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Contaminants in Fishes from Great Lakes-Influenced Sections and Above Dams of Three Michigan Rivers: III. Implications for Health of Bald Eagles

J. P. Giesy[1], W. W. Bowerman[1], M. A. Mora[1,*], D. A. Verbrugge[1], R. A. Othoudt[1], J. L. Newsted[1,**], C. L. Summer[2], R. J. Aulerich[4], S. J. Bursian[4], J. P. Ludwig[2], G. A. Dawson[3], T. J. Kubiak[5,***], D. A. Best[5], D. E. Tillitt[6]

[1] Department of Fisheries & Wildlife, Pesticide Research Center and Institute for Environmental Toxicology, Michigan State University, East Lansing, Michigan 48824-1222, USA
[2] SERE Group, 1670 Garnet Road, Victoria BC V8P 3C8, Canada
[3] Environmental Department, Consumers Power Company, 1945 West Parnall Road, Jackson, Michigan 49201, USA
[4] Department of Animal Science, Michigan State University, East Lansing, Michigan 48824, USA
[5] U.S. Fish and Wildlife Service, East Lansing Field Office, 1405 South Harrison Road, East Lansing, Michigan 48823, USA
[6] National Biological Service, Midwest Research Center, 4200 New Haven Road, Columbia, Missouri 65201, USA

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Abstract. Recently, there have been discussions of the relative merits of passage of fishes around hydroelectric dams on three rivers (Au Sable, Manistee, and Muskegon) in Michigan. A hazard assessment was conducted to determine the potential for adverse effects on bald eagles that could consume such fishes from above and below dams on the three primary rivers. The hazard assessments were verified by comparing the reproductive productivities of eagles nesting in areas where they ate primarily fish from either above or below dams on the three primary rivers, as well as on two additional rivers in Michigan, the Menominee and Thunder Bay. Concentrations of organochlorine insecticides (OCI), polychlorinated biphenyls (total PCBs), 2,3,7,8-tetrachlorodibenzo-*p*-dioxin equivalents (TCDD-EQ), and total mercury (Hg) were measured in composite samples of fishes from above and below hydroelectric dams on the Manistee and Muskegon Rivers, which flow into Lake Michigan, and the Au Sable River, which flows into Lake Huron. Mean concentrations of OCI, total PCBs, and TCDD-EQ were all greater in fishes from below the dams than in those from above. The hazard assessment indicated that current concentrations of Hg and OCI other than DDT (DDT + DDE + DDD) in fish from neither above nor below dams would present a significant hazard to bald eagles *(Haliaeetus leucocephalus)*. Both total PCBs and TCDD-EQ in fishes from below the dams currently present a significant hazard to bald eagles, since their mean hazard quotients (HQ) were all greater than one.

There is a certain sentiment, among some people, to return the rivers of Michigan to a condition more similar to their original state. This would include the removal of hydroelectric dams. Alternatively, fish could be passed over the dams to provide greater access to anadromous species by recreational anglers. The entire Great Lakes ecosystem is considerably altered from its status when it was first visited by Europeans. Species have been introduced, both intentionally and unintentionally. Thus, management decisions must be made in the context of these changes. One major change in the ecosystem has been the introduction of anadromous salmonid fishes. Another has been the introduction of pollutants, such as organochlorine insecticides (OCI) and industrial chemicals collectively termed synthetic halogenated hydrocarbons (SHH), that have accumulated in these fishes. There is little that can be done about the current concentrations of SHH in the Great Lakes Basin. However, management decisions must still be made. For instance, managers must consider questions such as, should stocking programs be initiated; should more rivers be opened to facilitate spawning; or should species be reintroduced to areas from which they have been extirpated? If self-reproducing popula-

* *Present address:* National Biological Service, Department of Wildlife & Fisheries, Texas A&M University, College Station, TX 77843, USA

** *Present address:* Department of Fisheries & Wildlife, University of Massachusetts, Amherst, MA 01003, USA

*** *Present address:* Division of Environmental Contaminants, U.S. Fish and Wildlife Service, 4401 N. Fairfax Drive, #330, Arlington, VA 22203, USA

Correspondence to: J. P. Giesy

tions of these species would not be expected to be maintained in these areas, due to exposure to toxicants, or if accumulation of contaminants would create a hazard to other species, it seems that such activities would be inappropriate.

Exposures of wildlife in the Great Lakes region to SHH have resulted in adverse effects (Fitchko 1986; Allan *et al.* 1991a, 1991b; Giesy *et al.* 1994a, 1994c), including subtle effects on their reproductive potential (Kubiak *et al.* 1989; Fox *et al.* 1991), such as deformities and lethality of embryos (Gilbertson 1983; Gilbertson *et al.* 1976). These effects, including declines in populations, have been best documented for colonial, fish-eating water birds (Gilbertson *et al.* 1991; Peakall and Fox 1987). The most dramatic effect on reproductive performance was the result of eggshell thinning caused primarily by DDE (Anderson and Hickey 1969; Weseloh *et al.* 1979; Elliott *et al.* 1988). Since the cessation of the manufacture and use of the most persistent and widespread contaminants, concentrations of these compounds in fish and birds have decreased (Allan *et al.* 1991a). Specifically, concentrations of DDE in bird eggs of most areas of the North American Great Lakes region have decreased such that they are less than the critical concentration for egg shell thinning (Peakall 1993). Subsequently, populations of many of the fish-eating, water birds have increased (Price and Weseloh 1986). However, other adverse effects, such as localized impairment of reproductive performance (Tillitt *et al.* 1992) and anatomical defects have persisted (Yamashita *et al.* 1992). Since the declines in concentrations of DDT, populations of bald eagles (*Haliaeetus leucocephalus*) and other fish-eating raptors have increased (Anthony 1989; Bowerman 1993; Bowerman *et al.* 1990). However, there are a number of open habitats along the Great Lakes shoreline and the productivity of bald eagles in areas where they have access to fishes from the Great Lakes have lesser productivity than those in more inland areas (Bowerman 1993; Bowerman et al. 1994).

Here we report the results of a hazard assessment conducted as part of the relicensing process mandated by the Federal Energy Regulatory Commission (FERC) for dams on three rivers in Michigan: Manistee, Muskegon, and Au Sable Rivers. Since anadromous fish that contain toxic chemicals move into the streams, it is possible for them to expose other fish and wildlife to potentially hazardous concentrations of persistent chemicals. Merna (1986), for example, documented the uptake of PCBs by resident populations of brook trout (*Salvelinus fontinalis*) by ingestion of salmon eggs at spawning grounds downstream of barriers on the Manistee and Muskegon Rivers. Also, in the study of Merna, the trimmed fillets of brown trout exceeded the total concentrations of PCBs recommended for human consumption. Fish-eating raptors are among the most sensitive species and are exposed to the greatest concentrations of bioaccumulative compounds. Therefore, a hazard assessment was conducted to determine the potential for adverse effects of removing or providing passage around the dams, which currently restrict anadromous fishes from all or part of the rivers, on populations of bald eagles living along these three rivers. Concentrations of SHH and Hg were measured in several species of fishes both above and below the dams, which separated the fish populations that had access to the Great Lakes from those that did not. The hazard assessment was conducted by comparing the concentrations of Hg, total PCBs, OCI, and TCDD-EQ in the fishes with dose-response relationships for effects of these compounds on bald eagles or, in the absence of bald eagle-specific information, to effects on appropriate model species.

