modern World*, B.-U. Meyburg and R.D. Chancellor, eds., World Working Group on Birds of Prey and Owls, Berlin, pp. 483-489.

Nisbet, I.C.T. and R.W. Risebrough. 1994. Relationship of DDE to productivity of bald eagles in California and Arizona, USA. In *Raptor Conservation Today*. B.U. Meyerburg and R.D. Chancellor, eds., World Working Group on Birds of Prey and The Pica Press, East Sussex, Great Britain.

Nisbet, I.C.T., D.M. Fry, J.J. Hatch, and B. Lynn. 1996. Feminization of male common tern embryos is not correlated to specific PCB congeners. *Bull. Environ. Contam. Toxicol.* 57: 895-901.

Nosek, J.A., J.R. Sullivan, S.R. Craven, A. Gendron-Fitzpatrick, and R.E. Peterson. 1993. Embryotoxicity of 2,3,7,8-tetrachloro-p-dioxin in the ring-necked pheasant. *Environ. Toxicol. Chem.* 12: 1215-1222.

Oliver, B.G. and A.J. Niimi. 1988. Trophodynamic analysis of polychlorinated biphenyl congeners and other chlorinated hydrocarbons in the Lake Ontario ecosystem. *Environ. Sci. Technol.* 22: 388-397.

Olsson, A., B. Helander, A. Bignert, K. Litzén, L. Asplund, and Å. Bergman. 1998. Is PCB responsible for embryo toxicity in white-tailed sea eagle (*Haliaeetus albicilla*) from the Swedish Baltic coast? *Organohal. Comp.* 39: 17-20.

Peakall, D.B. and J.L. Lincer. 1996. Do PCBs cause eggshell thinning? *Environ. Pollut.* 91: 127-129.

Peakall, D.B. and M.L. Peakall. 1973. Effects of a polychlorinated biphenyl on the reproduction of artificially and naturally incubated dove eggs. *J. Appl. Ecol.* 10: 863-868.

Peakall, D.B., J.L. Lincer, and S.E. Bloom. 1972. Embryonic mortality and chromosomal alterations caused by Aroclor 1254 in ring doves. *Environ. Health Perspect.* 1: 103-104.

Peden-Adams, M., K. Alonso, C. Godard, S. Skipper, W. Mashburn, J. Hoover, C. Charbonneau, D. Henshel, and R. Dickerson. 1998. Effects of environmentally relevant concentrations of 2,3,7,8-TCDD on domestic chicken immune function and CYP450 activity: F_1 generation and egg injection studies. *Chemosphere* 37: 1923-1939.

Pekarik, C., D.V. Weseloh, G.C. Barrett, M. Simon, C.A. Bishop, and K.E. Pettit. 1998. *An Atlas of Contaminants in Eggs of Fish-Eating Colonial Birds of the Great Lakes (1993-1997). Vol. I. Accounts by Location.* Environment Canada. Canadian Wildlife Service. Technical Report Series No. 321.

Petersen, R.E., H.M. Theobald, and G.L. Kimmel. 1993. Development and reproductive toxicity of dioxins and related compounds: Cross-species comparisons. *Crit. Rev. Toxicol.* 23: 283-335.

Pettit, K.E., C.A. Bishop, D.V. Weseloh, and R.J. Norstrom. 1994. An Atlas of Contaminants in Eggs of Fish-Eating Colonial Birds of the Great Lakes (1989-1992). Environment Canada. Canadian Wildlife Service. Technical Report Series No. 193.

Platonow, N.S. and H.S. Funnell. 1971. Anti-androgenic-like effect of polychlorinated biphenyls in cockerels. *Veterin. Record*. 83: 109-110.

Platonow, N.S. and B.S. Reinhart. 1973. The effects of polychlorinated biphenyls (Aroclor 1254) on chicken egg production, fertility and hatchability. *Can. J. Comp. Med.* 37: 341-346.

Postupalsky, S. 1971. Toxic Chemicals and Declining Bald Eagle and Cormorants in Ontario. Canadian Wildlife Service, Petic. Sec., MS Rep. 20. 70 pp.

Postupalsky, S. 1985. The bald eagles return. *Natural History* 87: 62-63.

