A Review of Factors Affecting Productivity of Bald Eagles in the Great Lakes Region: Implications for Recovery

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The bald eagle (Haliaeetus leucocephalus) population in North America declined greatly after World War II due primarily to the eggshell thinning effects of p,p' -DDE, a biodegradation product of DDT. After the banning of DDT in the United States and Canada during the early 1970s, the bald eagle population started to increase. However, this population recovery has not been uniform. Eagles nesting along the shorelines of the North American Great Lakes and rivers open to spawning runs of anadromous fishes from the Great Lakes still exhibit impaired reproduction. We have explored both ecological and toxicological factors that would limit reproduction of bald eagles in the Great Lakes region. Based on our studies, the most critical factors influencing eagle populations are concentrations of environmental toxicants. While there might be some continuing effects of DDE, total PCBs and most importantly 2,3,7,8-tetrachlorodibenzo-p-dioxin equivalents (TCDD-EQ) in fishes from the Great Lakes and rivers open to spawning runs of anadromous fishes from the Great Lakes currently represent a significant hazard to bald eagles living along these shorelines or near these rivers and are most likely related to the impaired reproduction in bald eagles living there. — Environ Health Perspect 103(Suppl 4):51–59 (1995)

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Introduction

The numbers of bald eagles (Haliaeetus leucocephalus) in North America declined greatly after World War II due primarily to the eggshell thinning effects of p,p' -DDE, a biodegradation product of DDT. After the banning of DDT in the United States and Canada during the early 1970s, the numbers of bald eagles increased. After the banning of DDT in the United States and Canada during the early 1970s, the numbers of bald eagles increased. (Figure 1). This population recovery has not been uniform however. Eagles nesting along the shorelines of the North American Great Lakes and rivers open to spawning runs of anadromous fishes from the Great Lakes still exhibit impaired reproduction (4) (Figure 2). It is also apparent that adult mortality in eagles nesting near the Great Lakes is greater than expected. To understand the implications of impaired reproduction and greater mortality of adults on the recovery of this species within the Great Lakes region, we will discuss factors related to bald eagle population dynamics.

Study Area

Our studies have focused on 10 subpopulations of bald eagles within the Great Lakes region (Figure 3). These areas were defined as: the area within 8.0 km of the United States' and Canadian shorelines of the Great Lakes and rivers open to Great Lakes fish runs, hereafter referred to as anadromous accessible rivers, along a) Lake Superior, b) Lake Michigan, c) Lake Huron, and d) Lake Erie; areas in Michigan greater than 8.0 km from the shorelines of the Great Lakes and not along anadromous accessible rivers in e) the lower peninsula, f) the eastern upper peninsula east of U.S. Highway 41, and g) the western upper peninsula west of U.S. Highway 41; and h) the Chippewa National Forest, i) the Superior National Forest outside of the Boundary Waters Canoe Area, and j) Voyageurs National Park in Minnesota (Figure 3).

Factors Affecting Eagle Populations

Although many potential factors could affect bald eagle reproductive success or productivity, the three primary factors currently influencing bald eagle productivity in the Great Lakes region are habitat availability, degree of human disturbance to the nesting eagles, and the concentrations...
of environmental contaminants in the prey of the nesting eagles. Territories unoccupied by another breeding pair must include sufficient nest, perch, and roost trees, foraging areas, and a prey base in sufficient quantities to successfully raise young to fledging (5). Human disturbances must be of a type, degree, amount, and timing not to cause nest abandonment by an individual breeding pair of eagles (5). Environmental contaminants, primarily chlorinated hydrocarbons such as \( p,p'-\text{DDE} \) and PCBs, must be below concentrations associated with impaired productivity, egg lethality, or teratogenicity to produce a viable population (6–8).

Currently, we feel that the most important factor controlling bald eagle reproduction along the shorelines of the Great Lakes where eagles currently nest is the influence of environmental contaminants. We have shown that concentrations of \( p,p'-\text{DDE} \) and PCBs in both abandoned eggs and plasma of nesting eagles are correlated with impaired reproductive potential of eagles along the shorelines of Lakes Superior, Michigan, Huron, and Erie, as well as at Voyageurs National Park (4) (Figures 4, 5). Furthermore, current concentrations of both PCBs and \( p,p'-\text{DDE} \) in eggs of bald eagles are sufficiently great, based on controlled laboratory studies, to cause adverse effects in birds (9). While eggshell thinning due to \( p,p'-\text{DDE} \) may still be influencing eagle reproduction in the Great Lakes region, we have shown that the negative correlation between productivity and PCBs in bald eagle eggs has become stronger and more statistically significant than the relationship between productivity and \( p,p'-\text{DDE} \) (9). The occurrence of teratogenic effects in nestlings, which are similar to those that are known to be caused by dioxinlike coplanar compounds including polychlorinated dibenzo-p-dioxins (PCDDs), polychlorinated dibenzofurans (PCDFs), and some PCB congeners, indicates that these compounds are the most likely causative agents (10). These effects also occur in other avian species exposed to relatively great concentrations of TCDD-EQ in controlled laboratory studies (10). We have shown that concentrations of mercury (Hg) are not correlated with bald eagle productivity (11) and are less than the no observable adverse effect concentration