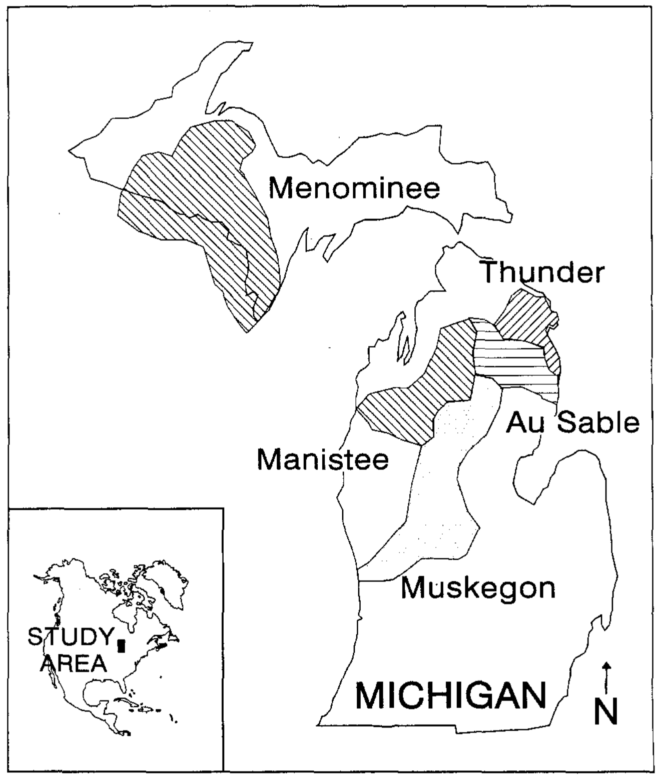

Fig. 1. Map of Michigan which shows the basins of the Manistee, Muskegon and Au Sable rivers where fish were collected, and the Menominee, and Thunder rivers which were used in addition to the previous three rivers to determine risk to bald eagles nesting within the five basins

Materials and Methods

Fish Collection

Fish were collected, primarily by electro-shocking, between 12 September and 5 December, 1989, and again between 26 March and 15 May, 1990, from above and below dams on the Au Sable, Manistee, and Muskegon Rivers of lower Michigan (Figure 1; Giesy *et al.* 1994b). On the Au Sable River, fish were also collected from a third location which was above the second most downstream dam. This location was designated as the "middle" sampling area. The results of the instrumental analyses for this location are given, but the values are not included in the means given for the "above" locations. After collection, five individual fish of the same species and approximately the same size were pooled and homogenized with a Hobart meat grinder. Part of the homogenate was stored in cleaned glass jars at −20°C until chemical analysis. The methods for sanple preparation, extraction, clean-up, and instrumental quantification have been given in detail elsewhere (Giesy *et al.* 1994b).

Hazard Assessment

The hazard of the SHH or Hg presented to the bald eagle or surrogate species was determined by the toxic units approach. One toxic unit was defined as the quotient of the concentration in the diet multiplied by the

Table 1. Hazard assessment of concentrations of total polychlorinated biphenyls (PCBs), DDE, dieldrin, TCDD-EQ and Hg in bald eagles from three river systems, lower Michigan. LOAEC, biomagnification factors, and total PCBs, DDE and OCls in fish

Toxicant Effect	**Total PCBs** Egg lethality	***p,p′*-DDE** Productivity	**Dieldrin** Egg lethality	**TCDD-EQ** Egg lethality	**Mercury** Egg lethality
NOAEC (mg/kg egg)	4.0[a]	3.5[b]	0.1[b]	7×10^{-6}[c]	0.5[b]
Dietary NOAEC (mg/kg fish)	0.14	0.16	1.4×10^{-2}	3.7×10^{-7}	0.5
BMF (fish to egg)	28[d]	22[d]	7[e]	19[f]	1[d]
Concentration in fish					
Shoreline (mg/kg)	2.1	0.3	3.3×10^{-2}	2.0×10^{-5}	0.2
HQ-Shoreline	15	1.9	2.4	54	0.3
Concentration in fish					
Interior (mg/kg)	0.2	3.5×10^{-2}	8.7×10^{-4}	0.7	0.3
HQ-Interior	1.3	0.22	0.06	1.9	0.7

[a] Wiemeyer 1990
[b] Wiemeyer *et al.* 1984
[c] White and Setinak 1994
[d] Calculated from concentrations in fish and bald eagle eggs from inland or coastal areas
[e] Braune and Norstrom 1989
[f] Kubiak and Best 1991

accumulation factor from fish to bird egg and then dividing by the no observable adverse effect concentration (NOAEC) for eggs (Equation 1). This ratio is also referred to as the hazard quotient (HQ). When the HQ is equal to 1 toxic unit, the concentration in individual species of fish would equal the threshold concentration below which no statistically significant response for which the toxic units were defined if 100%

$$HQ = \frac{[\text{Concentration Fish} \cdot \text{BMF}]}{[\text{EGG NOAEC}]} \tag{1}$$

of the diet was comprised of a particular species of fish. The units of concentration of the toxicant must be the same for the NOAEC and the concentration in the fish. Concentrations of the contaminants used in the hazard assessment have been reported previously (Giesy *et al.* 1994b). HQ were calculated for each species of fish to give a relative degree of hazard in the diet of bald eagles. Subsequently, a hazard assessment was conducted with weighted average exposure, based on the relative proportions of each species of fish in the diet. The relative proportions of each species of fish in the diet were determined from visual observations of the prey taken by eagles and from an analysis of the prey remains in or around the nests of eagles in the various areas (Bowerman 1993). When this was done, there was no difference in the conclusions that would be drawn for any of the toxicants on any of the three rivers studied if the simple HQ values had been used.

Dose-response relationships for the bald eagle were used when available. However, since the bald eagle is a threatened or endangered species in many areas, it is difficult, if not impossible, to conduct controlled laboratory experiments or make field collections. Thus, it is often necessary to use the results of studies with surrogate species. Here we used information for alternative species that were considered appropriate. We chose results from species that were similar to bald eagles or that are known to have similar sensitivities to compounds for which there is information for bald eagles. To verify the hazard assessments, they were reconciled with the current distributions of bald eagles and their exposure to the various toxicants. We have not applied any uncertainty factors to the estimates.

Biomagnification factors (BMF) were used to predict the magnification of toxicants from fishes to bald eagle eggs. Where possible, BMF was calculated from measurements of the toxicants in fishes and bald eagle eggs in a region. However, since it was not always possible to obtain empirical values, BMF values were used from the literature (Kubiak and Best 1991; Braune and Norstrom 1989; Table 1). Since there were not enough samples to test for significant differences in concentrations among species within and among rivers, predicted concentrations of toxicants in bald eagle eggs were calculated from mean concentrations representative of the concentrations observed in the fish populations both below and above dams. Two biomagnification factors (BMF), one for the inland population (more than 8 km from the Great Lakes shoreline) and another one for bald eagles living along the shoreline, were calculated. We then calculated threshold concentrations of individual SHH, Hg, and TCDD-EQ based on dietary intake of fish by bald eagles (Table 1). An average BMF was selected which allowed us to use a single threshold level to determine the toxic units in fish.