Powell, D.C., R.J. Aulerich, J.C. Meadows, D.E. Tillitt, J.P. Giesy, K.L. Stromborg, and S.J. Bursian. 1996. Effects of 3,3',4,4',5-pentachlorobiphenyl (PCB 126) and 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) injected into the yolks of chicken (*Gallus domesticus*) eggs prior to incubation. *Arch. Environ. Contam. Toxicol.* 31: 404-409.

Powell, D.C., R.J. Aulerich, J.C. Meadows, D.E. Tillitt, J.F. Powell, J.C. Restum, K.L. Stromborg, J.P. Giesy, and S.J. Bursian. 1997. Effects of 3,3',4,4',5-pentachlorobiphenyl (PCB 126), 2,3,7,8-tetrachlordibenzo-*p*-dioxin (TCDD), or an extract derived from field-collected cormorant eggs injected into double-crested cormorant (*Phalacrocorax auritus*) eggs. *Environ. Toxicol. Chem.* 16: 1450-1455.

Prestt, I., D.J. Jefferies, and N.W. Moore. 1970. Polychlorinated biphenyls in wild birds in Britain and their avian toxicity. *Environ. Pollut.* 1: 3-26.

Rattner, B.A., M.J. Melancon, T.W. Custer, R.L. Hothem, K.A. King, L.J. LeCaptain, J.W. Spann, B.R. Woodin, and J.L. Stegeman. 1993. Biomonitoring environmental contamination with pipping black-crowned night heron embryos: Induction of cytochrome P450. *Environ. Toxicol. Chem.* 12: 1719-1732.

Rattray, K.J. 1997. The nesting success and PCB burdens of yellow-headed blackbirds found in diked marshes along Green Bay, Lake Michigan. Masters Thesis to University of Wisconsin, Green Bay. December.

Robbins, S.D. 1991. *Wisconsin Birdlife. Population and Distribution Past and Present.* The University of Wisconsin Press, Madison.

Rodman, L.E., S.I. Shedlofsky, A. Manschreck, M. Puttman, A.T. Swim, and L.W. Robertson. 1991. Differential potency of atropisomers of polychlorinated biphenyls on cytochrome P450 induction and uroporphyrin accumulation in the chick embryo hepatocyte culture. *Biochem. Pharmacol.* 41: 915-922.

Ross, P.S. and C.V. Weseloh. 1988. A Study of the Development, Occurrence and Severity of Deformities in Double-Crested Cormorants on the Canadian Great Lakes and Lake Winnipegosis, 1988. Unpublished report of Canadian Wildlife Service.

Rottiers, D.V. and R.M. Tucker. 1982. Proximate composition and caloric content of 8 Lake Michigan fishes. USFWS Technical Paper Number 108.

Rozemeijer, M.J.C., J.P. Boon, C. Swennen, A. Brouwer, and A.J. Murk. 1995. Dioxin type and mixed type induction of the cytochrome P-450 system of common eider ducklings (*Somateria mollissima*) by PCBs: With indications for biotransformation. *Aquat. Toxicol.* 32: 93-113.

Ryckman, D.P., D.V. Weseloh, P. Hamr, G.A. Fox, B. Collins, P.J. Ewins, and R.J. Norstrom. In press. Spatial and temporal trends in organochlorine contamination and bill deformities in double-crested cormorants (*Phalacrocorax auritus*) from the Canadian Great Lakes. *Env. Monitor. Assess.*

Safe, S. 1994. Polychlorinated biphenyls (PCBs): Environmental impact, biochemical and toxic responses, and implications for risk assessment. *Crit. Rev. Toxicol.* 24: 87-149.

Salt, G.W. and D.E. Willard. 1971. The hunting behavior and success of Forster's tern. *Ecology* 52: 989-998.

Schulz, D.E., G. Petrick, and J.C. Duinker. 1989. Complete characterization of polychlorinated biphenyl congeners in commercial Aroclors and Clophen mixtures by multidimensional gas chromatography-electron capture detection. *Environ. Sci. Technol.* 23: 852-859.

Scott, M.L. 1977. Effects of PCBs, DDT, and mercury compounds in chickens and Japanese quail. *Feder. Proceed.* 36: 1888-1893.