**Figure 1.** Numbers of occupied breeding areas of bald eagles nesting within the Great Lakes region for the period 1977 to 1993.

**Figure 2.** Productivity of bald eagles within the Great Lakes region for the period 1977 to 1993.
in the diet divided by the dietary NOAEC (Equation 1):

\[ HI = \frac{[\text{Fish}]}{\text{Dietary NOAEC}} \]  

When the HI for an adverse effect was greater than one (1 toxic unit), the concentration in the diet was expected to be sufficiently great to equal the threshold concentrations to elicit a statistically significant response. The lowest observable adverse effect concentration (LOAEC) is approximately 10-fold greater than the NOAEC. One would not expect to see population-level effects at an HI of 1.0, but, depending on the slope of the dose–response relationship, values of 10 to 20 are related to population-level effects. We used a weighted average exposure to each chemical of interest, based on the relative proportions of each species of fish in the eagles’ diets. 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.

Dose–response relationships for different end points in the bald eagle were used when available (Table 1). However, since this 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. We have tried to choose results of 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 in the hazard assessment.

Biomagnification factors (BMF) were used to estimate the magnification of toxicants from fishes to bald eagle eggs. Where possible, we calculated BMFs 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, we also used BMF values from the literature (9,12) (Table 1). Since there were not enough samples to test for significant differences in concentrations among species within or 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 that drain into the Great Lakes. Two BMFs were calculated, one for the inland population (more than 8 km from the Great Lakes shoreline) and one for bald eagles living along the shoreline. We selected an average BMF that allowed us to use a single threshold level to determine the toxic units in fish. We then calculated NOAEC of individual organochlorine pesticides, PCBs, Hg, and TCDD-EQ in fishes based on the relative dietary intake of each species of fish eaten by bald eagles (Table 1).

**Results of Hazard Assessment**

**Dieldrin.** Whereas dieldrin is known to be toxic to birds and is suspected of having caused population-level effects (13,14), it is not likely that current concentrations of dieldrin in fishes of the Great Lakes present a significant hazard to bald eagles. The NOAEC used in the hazard assessment is conservative and based on the regression of Wiemeyer et al. (6), who relates dieldrin concentrations in eggs to productivities of individual pairs of bald eagles (Table 1). However, the authors of that study state that “while dieldrin concentrations greater than 1.0 \(\mu\)g/g fresh wet in eggs cannot be ruled out as having an effect on reproduction, the major effect of dieldrin was related to adult survival” (6). The apparent association of dieldrin concentrations with productivity of bald eagles is most likely an artifact due to cocorrelation of the concentrations of dieldrin with those of total PCBs and the DDT complex (6). When considered with the dose–response
relationships obtained in controlled laboratory studies of other avian species (15), we conclude that the correlation is most likely spurious and not indicative of actual toxicity of dieldrin at the concentrations observed. Thus, it was concluded that the current concentrations of dieldrin in fishes above the dams are well below the concentration that would be expected to cause any adverse effects. Dieldrin concentrations below the dams are slightly greater than the NOAEC, but are probably not sufficiently elevated to cause any population-level adverse effects (Figure 7). Dieldrin is probably not currently considered to be the critical contaminant limiting the distribution or productivity of bald eagles in Michigan.

DDE. The effects of $p,p'$-DDE on bald eagle reproduction have been correlated in field studies (6). However, the concentrations of DDE and total PCBs were significantly correlated, and separation of the effects of DDE from co-occurring toxicants such as PCBs is difficult (3,6). In laboratory studies, DDE has been linked to eggshell thinning in several species of birds (16–23). Therefore, we have compared the NOAEC predicted from a regression analysis of the effects of DDE on eggs in the field (6) with values from similar species and to the results of controlled laboratory studies where cocorrelation did not confound the analysis.