Population Analysis

The potential effects of fishes from the Au Sable, Manistee, Menominee, Muskegon, and Thunder Bay Rivers to reproductive productivity for bald eagles nesting along these rivers in relation to these proposed actions were evaluated. Nesting records were used to determine reproductive productivity of bald eagles nesting along each of the five rivers and their tributaries for the period 1989-1993. Terms related to reproductive productivity follow those of Postupalsky (1974). Reproductive statistics determined included number of occupied breeding areas, number of fledged young, and reproductive productivity (defined as the total number of fledged young divided by the number of occupied breeding areas for the five-year time interval). The impact of fish passage was assessed for individual river basins, and the impact of fish passage on the entire bald eagle population in the state of Michigan was then determined.

Results and Discussion

Biomagnification Factors

Empirical biomagnification (BMF) values could be calculated for total PCBs and DDT complex, which includes both *o-p′* and *p-p′* isomers of DDT, DDE, and DDD (Table 2). Since the

Table 2. Concentrations (mean) of total polychlorinated biphenyls (PCBs), TCDD-EQ, DDE, and Hg in fish species from below and above dams of three Michigan Rivers. The downstream locations are those which were influenced by fishes from the Great Lakes. Concentrations are given as mg/kg wet weight except for TCDD-EQ, which are in ng/kg, wet weight

	Total PCBs		TCDD-EQ		DDE		Dieldrin		Hg	
Species	Up	Down	Up	Down	Up	Down	Up	Down	Up	Down
Chinook S.	—	1.7	—	39	—	0.28	—	0.02	—	0.20
Steelhead	—	3.9	—	67	—	0.74	—	0.11	—	0.26
Carp	0.16	6.0	2.6	70	0.06	0.63	6.0×10^{-5}	0.06	0.05	0.23
Walleye	0.22	2.9	2.9	38	0.04	0.33	1.5×10^{-3}	0.03	0.38	0.15
W. Sucker	0.03	0.4	2.8	12	0.01	0.11	0.04	0.01	0.11	0.07
Pike	0.03	1.0	2.4	22	0.02	0.15	0.04	7.5×10^{-3}	0.18	0.19
Perch	0.07	0.8	3.0	23	0.12	0.02	1.9×10^{-3}	0.03	0.73	0.05
B. Trout	0.04	—	2.6	—	0.01	—	9.0×10^{-4}	—	0.45	—

differences among fishes within a sampling location were not significant for these two classes, the median concentration was representative of the concentrations observed in all of the fishes. The median concentrations of total PCBs from the three rivers were 1.3 and 6.4×10^{-2} mg total PCBs/kg, ww below and above the dams, respectively. The median concentrations of total DDT were 0.6 and 6.0×10^{-2} mg/kg, below and above the dams, respectively. Median total concentrations of PCBs and DDT complex in bald eagle eggs were 35 and 9 mg PCB/kg, ww and 13 and 3 mg DDT complex/kg, ww for the areas along the Great Lakes shorelines and inland, respectively (Bowerman *et al.* 1994). These values were used to calculate BMF values for both total PCBs and the DDT complex for the Great Lakes shoreline or inland (Table 1). An overall average value was also calculated.

Hazard Assessment

OCI: Bald eagles have been simultaneously exposed to a number of synthetic, halogenated compounds, including OCI. Because the concentrations of a number of these compounds have been inter-correlated, it has been difficult to separate the effects and determine which of these compounds were most likely to have caused adverse effects in populations of bald eagles from the Great Lakes. Historically, when concentrations of OCI in the tissue of fishes from the Great Lakes were greater, it was concluded that they were probably the primary cause of the effects that were observed (Postupalsky 1971). Although concentrations of most of these OCI have decreased since the 1970s (Allan *et al.* 1991a, 1991b), adverse effects to bald eagles related to the mode of action of these OCI continue to be documented in bald eagles that consume fishes from the Great Lakes (Best *et al.* 1994; Bowerman 1993). Since current concentrations in fishes are not associated with acute lethality, it is unlikely at this time that any of the OCI would, by themselves, restrict bald eagles from areas where they eat fish from the Great Lakes or their tributaries (Franson *et al.* 1974; Proulx *et al.* 1987). Of the OCI that were measured, based on their toxicity and concentrations measured in fish, the two most likely to present a hazard to bald eagles were dieldrin and the DDT complex.

Dieldrin: While dieldrin is known to be toxic to birds and is suspected of having caused population-level effects (Davison and Sell 1974; Blus *et al.* 1979), it is not likely that current concentrations of dieldrin in fishes of the Michigan rivers studied present a significant hazard to bald eagles. The lowest observable adverse effect concentration (LOAEC) utilized in the hazard assessment is conservative and based on the regression of Wiemeyer *et al.* (1984), which relates dieldrin concentrations in eggs to productivities of individual pairs of bald eagles. However, Wiemeyer *et al.* (1984) state that "while dieldrin concentrations greater than 1.0 mg/kg, ww in eggs can not be ruled out as having an effect on reproduction, the major effect of dieldrin was related to adult survival." While no controlled studies have been conducted with bald eagles, Mendenhall *et al.* (1983) found that lethality occurred in adult barn owls (*Tyto alba*) at a dietary concentration of 0.58 mg/kg, ww. Historically, adult bald eagles were reported to be killed by exposure to dieldrin, but currently the exposures form fish (Giesy *et al.* 1994a) is at least 30 times less than that associated with adult lethality.

The actual effects of dieldrin on reproductive productivity of bald eagles, reported by Wiemeyer *et al.* (1984), are most likely an artifact due to co-correlation of the concentrations of dieldrin with those of total PCBs and the DDT complex. When considered with the dose-response relationships obtained in controlled laboratory studies with other adult raptors, such as adult barn owls, we conclude that the correlation is most likely spurious and not indicative of actual toxicity of dieldrin at the concentrations observed. Concentrations of dieldrin in eggs of other species of birds have been found to range from 9.4×10^{-1} mg/kg, ww in brown pelicans (*Pelecanus occidentalis*) (Blus *et al.* 1979) to 1.8×101 ppm in purple gallinules (*Prophyrula martinica*) (Fowler *et al.* 1971). However, Potts (1968) suggested that concentrations of 2 to 3 mg/kg, ww in eggs of Shag (*Phalacrocorax aristotelis*) caused total reproductive failure. Thus, the NOAEC of 0.1 mg dieldrin/kg, ww in the egg which we selected (Table 1) seems reasonable.