Sheffield, S.R., J.M. Matter, B.A. Rattner, and P.D. Guiney. 1998. Fish and wildlife species as sentinels of environmental endocrine disruptors. Chapter 17 in *Principles and Processes for Evaluating Endocrine Disruption in Wildlife,* R.J. Kendall (ed). SETAC Press, Pensacola, FL, pp. 369-430.

Smith, L.M., T.R. Schwartz, and K. Feltz. 1990. Determination and occurrence of AHH-active polychlorinated biphenyls, 2,3,7,8-tetrachloro-p-dioxin and 2,3,7,8-tetrachlorodibenzofuran in Lake Michigan sediment and biota. The question of their relative toxicological significance. *Chemosphere* 21(9): 1063-1085.

Spear, P.A., D.H. Bourbonnais, D.B. Peakall, and T.W. Moon. 1989. Dove reproduction and retinoid (vitamin A) dynamics in adult females and their eggs following exposure to 3,3',4,4'-tetrachlorobiphenyl. *Can. J. Zool.* 67: 908-913.

Sprunt, A., IV. 1963. Bald eagles aren't producing enough young. *Audubon Mag.* 65: 32-35.

Sprunt, A., IV and F.J. Ligas. 1966. Audubon Bald Eagle Studies — 1960-1965. In *Proceedings of the 62nd Annual Convention of the National Audubon Society,* Sacramento, CA.

Sprunt, A., IV, W.B. Robertson, Jr., S. Postupalsky, R.J. Hensel, C.E. Knoder, and F.J. Ligas. 1973. Comparative productivity of six bald eagle populations. In *Transactions of the 38th North American Wildlife & Natural Resources Conference,* pp. 98-106.

Stickel, W.H., L.F. Stickel, R.A. Dyrland, and D.L. Hughes. 1984. Aroclor 1254® residues in birds: Lethal levels and loss rates. *Arch. Environ. Contam. Toxicol.* 13: 7-13.

Stickel, L.F., N.J. Chura, P.A. Stewart, C.M. Menzie, R.M. Prouty, and W.L. Reichel. 1966. Bald eagle-pesticide relations. In *Transactions of the 31st North American Wildlife & Natural Resources Conference,* pp. 190-200.

Struger, J. and D.V. Weseloh. 1985. Great Lakes Caspian terns: Egg contaminants and biological implications. *Colon. Waterbirds* 8: 142-148.

Temple, S.A. and J.R. Cary. 1987. *Wisconsin Birds. A Seasonal and Geographical Guide.* The University of Wisconsin Press, Madison.

Thiel, D.A., S.G. Martin, J.W. Duncan, M.J. Lemke, W.R. Lance, and R.E. Peterson. 1988. Evaluation of the effects of dioxin-contaminated sludges on wild birds. In *Proceedings, 1988 TAPPI Environmental Conference,* Norcross, GA, pp. 487-506.

Tillitt, D.E., T.J. Kubiak, G.T. Ankley, and J.P. Giesy. 1993. Dioxin-like toxic potency in Forster's tern eggs from Green Bay, Lake Michigan, North America. *Chemosphere* 26(11): 2079-2084.

Tillitt, D.E., G.T. Ankley, J.P. Giesy, J.P. Ludwig, H. Kurita-Matsuba, D.V. Weseloh, P.S. Ross, C. Bishop, L. Sileo, K.L. Stromborg, J. Larson, and T. Kubiak. 1992. Polychlorinated biphenyl residues and egg mortality in double-crested cormorants from the Great Lakes. *Environ. Toxicol. Chem.* 11: 1281-1288.

Tilson, H.A., J.A. Jacobs, and W.J. Rogan. 1990. Polychlorinated biphenyls and the developing nervous system: Cross-species comparisons. *Neurotoxicol. Teratol.* 12: 239-248.

Tori, G.M. and T.J. Peterle. 1983. Effects of PCBs on mourning dove courtship behavior. *Bull. Environ. Contam. Toxicol.* 30: 44-49.

Trick, J.A. 1982. Reproductive success and foraging success as related to populations of Forster's terns (*Sterna forsteri*) on Green Bay, Wisconsin. MS Thesis, University of Wisconsin, Green Bay.