As concentrations of DDE in the environment have declined, populations of bald eagles have increased (1,2). Although populations of bald eagles seem to be doing well at the interior locations in Michigan where they do not eat fish from the Great Lakes or anadromous accessible rivers, there is some reason for concern; it has been projected that the concentrations of DDE need to decrease to almost zero before there is no predicted adverse effect (3). Current concentrations of DDE in several populations of raptors may still be limiting reproductive success (24).

Currently, concentrations of DDE in bald eagle eggs collected from two of four breeding areas along the shores of Lakes Michigan and Huron exceed the 15.0 µg/g of $p,p'$-DDE associated with a 75% reduction in productivity (6). However, the current observed concentrations of DDE in prey from these areas are not greatly in excess of the NOAEC. This supports the results of the hazard assessment, which indicates there should still be some effects of DDE on the reproduction of bald eagles due to eggshell thinning (Figure 7).

Total PCBs. 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 productivity of fish-eating birds (25). Total concentrations of PCBs in the eggs of bald eagles have been inversely correlated with productivity (4,6,26–31). PCBs have also been identified as a major cause of birth defects in the white-tailed sea eagle (Haliaeetus albicilla) in Europe (32). It has been difficult to demonstrate a cause–effect relationship between concentrations of PCBs in bald eagle eggs and impairment of productivity because the concentrations of PCBs are always cocorrelated with the concentrations of other organochlorine toxicants, such as the DDT complex (3,6,28).

However, as the concentrations of DDE in bald eagle eggs have declined, egg mortality due to eggshell thinning has decreased. Current concentrations of DDE are less than that thought to be necessary to cause a critical degree of eggshell thinning. As the concentrations of DDE have decreased, the negative correlation between productivity and concentrations of DDE has become poorer ($r^2 = 0.63)$, 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 ($r^2 = 0.80$) (9). 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 (3,6,9). These other chemicals are thought to be responsible for most of the currently observed adverse effects.

The threshold egg concentration of PCBs to maintain healthy bald eagle productivity (>1.0 young per occupied nest), based on analysis of samples from Michigan and Ohio, has been estimated to be approximately 6.0 mg PCB/kg ww (ppm) (4)). This value is similar to the NOAEC of 4.0 mg PCB/kg that has been suggested based on regression analysis of samples throughout continental North America (6). Determination of the critical concentration for effects in eggs from regression analyses in field studies is limited by the statistical influence of cocorrelation with other compounds and the slope of the dose–response relationship (33,34).

Ideally, these values for NOAEC estimated from regression analysis should be compared to the results from studies under more controlled conditions. There are no controlled studies of PCBs with bald eagles and few with other raptors (25). Laboratory studies with a number of species of birds have demonstrated that

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Table 1. Hazard assessment of concentrations of total PCBs and DDE in bald eagles from three river systems, lower peninsula of Michigan. NOAEC and biomagnification factors of total PCBs and DDE and organochlorine pesticides in fish.

| Toxicant       | Total PCBs | $p,p'$-DDE | Dieldrin | TCDD-EQ | Mercury |
|----------------|------------|------------|----------|---------|---------|
|                | Egg lethality | Eggshell thinning | Egg lethality | Egg lethality | Egg lethality |
| NOAEC, mg/kg egg | 4.0$^a$ | 3.5$^b$ | 0.1$^b$ | 7x10$^{-6}$ | 0.5$^b$ |
| Dietary NOAEC, mg/kg fish | 0.14 | 0.16 | 1.4x10$^{-2}$ | 3.7x10$^{-7}$ | 0.5 |
| BMF, fish to egg | 28$^d$ | 22$^d$ | 7$^e$ | 19$^f$ | 1$^f$ |
| Concentration in fish shoreline, mg/kg | 2.1 | 0.3 | 3.3x10$^{-2}$ | 2.0x10$^{-5}$ | 0.2 |
| Hazard index, shoreline | 15 | 1.9 | 2.4 | 54 | 0.3 |
| Concentration in fish interior, mg/kg | 0.2 | 3.5x10$^{-2}$ | 6.7x10$^{-4}$ | 0.7 | 0.3 |
| Hazard index, interior | 1.3 | 0.22 | 0.06 | 1.9 | 0.7 |