The concentrations of dieldrin in the fishes ranged from less than the method detection limit of 6.0×10^{-5} to 1.1×10^{-1} mg/kg, ww, with the greatest concentration occurring in steelhead trout (Giesy *et al.* 1994b). When average concentrations (Table 2) were used, the HQ were greater than 1.0 below the dams and less than 1.0 above the dams on the Manistee and Muskegon Rivers, which drain into Lake Michigan (Table 3). The HQ was less than 1.0 both below and above the dams on the Au Sable River, which drains into Lake Huron. The ratio between the HQ below the dams to that above the dams was as great as 109 on the Manistee River; this was the greatest ratio observed for dieldrin. Based on minimum concentrations in fishes, dieldrin did not exceed the HQ of 1.0 below or above the

Table 3. Mean concentrations of toxicants in fishes above and below dams on three Michigan rivers. Hazard quoteint (HQ) and the ratio between the HQ values below and above the dams are also given

River	[mg/kg] wet wt. Below	[mg/kg] wet wt. Above	HQ Below	HQ Above	Ratio (B/A)
Total polychlorinated biphenyls					
Manistee	1.9	0.020	13.5	0.14	96
Muskegon	3.4	0.195	23.9	1.36	18
Au Sable	1.1	0.061	7.6	0.43	18
All rivers	2.1	0.09	15	0.6	24
DDE					
Manistee	0.4	0.010	2.35	0.06	39
Muskegon	0.4	0.082	2.51	0.51	5
Au Sable	0.1	0.011	0.76	0.07	11
All rivers	0.3	0.350	1.88	0.22	9
Dieldrin					
Manistee	4.6×10^{-2}	3.5×10^{-4}	3.29	0.03	110
Muskegon	4.5×10^{-2}	1.3×10^{-3}	3.21	0.09	36
Au Sable	8.8×10^{-3}	9.7×10^{-4}	0.63	0.07	9
All rivers	3.3×10^{-2}	8.7×10^{-4}	2.36	0.06	39
TCDD-EQ					
Manistee	2.2×10^{-5}	7.3×10^{-7}	59.46	1.97	30
Muskegon	2.8×10^{-5}	6.9×10^{-7}	75.68	1.86	41
Au Sable	9.9×10^{-6}	7.3×10^{-7}	26.76	1.97	14
All rivers	2.0×10^{-5}	7.0×10^{-7}	54.05	1.89	29
Hg					
Manistee	0.207	0.180	0.41	0.36	1
Muskegon	0.215	0.490	0.43	0.98	0.4
Au Sable	0.077	0.333	0.15	0.66	0.2
All rivers	0.166	0.333	0.33	0.66	0.5

dams, but based on the maximum observed concentrations in fishes, the HQ was greater than 1.0 below the dams, but not above. When the concentrations of dieldrin in fishes were adjusted for the dietary consumption of fishes in the diet of bald eagles, the HQ from above and below the dams was near 1.0, with that from the upstream location approximately one-third greater than that below the dams (Table 3). Thus, it is concluded that the current concentrations of dieldrin in fishes above the dams are well below any concentration that would be expected cause any adverse effects, while those below the dams are slightly greater than the NOAEC, but not sufficiently so to be expected to cause any adverse effects. Dieldrin is not the critical toxicant limiting the distribution or reproductive potential of bald eagles in Michigan.

It has been suggested that lindane, endrin, and oxychlordane may act in an additive manner with dieldrin to cause adult lethality in raptors. If concentrations (Giesy *et al.* 1994a) of all of these insecticides are considered in a simple additive model, the total would be approximately the same as dieldrin alone. Thus, these compounds would not be expected to cause any lethality of adult bald eagles. This prediction is substantiated by the current lack of mortality of adult bald eagles.

DDE: The effects of *p,p'*-DDE on bald eagle reproduction have been documented in the field (Wiemeyer *et al.* 1984). However, the effects of DDE and total PCBs are significantly correlated and separation of effects is difficult (Wiemeyer *et al.* 1984). The effect-concentrations associated with eggshell thinning in osprey (*Pandion haliaetus*) were 2.0, 4.2, and 8.7 mg DDE/kg, ww for reductions of eggshell thicknesses of 10, 15, and 20%, respectively (Wiemeyer *et al.* 1988). Since we used 15% thinning as the critical degree of eggshell thinning as the estimate of the NOAEC, this result is similar to that found for bald eagles.

In laboratory studies, DDE has been linked to eggshell thinning in several species of birds (Heath *et al.* 1972; Wiemeyer and Porter 1970; Longcore *et al.* 1971; McClain and Hall 1972; Peakall *et al.* 1973; Lincer 1975; Newton 1979; Mendenhall *et al.* 1983). Therefore, the NOAEC was compared based on productivity predicted from the regression analysis (Wiemeyer *et al.* 1984) with values from similar species and to the results of controlled laboratory studies where co-correlation did not confound the analysis.

The HQ associated with DDE in fishes from the three rivers was small. The HQ ranged from 7.6×10^{-1} to 2.5 below the dams and from 7.0×10^{-2} to 5.1×10^{-1} above (Table 3). Utilization of some species of fish, such as chinook salmon (*Onchorhyncus tshawytscha*), steelhead trout (*Onchorhyncus mykiss*), carp (*Cyprinus carpio*), and walleye (*Stisostedium vitreum*), from below the dams as a sole food source, however, would result in HQ values greater than 1.0. Except for carp, these species are rarely taken during the breeding season. However, their utilization during spring and fall spawning runs by bald eagles has been documented (Bowerman 1993) and could result in some adverse effects on bald eagles if they were taken extensively. It is difficult to know what effect this exposure could have on reproductive performance, since the exposure would not be occurring at the time when it would have the maximum potential for being passed to the eggs. As concentrations of DDE in the environment have declined, populations of bald eagles have increased (Grier 1980; Postupalsky 1985).

Other Organochlorine Insecticides

The role of other OCI, including hexachlorobenzene, mirex, kepone, and toxaphene, were previously explored by Wiemeyer *et al.* (1984); none were found to be related to reproductive productivity or eggshell thinning in bald eagles. The concentrations observed in fishes from the three Michigan rivers studied here are not sufficiently great at this time to cause adverse population-level effects on bald eagles. Concentrations of hexachlorobenzene as great as 100 mg/kg, ww in the diet had no adverse effects on reproductive productivity of chickens (Hansen *et al.* 1978). Dietary exposure of Japanese Quail (*Coturnix japonica*) to 10 mg kepone/kg did not adversely affect either survival of the adults or their reproductive productivity (Eroschenko and Hackmann 1981). Although it is unlikely that these OCI are currently occurring at concentrations which would represent a hazard to bald eagles, it is possible that there could be subtle effects on bald eagles. However, as will be demonstrated below, several other contaminants are currently occurring at concentrations which would be expected to be hazardous to bald eagles.

Total Polychlorinated Biphenyls

Concentrations of PCBs in the food and eggs of birds of the Great Lakes region have been suggested as a major causative agent for the observed adverse effects on reproduction of fish-eating birds (Giesy *et al.* 1994a). Total concentrations of PCBs in the eggs of bald eagles have been inversely correlated with reproductive success (Wiemeyer *et al.* 1984) and productivity (Bowerman 1993; Wiemeyer *et al.* 1993; Best *et al.* 1994). PCBs have been identified as a major cause of birth defects in the white-tailed sea eagle (*Haliaeetus albicilla*) in Europe (Helander 1983). It has been difficult to demonstrate a cause-effect relationship between concentrations of PCBs in bald eagle eggs and impairment of reproduction because the concentrations of PCBs are inter-correlated with the concentrations of other organochlorine toxicants, such as the DDT complex (Wiemeyer *et al.* 1984). Therefore, we have not relied solely on field regression data to derive a NOAEC. However, as the concentrations of DDE in bald eagle eggs have declined, egg mortality due to eggshell thinning has decreased. As the concentrations of DDE have decreased, the negative correlation between reproductive productivity and concentrations of DDE has become poorer, but the negative correlation between productivity and concentrations of PCBs in bald eagle eggs in the Great Lakes region has become stronger and more statistically significant (Kubiak and Best 1991). Currently, even though there is still a correlation between the concentrations of PCBs and those of DDE, the coefficient of determination (r^2) for the regression of PCBs or DDE against the reproductive productivity of bald eagles is greater for total PCBs (0.80) than for DDE (0.63) (Kubiak and Best 1991). When the effects of DDE (primarily on eggshell thinning) are removed statistically, there is still a significant inverse relationship between the concentrations of other chemicals, primarily total PCBs, and productivity of bald eagles (Colborn 1991); PCBs are thought to be responsible for the currently observed adverse effects.