Tumasonis, C.F., B. Bush, and F.D. Baker. 1973. PCB levels in egg yolk associated with embryonic mortality and deformity of hatched chicks. *Arch. Environ. Contam. Toxicol.* 1: 312-324.

USDI. 1974. United States estimated to have 1,000 pairs of bald eagles in lower 48 states. U.S. Department of Interior, U.S. Fish and Wildlife Service, News Release, 27 February.

U.S. EPA. 1980. Ambient Water Quality Criteria for Polychlorinated Biphenyls. EPA 440/5-80-068. U.S. Environmental Protection Agency, Office of Water, Washington, DC.

U.S. EPA. 1998a. Guidelines for Ecological Risk Assessment. Final. U.S. Environmental Protection Agency, Washington, DC.

U.S. EPA. 1998b. Workshop on the Application of 2,3,7,8-TCDD Toxicity Equivalency Factors to Fish and Wildlife. U.S. Environmental Protection Agency. Chicago, IL. January 20-22.

USFWS. 1993. Reconnaissance Surveys of Contaminants Potentially Affecting Green Bay and Gravel Island National Wildlife Refuges. U.S. Fish and Wildlife Service, Ecological Services Field Office, Green Bay, WI.

USGS. 1990. USGeoData 1:250,000 and 1:100,000 Scale Land Use and Land Cover and Associated Maps Digital Data. U.S. Geological Survey, Reston, VA.

Van den Berg, M., B.L.H.J. Craane, T. Sinnige, S. van Mourik, S. Dirksen, T. Boudewijn, M. van der Gaag, I. J. Lutke-Schipholt, B. Spenkelink, and A. Brouwer. 1994. Biochemical and toxic effects of polychlorinated biphenyls (PCBs), dibenzo-p-dioxins (PCDDs) and dibenzofurans (PCDFs) in the cormorant (*Phalacrocorax carbo*) after in ovo exposure. *Environ. Toxicol. Chem.* 13: 803-816.

Van den Berg, M., L. Birnbaum, A.T.C. Bosveld, B. Brunstrom, P. Cook, M. Feely, J.P. Giesy, A. Hanberg, R. Hasegawa, S.W. Kennedy, T. Kubiak, J.C. Larsen, F.X. Rolaf van Leeuwen, A.K. Djien Liem, C. Nolt, R.E. Peterson, L. Poellinger, S. Safe, D. Schrenk, D. Tillitt, M. Tysklind, M. Younes, F. Waern, and T. Zacharewski. 1998. Toxic equivalency factors (TEFs) for PCBs, PCDDs, and PCDFs for humans and wildlife. *Environ. Health Perspect.* 106: 775-792.

Walker, C.H. 1995. Biochemical biomarkers in ecotoxicology-some recent developments. *Sci. Total Environ.* 171: 189-195.

Weseloh, D.V. and P.J. Ewins. 1994. Characteristics of a rapidly increasing colony of double-crested cormorants (*Phalacrocorax auritus*) in Lake Ontario: Population size, reproductive parameters and band recoveries. *J. Great Lakes Res.* 20(2): 443-456.

Weseloh, D.V., P.J. Ewins, and K.A. Pettit. 1995a. Continued growth of the double-crested cormorant populations on the Great Lakes. Abstract in *Bull. Colon. Waterbird Soc.* 19: 2.

Weseloh, D.V., P.J. Ewins, J. Struger, P. Mineau, C.A. Bishop, S. Postupalsky, and J.P. Ludwig. 1995b. Double-crested cormorants of the Great Lakes: Changes in population size, breeding distribution and reproductive output between 1913 and 1991. *Colon. Waterbirds* 18: 48-59.

White, D.H. and E. Cromartie. 1977. Residues of environmental pollutants and shell thinning in merganser eggs. *Wilson Bull.* 89: 532-542.

White, D.H. and D.J. Hoffman. 1995. Effects of polychlorinated dibenzo-*p*-dioxins and dibenzofurans on nesting wood ducks *(Aix sponsa)* at Bayou Meto, Arkansas. *Environ. Health Perspect.* 103(Supplement 4): 37-39.