The threshold concentration to maintain healthy productivity (> 1.0 young per occupied nest) for bald eagles in the presence of total PCBs has been estimated to be approximately 6.0 mg PCB/kg, ww (ppm) in the egg (Kubiak and Best 1991), which is similar to the NOAEC of 4.0 mg PCB/kg, which has been suggested by Wiemeyer *et al.* (1984). Determination of the critical effects concentration in tissues from regression analyses in field studies is limited by the effects of co-correlation and slope of the dose-response relationship (Blus 1984; Shirazi *et al.* 1988); thus, it would be desirable to compare the NOAEC estimated from regression analysis to the results of studies under more controlled conditions.

There are no controlled studies with bald eagles and few with raptors (Bowerman 1993). However, laboratory studies with a number of species of birds have demonstrated that PCBs can result in effects on the survival of bird embryos (Britton and Huston 1973) that can result in population-level effects (Giesy 1994a). The NOAEC used in our assessment was similar to the concentration for threshold effects in chicken eggs (Scott *et al.* 1975). Chronic exposure to 5 mg Aroclor® 1254/kg had no effect on the production of chickens (Platanow and Reinhart 1973). In fact, concentrations of Aroclor® 1254 as great as 40 mg/kg body weight in the diet of white leghorn chickens did not affect production (Scott 1977). Deformities were observed in white leghorn chickens when the concentration of Aroclor® 1254 reached 10 mg/kg, ww in the yolk (Tumasonis *et al.* 1973). Therefore, we feel that the concentration of 4.0 mg PCB/kg, ww in eggs is a reasonable estimate of the concentration required to cause effects in bird eggs. The NOAEC (4.0 mg/kg) (Table 1) used in our hazard assessment was derived from the regression given by Wiemeyer *et al.* (1984, 1993) (Table 1). The value selected for our hazard assessment is the same as that used by Kubiak and Best (1991), but is 10-fold greater than the value of 0.4 mg PCB/kg, ww in bald eagle eggs, which has been suggested by Ludwig *et al.* (1993a). Our hazard assessment was based on the effects on reproduction, but it should be remembered that survival of the reproducing adults is an important parameter in determining the success of bald eagle populations (Grier 1980). It is therefore possible that subtle effects of toxicants, such as PCBs, may affect adults at concentrations that are less than those required to affect egg survival.

The HQ for total PCBs was near or less than 1.0 at the upstream locations for all three of the rivers, but greater than 1.0 for the downstream locations (Table 3). The greatest HQ for upstream locations was observed on the Muskegon River. The mean HQ for all of the upstream locations was 0.6, while that for the downstream locations was 15 (Table 3). The greatest HQ at the downstream locations was observed on the Manistee and Muskegon Rivers, which run into Lake Michigan, while the least HQ was observed for the downstream location on the Au Sable River, which runs into Lake Huron. The minimum difference between the HQ from downstream to that from upstream was observed on the Muskegon and Au Sable Rivers, while the maximum of 96 was observed on the Manistee River.

When the exposure of bald eagles to total PCBs was corrected for the relative proportions of the fishes in the diet (Table 4), the trend was similar to that predicted by the mean HQ values (Tables 3 and 5). The HQ corrected for relative proportions of fishes in the diet for upstream was approximately 1.0, which indicates that there should be no effects of total PCBs on bald eagles from these areas. The mean adjusted HQ for the

Table 4. Estimated intake of total polychlorinated biphenyls (PCBs) and DDE (μg/g) by bald eagles based on the percent biomass of fish consumed in three Michigan rivers (Bowerman 1993)

Species		Sucker	Pike	Salmon[a]	Other[b]	Carp	All Fish
% in Diet		51.5	15.4	2.0	26.7	4.4	100
PCBs	-Above	0.015	0.005	0.0008	0.039	0.012	0.072
	-Below	0.206	0.154	0.056	0.512	0.264	1.192
TCDD-EQ	-Above	1.5×10^{-6}	3.7×10^{-7}	5.2×10^{-8}	8.0×10^{-7}	1.1×10^{-7}	2.8×10^{-6}
	-Below	6.2×10^{-6}	3.4×10^{-6}	1.1×10^{-6}	8.3×10^{-6}	3.1×10^{-6}	2.2×10^{-5}
DDE	-Above	0.0051	0.003	0.000	0.021	0.0026	0.032
	-Below	0.057	0.023	0.010	0.048	0.0277	0.166
Dieldrin	-Above	2.1×10^{-2}	6.2×10^{-3}	1.8×10^{-5}	4.5×10^{-4}	2.6×10^{-6}	2.7×10^{-2}
	-Below	3.9×10^{-3}	1.2×10^{-4}	1.3×10^{-3}	8.0×10^{-3}	2.6×10^{-3}	1.7×10^{-2}
Mercury	-Above	9.3×10^{-2}	2.8×10^{-2}	1.5×10^{-1}	1.5×10^{-1}	2.2×10^{-3}	2.8×10^{-1}
	-Below	9.9×10^{-2}	2.9×10^{-2}	3.7×10^{-2}	3.7×10^{-2}	1.0×10^{-2}	1.8×10^{-1}

[a] Brown Trout above; Steelhead Trout and Chinook Salmon below
[b] Percentages of fish were: Walleye, 1.5; Bowfin, *Amia calva*, 17.4; bass, *Micropterus salmoides, M dolomieu*, 4.4; bullheads, *Ictalurus*, spp., 1.8; sunfish, Family Centrarchidae, and Yellow Perch, 1.6.

Table 5. Hazard assessment of effects of concentrations of total polychlorinated biphenyls (PCBs), TCDD-EQ, DDE, dieldrin, and mercury in fish three Michigan rivers on bald eagles. The minimum and maximum concentrations of each toxicant are given for the areas above and below the dams. The hazard quotient (HQ) (concentration in each species of fish/NOAEC) are given. The threshold for effects would be one (1) toxic unit. An assessment was also done for each toxicant for each area (above or below, averaged across all rivers) for an intake of all species of fishes adjusted for the relative portion of the diet contributed by each species to the diet of bald eagles (% diet in Table 4)

Toixicant	Range	Concentration Above	Concentration Below	HQ Above	HQ Below	Ratio (Below/Above)
Total PCBs	-Max.	0.366	5.991	7.63	124	30.9
	-Min.	0.012	0.371	0.25	7.73	16.4
	-Adjusted	0.072	1.193	0.51	8.52	16.6
TCDD-EQ	-Max.	1.6×10^{-6}	1.5×10^{-5}	4.32	40	9.31
	-Min.	6.7×10^{-8}	1.7×10^{-6}	0.18	4.6	26
	-Adjusted	2.8×10^{-6}	2.2×10^{-5}	0.8	5.9	7.8
DDE	-Max.	0.026	0.121	1.08	5.04	4.6
	-Min.	0.001	0.008	0.04	0.33	8.3
	-Adjusted	0.032	0.166	0.20	1.04	5.1
Dieldrin	-Max.	2.3×10^{-2}	8.8×10^{-3}	16.4	0.63	3.8×10^{-2}
	-Min.	2.9×10^{-5}	6.0×10^{-4}	2.1×10^{-2}	4.3×10^{-2}	2.0
	-Adjusted	2.7^{-2}	1.7×10^{-2}	1.9	1.2	0.62
Mercury	-Max.	6.7×10^{-2}	0.13	0.13	0.26	2.0
	-Min.	3.5×10^{-3}	8.7×10^{-3}	7.0×10^{-3}	1.74×10^{-2}	2.5
	-Adjusted	0.28	1.8×10^{-1}	0.56	0.36	0.6

downstream locations was 25, which indicates that effects would be expected at these locations.

The results of the hazard assessment indicate that current concentrations of total PCBs in fishes of the three rivers should not be having adverse effects on bald eagles feeding on fish living upstream of the dams, but that fishes below the dams would present a significant hazard to bald eagles living along the Great Lakes shoreline or on rivers below the downstream-most dams.

This result of the hazard assessment predicts the observed productivities of bald eagles in the two areas. Field monitoring shows that the productivity of bald eagles upstream of the dams was greater than 1.0 and indicative of a healthy bald eagle population, while that of bald eagles along the shorelines of the Great Lakes or along anadromous-accessible populations was approximately 0.5, which is less than the 1.0 necessary for a healthy population and 0.7 required for a stable population (Sprunt *et al.* 1973; Kubiak and Best 1991; Bowerman 1993; Bowerman *et al.* 1994; Best *et al.* 1994). Total concentrations of weathered PCBs in addled bald eagle eggs of 83 and 98 mg PCB/kg, ww have been measured for Lakes Michigan and Huron, respectively (Kubiak and Best 1991). The greater of these two concentrations is approximately 25 times greater than the NOAEC used in our hazard assessment. Thus, there is field verification of exposure to total PCBs, which would be expected to be causing adverse, population-level effects.

TCDD-EQ

Recently, several authors have suggested that a better predictor of effects than total concentrations of PCBs, PCDDs, and PCDFs could be made from the concentrations of dioxin-like congeners that bind to the aromatic hydrocarbon receptor (A*h*-receptor) through which most of their toxic effects are proposed to be mediated (Safe 1987; Giesy *et al.* 1994a). When the concentrations of individual congeners are corrected for their relative potency, much better correlations are observed between lethality and deformities in embryos and chicks of colonial fish-eating water birds of the North American Great Lakes (Tillitt *et al.* 1992; Yamashita *et al.* 1992). The potency of individual PCDH congeners to cause toxic effects can be compared to that of the most toxic PCDH, which is 2,3,7,8-TCDD. This is done by the use of toxic equivalency factors (TEF), based on several toxic effects, including lethality, deformities, or enzyme induction (Jones *et al.* 1993a) or bioassays (Giesy *et al.* 1994).

Since bald eagles are a threatened or endangered species, there have been no controlled laboratory studies of the effects of TCDD-EQ on bald eagles, and there have been few field studies that have correlated concentrations of TCDD-EQ. Also, there have been few studies that have correlated the concentrations of TCDD-EQ in the diet or eggs of bald eagles with observed effects. Thus, a NOAEC must be derived from the effects of TCDD on surrogate species; there are a number of factors that make it difficult to predict the most appropriate NOAEC to use in a hazard assessment. When the literature on effects of TCDD-EQ on birds was surveyed, the LOAEC values were in the range of 10 ng TCDD-EQ/kg, ww in avian eggs and tissues (Giesy *et al.* 1994c). The LC-50 (concentration to be lethal to 50% of the eggs exposed) for the toxicity of PCB congener #126 as determined by egg injection studies of the eggs of the American kestrel (*Falco sparverius*) is between 40 and 70 μg/kg, ww ppb (Hoffman *et al.* 1995). The relative toxicity of PCB #126 to that of 2,3,7,8-TCDD is approximately 0.015 for avian species (Jones *et al.* 1993a, 1994; De Vito *et al.* 1993). Application of this factor to the LC-50 for PCB #126 in American kestrels results in an apparent LC-50 of between 0.6 and 1.0 ug TCDD-EQ/kg, ww in the egg. The ratio between the LOAEC and LC-50 of TCDD in white leghorn chickens is approximately 100 (Henshel *et al.* 1993). Application of this ratio to the LC-50 for lethality of American kestrel eggs results in a LOAEC of between 6 and 10 ng/kg, ww in egg. The LOAEC, based on lethality, has been reported to be 10 pg/g, ww for the chicken embryo (Henshel *et al.* 1993). If this value is divided by a 10-fold application (safety) factor to extrapolate from the LOAEC to the NOAEC for lethality, a value of 1 ng 2,3,7,8-TCDD/kg in the egg is derived. This concentration injected into chicken eggs results in a rate of 6-15% deformities. The LC-50 for wood ducks (*Aix sponsa*) has been reported to be approximately 70 ng TCDD-EQ/kg in the eggs of wood ducks (White and Setinak 1994). If this value is divided by an application factor of 10, the estimated NOAEC for eggs is estimated to be approximately 7 ng TCDD-EQ/kg. Alternatively, based on a LOAEC of 21 for the effects of TCDD-EQ on wood ducks under field conditions (White and Setinak 1994), a NOAEC of 2.1 pg TCDD-EQ/g in the egg can be estimated by using the standard 10X application factor. Based on the above information, we chose a value of 7 ng/kg, ww in egg as the LOAEC/NOAEC to be used in the hazard assessment. While on the conservative side, the value selected for the NOAEC is near the median for the NOAEC values calculated from the literature information on the toxicity of TCDD to avian species.

The NOAEC used in our HQ is similar to, but not identical to, those suggested by other workers. Our value is approximately 16-fold less than that derived by the USEPA in their guidance document for wildlife hazard assessments of the effects of 2,3,7,8-TCDD on wildlife, but it is in the range of values predicted from other species (Cook *et al.* 1993). The EPA determined a concentration of 6 ng TCDD/kg in fish that would be associated with little hazard to fish-eating birds, based on assumptions about the proportions of fish in the diet, the BMF values, and the NOAEC of 100 ng 2,3,7,8-TCDD/kg in pheasant (*Phasianus colchicus*) eggs (Nosek *et al.* 1992). Based on the BMF value of 19 used in our study, this would be equivalent to approximately 114 ng/kg in the eggs of bald eagles. In their hazard assessment of the effects of TCDD-EQ on bald eagles, Kubiak and Best (1991) used a NOAEC of 20 ng TCDD-EQ/kg in the egg, an estimate from the effects of 2,3,7,8-TCDD on the white leghorn chicken (Verret 1976). This value is three times greater than the value used in our assessment. The NOAEC determined for wood ducks under field conditions is approximately 3-fold less than our value. However, in their field study, White and Setinak (1994) did not measure the concentrations of other compounds, such as PCBs, which would likely contribute to the total TCDD-EQ. Thus, it would be expected that their value would be an underestimate of the NOAEC (overestimate of the toxicity of the measured TCDD-EQ). A dietary NOAEC of 1.5 ng TCDD-EQ/kg in the egg has been suggested to protect sensitive avian species (Ludwig *et al.* 1993). By predicting the NOAEC in eggs by use of the BMF of 19, this would correspond to a value of 28.5 ng/kg in the egg. The analysis of the potential range of NOAEC values, based on literature values and assumptions, yields a range of NOAEC values for eggs from 1 to 114 ng/kg. Thus, the value used in our hazard assessment is greater than that which would be derived from a simple application of the results with the chicken, which is a very sensitive species, but less than those based on the pheasant, which is one of the more tolerant species. The value we used is similar to that predicted for the kestrel and similar to that derived for several other species. Implicit in our choice of an NOAEC is the assumption that bald eagles are more sensitive to the effects of TCDD-EQ than are pheasants, but less sensitive than are white leghorn chickens. The value chosen is approximately in the middle of the range of NOAEC values observed in bird eggs: 10× less than the more tolerant species and 10× greater than the least tolerant. Thus, our value is conservative and protective of most species, and thus there would seem to be no need to apply a safety factor to our NOAEC to protect eagles.