Wiemeyer, S.N., C.M. Bunck, and C.J. Stafford. 1993. Environmental contaminants in bald eagle eggs — 1980-84 — and further interpretations of relationships to productivity and shell thickness. *Arch. Environ. Contam. Toxicol.* 24: 213-227.

Wiemeyer, S.N., T.G. Lamont, C.M. Bunck, C.R. Sindelar, F.J. Gramlich, J.D. Fraser, and M.A. Byrd. 1984. Organochlorine pesticide, polychlorobiphenyl, and mercury residues in bald eagle eggs — 1969 and 1979 — and their relationships to shell thinning and reproduction. *Arch. Environ. Contam. Toxicol.* 13: 529-549.

Wiemeyer, S.N., B.M. Mulhern, F.J. Ligas, R.J. Hensel, J.E. Mathisen, F.C. Robards, and S. Postupalsky. 1972. Residues of organochlorine pesticides, polychlorinated biphenyls, and mercury in bald eagle eggs and changes in shell thickness — 1969 and 1970. *Pest. Monitor. J.* 6(1): 50-55.

Williams, L.L., J.P. Giesy, D. Verbrugge, S. Jurzystra, and K.L. Stromborg. 1995a. Polychlorinated biphenyls and 2,3,7,8-tetrachlorodibenzo-*p*-dioxin equivalents in eggs of double-crested cormorants from a colony near Green Bay, Wisconsin, USA. *Arch. Environ. Contam. Toxicol.* 29: 327-333.

Williams, L.L., J.P. Giesy, D. Verbrugge, S. Jurzystra, H. Heinz, and K.L. Stromborg. 1995b. Polychlorinated biphenyls and 2,3,7,8-tetrachlorodibenzo-*p*-dioxin equivalents in eggs of red-breasted mergansers near Green Bay, Wisconsin, USA, in 1977-78. *Arch. Environ. Contam. Toxicol.* 29: 52-60.

Wilson, C.L. and S. Safe. 1998. Mechanisms of ligand-induced aryl hydrocarbon receptor-mediated biochemical and toxic responses. *Toxicol. Pathol.* 26(5): 657-671.

Wisconsin DNR. 1971-1995. State of Wisconsin fish contaminants monitoring database. Provided to Stratus Consulting Inc. in electronic format by J. Amrhein, Wisconsin Department of Natural Resources.

Wisconsin DNR. 1987. Press release detailing imposition of waterfowl consumption advisory. Wisconsin Department of Natural Resources, Madison. June 25.

Wisconsin DNR. 1998a. Fox River and Green Bay PCB Fate and Transport Model Evaluation: Compilation and Estimation of Historical Discharges of Total Suspended Solids and PCB from Fox River Point Sources. Wisconsin Department of Natural Resources. Draft.

Wisconsin DNR. 1998b. WISCLAND Land Cover (WLCGW930). Wisconsin Department of Natural Resources, Madison.

Woodford, J.E., Karasov, W.H., Meyer, M.W., and Chambers, L. 1998. Impact of 2,3,7,8-TCDD exposure on survival, growth, and behavior of ospreys breeding in Wisconsin, USA. *Environ. Toxicol. Chem.* 17: 1323-1331.

Yamashita, N., S. Tanabe, J.P. Ludwig, H. Kurita, M.E. Ludwig, and R. Tatsukawa. 1993. Embryonic abnormalities and organochlorine contamination in double-crested cormorants (*Phalacrocorax auritus*) and Caspian terns (*Hydroprogne caspia*) from the upper Great Lakes in 1988. *Environ. Pollut.* 79: 163-173.

Zubay, G. 1983. *Biochemistry.* Addison-Wesley Publishing Company, Reading, MA. 1268 pp.