The uncertainty in the BMF for accumulation of TCDD-EQ from fish to the eggs of bald eagles is not as great as that for estimates of the NOAEC. The BMF values of Braune and Norstrom (1989) for the accumulation of PCDD and PCDF from Great Lakes fishes to the eggs of fish-eating colonial water birds indicate that the BMF would be approximately 21. We have used the consensus BMF reported for the accumulation of TCDD-EQ from fish to bald eagle eggs of 19 (Kubiak and Best 1991). If this biomagnification factor is applied, a value of approximately 0.37 ng TCDD-EQ/kg, ww is obtained for the dietary NOAEC (Table 1). As will be seen, even if the NOAEC were more like one of the two extreme values, it would not change the conclusions made about the relative hazard of fish consumption above or below the dams.

The greatest uncertainty in predicting the concentration of TCDD-EQ likely to be deposited in eggs of bald eagles from eating fish would be due to the relative proportion of the diet that is fish. We derived weighted average dietary content of fishes in the diet based on measurements of the relative proportion of each species of fish in the diet (Bowerman 1993) at each of the locations for which an HQ was calculated. The predicted concentration of TCDD-EQ in the eggs would be underestimated if the eagles ate other fish-eating birds in their diet, since there would be an additional trophic magnification step. It was felt that bald eagles could take less contaminated mammals in the diet which would result in an overestimate of exposure. Thus, no corrections were made.

The HQ values for the fishes upstream of the dams were approximately 2 at all of the locations (Table 3). The mean HQ value for the downstream locations was almost 54, with the greatest HQ calculated for the downstream portion of the Muskegon River. When the HQ was adjusted for the relative proportions of fish in the diet of the bald eagles in each region, the HQ was intermediate between estimates based on the minimum or maximum concentrations for both the upstream and downstream locations (Table 5).

Concentrations of TCDD-EQ in unhatched eggs of bald eagles as great as 1,650 ng TCDD-EQ/kg, ww have been measured in eggs of bald eagles living on the shoreline of Lake Huron (Kubiak and Best 1991). This is approximately 236 times greater than the NOAEC that we used in our hazard assessment, 16.5 times greater than the NOAEC for pheasants (Nosek *et al.* 1992), and approximately 165 times greater than the NOAEC in white leghorn chicken eggs (Giesy *et al.* 1994c). Since these values were determined with the H4IIE assay used in our studies, the values are directly comparable to those reported here. Thus, current concentrations of TCDD-EQ are sufficient to be causing the observed reduction in productivity of bald eagles living along the shores of the Great Lakes or anadromous-accessible rivers (Bowerman 1993). This substantiates the hazard assessment conducted for consumption of fishes that TCDD-EQ would be a primary cause of observed adverse effects in populations of bald eagles that consume fishes from the Great Lakes. The observed increase for these bald eagle eggs is about 10 times greater than would be predicted from total concentrations of PCBs. This is due to weathering and trophic level enrichment of the TCDD-EQ, relative to total concentrations of PCBs (Giesy *et al.* 1994a). Thus, it can be concluded that at this time, TCDD-EQ is the critical contaminant in the eggs of bald eagles along the Great Lakes and that the greatest proportion of the TCDD-EQ is contributed by the non-ortho-substituted PCBs.

Mercury

It is difficult to establish a NOAEC for mercury in bald eagles. A theoretical NOAEC for Hg in the eggs of bald eagles is given as 0.5 mg Hg/kg, ww (Wiemeyer *et al.* 1984). This value was derived from a study with mallards (*Anas platyrhynchos*) fed a dietary dose of 0.5 mg Hg/kg, ww (Heinz 1979). The concentration of *p,p'*-DDE contained in eggs of bald eagles from Wiemeyer's studies, where mercury concentrations were above 0.5 mg Hg/kg, ww, were, however, greater than the *p,p'*-DDE concentration associated with greater than a 50% decrease in productivity. Thus, it would be difficult to ascribe the observed effects to mercury alone. No effects of Hg have been observed on reproduction of the white-tailed sea eagle, a species similar to the bald eagle (Helander *et al.* 1982). A theoretical concentration for effect in eggs was given as 1.0 mg Hg/kg, ww although no direct linkage to adverse effects was noted (Helander *et al.* 1982). Concentrations of Hg in feathers of white-tailed sea eagles of the Baltic ranged from 40 to 65 mg Hg/kg, ww. When concentrations of Hg are this great in the eggs of white-tailed sea eagles, they seldom hatch (Berg *et al.* 1966). It should be noted that no organochlorine pesticide analysis had been completed as of the time of publication for these data. Thus, it is likely that the observed effects on hatchability of white-tailed sea eagle eggs are due to the effects of organochlorine compounds. Subsequent reports refute the Hg/reproduction theory of Berg *et al.* (1966) and link white-tailed sea eagle reproductive problems primarily to *p,p'*-DDE and PCBs (Koivusaari *et al.* 1980; Helander *et al.* 1982). The effects of Hg on wild populations of nesting bald eagles is difficult to assess since there are nearly always organochlorine compounds present (Frenzel 1984). This is also true in the Great Lakes Basin, where *p,p'*-DDE and PCBs have been correlated with reproductive effects in bald eagles (Best *et al.* 1994) and not Hg (Bowerman *et al.* 1994b).

Hazard quotient values based on Hg were less than 1.0 for the average of all fishes from above and below the dams (Table 3). Northern pike (*Esox lucius*), known to accumulate Hg to greater concentrations than other fishes, did not contain the greatest concentrations of Hg in the rivers studied in Michigan. Interestingly, yellow perch (*Perca flavescens*) from below the dams and walleye from above the dams on the Muskegon River were the only fish with hazard quotients above 1.0. There was no difference in the mean hazard quotients among the three rivers (Table 3).

Mercury in fishes from the upstream locations would represent a greater hazard to bald eagles than that at the downstream locations. However, the HQs, which were based on a conservative estimate of the NOAEC, were not very great. Certainly, Hg is not the most critical contaminant in the fish of these river systems, but it is currently greater than the NOAEC for bald eagles if they ate exclusively several of the species of fish studied. However, the relative proportion of yellow perch and walleye in the diet of bald eagles along the three streams is small (<3% of total diet; Bowerman 1993). Since these fishes are not a large part of the diet of bald eagles, it is not probable that current concentrations of Hg are the cause of any population-level effects. Recent monitoring of Hg concentrations in bald eagle feathers (Bowerman *et al.* 1994b) found no relationship between concentrations of Hg in the adult bald eagles and reproductive performance.