APPENDIX A
SCIENTIFIC NAMES OF BIRD SPECIES MENTIONED IN TEXT

English Name	Scientific Name
Alder flycatcher	*Empidonax virescens*
American bittern	*Botaurus lentiginosus*
American black duck	*Anas rubripes*
American coot	*Fulica americana*
American crow	*Corvus brachyrhynchos*
American goldfinch	*Carduelis tristis*
American kestrel	*Falco sparverius*
American redstart	*Setophaga ruticilla*
American robin	*Turdus migratorius*
American tree sparrow	*Spizella arborea*
American wigeon	*Anas americana*
American woodcock	*Scolopax minor*
Baird's sandpiper	*Calidris bairdii*
Bald eagle	*Haliaeetus leucocephalus*
Bank swallow	*Riparia riparia*
Barn swallow	*Hirundo rustica*
Barred owl	*Strix varia*
Bay-breasted warbler	*Dendroica castanea*
Belted kingfisher	*Ceryle alcion*
Bell's vireo	*Vireo bellii*
Black tern	*Chlidonias niger*
Black-and-white warbler	*Mniotilta varia*
Black-bellied plover	*Pluvialis squatorola*
Black-billed cuckoo	*Coccyzus erythropthalmus*
Black-capped chickadee	*Parus atricapillus*
Black-crowned night heron	*Nycticorax nycticorax*
Black-headed gull	*Larus ridibundus*
Black-throated blue warbler	*Dendroica caerulescens*
Black-throated green warbler	*Dendroica virens*
Blackburnian warbler	*Dendroica fusca*
Blackpoll warbler	*Dendroica striata*
Blue jay	*Cyanocitta cristata*
Blue-gray gnatcatcher	*Polioptila caerulea*
Blue-winged teal	*Anas discors*
Blue-winged warbler	*Vermivora pinus*

Bobwhite	*Colinus virginianus*
Bohemian waxwing	*Bombycilla garrulus*
Bonaparte's gull	*Larus philadelphia*
Boreal chickadee	*Parus hudsonicus*
Brewer's blackbird	*Euphagus cyanocephalus*
Broad-winged hawk	*Buteo platypterus*
Brown creeper	*Certhia americana*
Brown thrasher	*Toxostoma rufum*
Brown-headed cowbird	*Molothrus ater*
Bufflehead	*Bucephala albeola*
Canada goose	*Branta canadensis*
Canada warbler	*Wilsonia canadensis*
Canvasback	*Aythya valisineria*
Cape May warbler	*Dendroica tigrina*
Caspian tern	*Hydroprogne caspia*
Cattle egret	*Bubulcus ibis*
Cedar waxwing	*Bombycilla cedrorum*
Cerulean warbler	*Dendroica cerulea*
Chestnut-sided warbler	*Dendroica pensylvanica*
Chicken	*Gallus gallus*
Chimney swift	*Chaetura pelagica*
Chipping sparrow	*Spizella passerina*
Clay-colored sparrow	*Spizella pallida*
Cliff swallow	*Hirundo pyrrhonota*
Common eider	*Somateria mollissima*
Common goldeneye	*Bucephala clangula*
Common grackle	*Quiscalus quiscula*
Common loon	*Gavia immer*
Common merganser	*Mergus merganser*
Common moorhen	*Gallinula chloropus*
Common murre	*Uria aalge*
Common nighthawk	*Chordeiles minor*
Common raven	*Corvus corax*
Common redpoll	*Carduelis flammea*
Common snipe	*Gallinago gallinago*
Common tern	*Sterna hirundo*
Common yellowthroat	*Geothlypis trichas*
Connecticut warbler	*Oporornis agilis*
Cooper's hawk	*Accipter cooperii*
Dark-eyed junco	*Junco hyemalis*
Dickcissel	*Spiza americana*
Domestic goose	*Anser anser*

Double-crested cormorant	*Phalacrocorax auritus*
Downy woodpecker	*Picoides pubescens*
Dunlin	*Calidris alpina*
Eastern bluebird	*Sialia sialia*
Eastern kingbird	*Tyrannus tyrannus*
Eastern meadowlark	*Sturnella magna*
Eastern phoebe	*Sayornis phoebe*
Eastern screech owl	*Otus asio*
Eastern wood-pewee	*Contopus virens*
European shag	*Phalacrocorax aristotelis*
Evening grosbeak	*Coccothraustes vespertinus*
Field Sparrow	*Spizella pusilla*
Forster’s tern	*Sterna forsteri*
Fox sparrow	*Passerella iliaca*
Franklin’s gull	*Larus pipixcan*
Gadwall	*Anas strepera*
Glaucus gull	*Larus hyperboreus*
Golden-winged warbler	*Vermivora chrysoptera*
Grasshopper sparrow	*Amodramus savannarum*
Gray catbird	*Dumetella carolinensis*
Gray jay	*Perisoreus canadensis*
Gray partridge	*Perdix perdix*
Gray-cheeked thrush	*Catharus minimus*
Great blue heron	*Ardea herodias*
Great crested flycatcher	*Myarchus crinitus*
Great egret	*Casmerodius albus*
Great horned owl	*Bubo virginianus*
Greater scaup	*Aythya marila*
Greater yellowlegs	*Tringa melanoleuca*
Green-backed heron	*Butorides virescens*
Green-winged teal	*Anas crecca*
Hairy woodpecker	*Picoides villosus*
Harris’s sparrow	*Zonotrichia querula*
Hermit thrush	*Catharus guttatus*
Herring gull	*Larus argentatus*
Hooded merganser	*Lophodytes cucullatus*
Hooded warbler	*Wilsonia citrina*
Horned grebe	*Podiceps auritus*
Horned lark	*Eremophila alpestris*
House sparrow	*Passer domesticus*
House wren	*Troglodytes aedon*
Hudsonian godwit	*Limosa haemastica*

Indigo bunting	*Passerina cyanea*
Japanese quail	*Coturnix japonica*
Killdeer	*Charadrius vociferus*
Lapland longspur	*Calcarius lapponicus*
Lark sparrow	*Chondestes grammacus*
Le Conte's sparrow	*Ammodramus leconteii*
Least bittern	*Ixobrychus exilis*
Least Flycatcher	*Empidonax minimus*
Least sandpiper	*Calidris minutilla*
Lesser scaup	*Aythya affinis*
Lesser yellowlegs	*Tringa flavipes*
Lincoln's sparrow	*Melospiza lincolnii*
Little gull	*Larus minutilla*
Loggerhead shrike	*Lanius ludovicianus*
Long-billed dowitcher	*Limnodromus scolopaceus*
Long-eared owl	*Asio otus*
Louisiana waterthrush	*Seiurus motacilla*
Magnolia warbler	*Dendroica magnolia*
Mallard	*Anas platyrhynchos*
Marbled godwit	*Limosa fedoa*
Marsh wren	*Telmatodytes palustris*
Merlin	*Falco columbarius*
Mourning dove	*Zenaidura macroura*
Mourning warbler	*Oporornis philadelphia*
Mute swan	*Cygnus olor*
Nashville warbler	*Vermivora ruficapilla*
Northern cardinal	*Cardinalis cardinalis*
Northern flicker	*Colaptes auratus*
Northern goshawk	*Accipiter gentilis*
Northern harrier	*Circus cyaneus*
Northern mockingbird	*Mimus polyglottos*
Northern oriole	*Icterus galbula*
Northern parula	*Parula americana*
Northern pintail	*Anas acuta*
Northern rough-winged swallow	*Stelgidopteryx serripennis*
Northern saw-whet owl	*Aegolius acadicus*
Northern shoveller	*Anas clypeata*
Northern shrike	*Lanius excubitor*
Northern waterthrush	*Seiurus noveboracensis*
Oldsquaw	*Clangula hyemalis*
Olive-sided flycatcher	*Contopuis borealis*
Orange-crowned warbler	*Vermivora celata*