Population Analysis

Bald eagle reproductive productivity within the five river basins proposed for passage of fish around hydroelectric dams was at or above the healthy level (1.0 young per occupied nest) (Sprunt *et al.* 1973), with the exception of bald eagles nesting along the Menominee and Manistee Rivers (Table 6) (Bowerman 1993). Based on the proposed passage of anadromous fish around hydroelectric dams along these five rivers, the percentage of Great Lakes breeding areas would increase from 35.2 % to 66.1% of the state bald eagle population. The mean, statewide, reproductive productivity within the state of Michigan would decline from 0.91 young per occupied nest to 0.86 (Bowerman 1993). One would expect, based on the decline in the number of young produced after fish passage, that on aver-

Table 6. Reproductive productivity of bald eagles nesting in the basins of the Au Sable, Manistee, Menominee, Muskegon, and Thunder rivers (Figure 1), and for Great Lakes and interior nest sites in Michigan, 1989–1993, in relation to an assessment of potential effects of anadromous fish passage around hydroelectric dams on these rivers

	Au Sable	Manistee	Menominee	Muskegon	Thunder
Breeding areas	89	21	151	33	63
Fledged young	110	20	124	41	68
Productivity[a]	1.24	0.95	0.82	1.24	1.08
Present Situation	Great Lakes[b]	Interior[c]	Five Rivers	Michigan Totals	% Great Lakes[b]
Breeding areas	326	337	357	1020	35.2
Fledged young	241	371	363	975	
Productivity[a]	0.74	1.10	1.02	0.91	
Predicted Situation with Fish Passage					
Breeding areas	326	337	357	1020	66.1
Fledged young	241	371	269[d]	881	
Productivity[a]	0.74	1.10	0.74	0.86	

[a]Number of fledged young divided by the total number of occupied breeding areas
[b]Breeding areas within 8.0 km of a Great Lakes shoreline or along a river accessible to Great Lakes anadromous fish runs
[c]Breeding areas not defined as Great Lakes or along the five rivers
[d]Number of fledged young produced during 1989–93 multiplied by the Great Lakes productivity rate

age, 19 fewer young would be fledged within the state related to this management technique.

It has previously been shown that bald eagles nesting along anadromous-accessible streams in Michigan produced significantly fewer young than bald eagles nesting in interior breeding areas, but were not significantly different from bald eagles nesting along the Great Lakes shorelines (Bowerman 1993; Best *et al.* 1994; Bowerman *et al.* 1994). The lesser reproduction was attributed to greater concentrations of PCBs and *p,p'*-DDE in eggs and blood of bald eagles nesting in the Great Lakes and anadromous-accessible breeding areas (Kubiak and Best 1991; Best *et al.* 1994; Bowerman 1993). It is therefore reasonable to assume that passage of Great Lakes fishes around these hydroelectric dams would cause bald eagles that are currently not exposed to these contaminants to reflect the reproductive productivity of those bald eagles that currently nest along the Great Lakes or anadromous-accessible areas. Although the reproductive productivity observed in Great Lakes breeding areas from 1989–1993 is greater than that previously reported (0.68; Bowerman 1993), this may be attributed to the influence of younger, less contaminated birds occupying newly established breeding areas that have not yet reached burdens of contaminants related to reproductive effects. During this time period, the occupied breeding areas along the Great Lakes and anadromous-accessible areas in Michigan increased from 47 to 86. Over time, bald eagles breeding along these areas have become less productive (Best *et al.* 1994; Bowerman 1993). Therefore, the analysis of reproductive effects presented here is conservative, since we cannot predict how many more Great Lakes/anadromous breeding areas will appear in the future, although the percentage of the state population within this category has increased from 12% in 1977 to 35.1% in 1993 (Bowerman 1993), and is the most rapidly increasing portion of this population. Additionally, these younger, less contaminated birds mask the effects of environmental contaminants within this population, since it takes a few years of exposure for these sub-chronic effects to become evident.

Based on this analysis, bald eagle reproductive productivity would be negatively impacted from the passage of anadromous fishes from the Great Lakes around hydroelectric dams along these five rivers. These calculations substantiate the hazard assessment conducted based on the concentrations of organochlorine compounds in fish from the lower portions of the rivers.

Uncertainties

One potential for error in the assessment would be the contribution of contaminants in non-fish species for which we did not have estimates. Bald eagles living along the coasts of the Great Lakes are known to take other fish-eating birds, such as herring gulls (*Laurus argentinus*), in their diet (Kozie and Anderson 1991). However, few of these birds would be expected in the diet of bald eagles nesting at inland locations. Therefore, the error presented by the other items in the diet would more likely result in an underestimate of the weighted HQ. Mammals taken by bald eagles are herbivores and would be expected to have lesser concentrations of the toxicants of interest than do fishes or fish-eating birds. Since the HQ values from upstream of the dams was less than that deemed to be a significant hazard to bald eagles, this error would not have any impact on the conclusions drawn. In the analysis, it was assumed that 10% of the diet would be make up of non-fish prey. Since our hazard assessment was based only on the portion of the diet that was comprised of fishes, the conclusions for the downstream locations would not differ because the HQ would be a slight underestimate of the actual values. Any PCBs or TCDD-EQ in the non-fish portion of the diet would make the HQ greater. Therefore, we feel that any errors caused by this would be small and not affect the overall conclusions from the analysis.

Conclusions

The results of the hazard assessment indicate that current concentrations of DDE, total PCBs, TCDD-EQ, or Hg in fishes of the three rivers are not having adverse effects on bald eagles living above the dams. While there might be some effects of DDE on bald eagle productivity, it would not, at this time, be the critical contaminant. Concentrations of both total PCBs and TCDD-EQ in fishes below the dams currently represent a more significant hazard to bald eagles living along the Great Lakes shoreline or on rivers below the downstream-most dams. Of these two measures of contamination, TCDD-EQ currently are the more critical of the two. Even though the majority of the TCDD-EQ found in Great Lakes fishes are contributed by the planar PCB congeners, there are additional sources of TCDD-EQ, which result in more TCDD-EQ than would be expected from Aroclors® alone. Also, weathering of Aroclor® mixtures results in an enrichment of the non-ortho-substituted congeners, which results in a PCB mixture in both fishes and bald eagle eggs that contains more TCDD-EQ than would be expected in the original Aroclor® technical mixtures. Our findings indicate that the known toxicants, total PCBs and TCDD-EQ, are occurring at sufficient concentrations in fishes and in bald eagle eggs to explain the poorer productivity observed along the coastlines and along Great Lakes-accessible sections of rivers, without the need to invoke other causes, such as weather, food availability, or other, as yet undefined, contaminants.

The results of the hazard assessment are supported by the observed productivities of bald eagles in the upstream and downstream areas. Field monitoring showed that the productivity of bald eagles above the dams is greater than 1.0 and indicative of a healthy bald eagle population, while that of bald eagles along the shorelines of the Great Lakes or along anadromous-accessible populations have productivities of approximately 0.7, which is less than the 1.0 necessary for a healthy population and 0.7 for a stable population (Sprunt *et al.* 1973; Kubiak and Best 1991; Bowerman 1993; Bowerman *et al.* 1994; Best *et al.* 1994).

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