Orchard oriole	*Icterus spurius*
Osprey	*Pandion haliaetus*
Ovenbird	*Seiurus aurocapillus*
Palm warbler	*Dendroica palmarum*
Pectoral sandpiper	*Calidris melanotus*
Peregrine falcon	*Falco peregrinus*
Philadelphia vireo	*Vireo philadelphicus*
Pied-billed grebe	*Podilymbus podiceps*
Pileated woodpecker	*Dryocopus pilleatus*
Pine grosbeak	*Pinicola enucleator*
Pine siskin	*Carduelis pinus*
Pine warbler	*Dendroica pinus*
Prothonotary warbler	*Protonotaria citrea*
Purple finch	*Carpodacus purpureus*
Purple martin	*Progne subis*
Red crossbill	*Loxia curvirostra*
Red-bellied woodpecker	*Melanerpes carolinus*
Red-breasted merganser	*Mergus serrator*
Red-breasted nuthatch	*Sitta canadensis*
Red-eyed vireo	*Vireo olivaceus*
Red-headed woodpecker	*Melanerpes erythrocephalus*
Red-necked grebe	*Podiceps grisegena*
Red-necked phalarope	*Phalaropus lobatus*
Red-shouldered hawk	*Buteo lineatus*
Red-tailed hawk	*Buteo jamaicensis*
Red-throated loon	*Gavia stellata*
Red-winged blackbird	*Agelaius phoenicius*
Redhead	*Aythya americana*
Ring-billed gull	*Larus delawarensis*
Ring-necked dove	*Streptopelia risoria*
Ring-necked duck	*Aythya collaris*
Ring-necked pheasant	*Phasianus colchicus*
Rock dove	*Columba livia*
Rose-breasted grosbeak	*Pheucticus ludovicianus*
Rough-legged hawk	*Buteo lagopus*
Ruby-throated hummingbird	*Archilocus colubris*
Ruddy duck	*Oxyura jamaicensis*
Ruddy turnstone	*Arenaria interpres*
Ruffed grouse	*Bonasa umbellus*
Rufous-sided towhee	*Pipilo erythrophthalmus*
Rusty blackbird	*Euphagus carolinus*
Sanderling	*Calidris alba*

Sandhill crane	*Grus canadensis*
Savannah sparrow	*Passerculus sandwichensis*
Scarlet tanager	*Piranga olivacea*
Sedge wren	*Cistothorus platensis*
Semipalmated sandpiper	*Calidris pusilla*
Sharp-shinned hawk	*Accipter striatus*
Sharp-tailed grouse	*Tympanuchus phasianellus*
Short-billed dowitcher	*Limnodromus griseus*
Short-eared owl	*Asio flammeus*
Snow bunting	*Plectrophenax nivalis*
Snow goose	*Anser caerulescens*
Snowy owl	*Nyctea scandiaca*
Solitary sandpiper	*Tringa solitaria*
Solitary vireo	*Vireo solitarius*
Song sparrow	*Melospiza melodia*
Sora	*Porzana carolina*
Spotted sandpiper	*Actitus macularia*
Starling	*Sturnus vulgaris*
Stilt sandpiper	*Calidris himantopus*
Swainson's thrush	*Catharus ustulatus*
Swamp sparrow	*Melospiza georgiana*
Tennessee warbler	*Vermivora peregrina*
Tree swallow	*Iridoprocne bicolor*
Tufted titmouse	*Parus bicolor*
Tundra swan	*Cygnus columbianus*
Turkey	*Meleagris gallopavo*
Turkey vulture	*Cathartes aura*
Upland sandpiper	*Bartramia longicauda*
Veery	*Catharus fuscescens*
Vesper sparrow	*Phooecetes gramineus*
Virginia rail	*Rallus limicola*
Warbling vireo	*Vireo gilvus*
Water pipit	*Anthus spinoletta*
Western meadowlark	*Sturnella neglecta*
White pelican	*Pelecanus erythrorhynchos*
White-crowned sparrow	*Zonotrichia leucophrys*
Whip-poor-will	*Caprimulgus vociferus*
White-breasted nuthatch	*Sitta carolinensis*
White-rumped sandpiper	*Calidris fuscicollis*
White-tailed eagle	*Haliaeetus albicilla*
White-throated sparrow	*Zonotrichia albicollis*
White-winged crossbill	*Loxia leucoptera*

White-winged scoter	*Melanitta deglandi*
Willet	*Catoptrophorus semipalmatus*
Willow flycatcher	*Empidonax traillii*
Wilson's phalarope	*Phalaropus tricolor*
Wilson's warbler	*Wilsonia pusilla*
Winter wren	*Troglodytes troglodytes*
Wood duck	*Aix sponsa*
Wood thrush	*Hylocichla mustelina*
Yellow warbler	*Dendroica petechia*
Yellow-bellied flycatcher	*Empidonax flaviventris*
Yellow-bellied sapsucker	*Sphyrapicus varius*
Yellow-billed cuckoo	*Coccyzus americanus*
Yellow-breasted chat	*Icteria virens*
Yellow-headed blackbird	*Xanthocephalus xanthocephalus*
Yellow-rumped warbler	*Dendroica coronata*
Yellow-throated vireo	*Vireo flavifrons*