$^a$From Wiemeyer et al. (31). $^b$From Wiemeyer et al. (6). $^c$From White and Setinak (48). $^d$Calculated from concentrations in fish and bald eagle eggs from inland or coastal areas. $^f$From Braune and Norstrom (12). $^g$From Kubiak and Best (9).
PCB exposure can result in effects on the survival of bird embryos (6,35) and result in population-level effects (25). The NOAEC used in our assessment was similar to the concentration for threshold effects in chicken eggs (36). Chronic exposure to 5 mg/kg of Aroclor 1254 in the diet had no effect on the productivity of chickens (37). In fact, concentrations of Aroclor 1254 as great as 40 mg/kg, in the diet of white leghorn chickens did not affect productivity (38). Deformities were observed in white leghorn chickens when the concentration of Aroclor 1254 reached 10 mg/kg, ww, in the yolk (39). Therefore, the concentration of 4.0 mg PCB/kg, ww, in eggs is a reasonable estimate of the concentration that causes effects in bald eagle eggs. The NOAEC (4.0 mg/kg, ww; Table 1) used in our hazard assessment was derived from the regression given by Wiemeyer et al. (6,31) (Table 1). The value that we have selected for our hazard assessment is the same as that used by Kubiak and Best (9) but is 10-fold greater than the value of 0.4 mg PCB/kg, ww, in bald eagle eggs, as suggested by Ludwig et al. (40). We have based our hazard assessment on reproductive effects, but it should be remembered that survival of the adults is also an important parameter in determining the success of bald eagle populations (41). It is possible that toxicants such as PCBs may affect adults in subtle ways at concentrations less than those required to affect egg survival.

The results of the hazard assessment indicate that current concentrations of total PCBs in fishes of the three rivers (Figure 6) upstream of the barrier dams should not be having adverse effects on bald eagles but that anadromous fishes below the lowest dams would present a significant hazard to bald eagles living near rivers below the dams and along the Great Lakes shoreline (Figure 7). This hazard assessment predicts the observed productivities of bald eagles in the two areas. Our field monitoring indicates that the productivity of bald eagles upstream of the dams was greater than 1.0 young per occupied nest and indicated a healthy bald eagle population. Bald eagles along the shorelines of the Great Lakes or along anadromous-accessible rivers had productivities of approximately 0.67 young per occupied nest, which is less than the 1.0 necessary for a healthy population and the 0.7 required for a stable population (4,9,25–27). Total concentrations of weathered PCBs in added bald eagle eggs of 83 and 98 mg PCB/kg, ww, have been measured for Lakes Michigan and Huron, respectively (9). These concentrations are approximately 20 times greater than the NOAEC used in our hazard assessment and indicate that exposure of these populations to total PCBs is causing adverse, population-level effects.

**TCDD-EQ.** The polychlorinated diaromatic hydrocarbons that can attain a planar configuration and cause effects similar to those of 2,3,7,8-TCDD have been demonstrated through both laboratory and field studies (8,24,42) to be the current critical factors that could cause the effects observed in most wildlife populations, especially the deformities of bald eagles (9). TCDD-EQ can be contributed by a number of compounds, including the PCDDs, PCDFs, and planar PCBs (3). In the Great Lakes, PCB congeners contribute a great proportion of the TCDD-EQ and may be responsible for the observed toxicity. For these reasons, we conducted a hazard assessment of the potential effects of TCDD-EQ on the bald eagle populations living along the three rivers (Figure 6).

Because bald eagles are a threatened or endangered species, there have been no controlled laboratory studies of the effects of TCDD-EQ on bald eagles. Similarly, there have been few field studies that have correlated concentrations of TCDD-EQ in the diet or eggs of bald eagles with observed effects. We have therefore derived a range of values for the LOAEC and NOAEC based on the studies of the effects of TCDD and dioxinlike compounds on surrogate species to calculate an HI. Published LOAEC values were in the range of 10 ng TCDD-EQ/kg, ww, in avian eggs (25). The LC$_{50}$ (concentration to be lethal to 50% of the fish 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 (43). The relative toxicity of PCB #126 to that of 2,3,7,8-TCDD is approximately 0.015 for avian species (44,45). Application of this factor to the LC$_{50}$ for PCB #126 in American kestrels results in a LC$_{50}$ of between 0.6 and 1.0 µg TCDD-EQ/kg, ww, in the egg. The ratio between the LOAEC value and LC$_{50}$ of TCDD in white leghorn chicken eggs is approximately 100 (46). Application of this ratio to the LC$_{50}$ for lethality of American kestrel eggs results in an LOAEC value for TCDD of between 6 and 10 ng/kg, ww, in egg. The LOAEC value for TCDD, based on lethality has been reported to be 10 pg/g, ww, for the chicken embryo (47). 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 resulted in 6 to 15% deformities. The LC$_{50}$ for wood ducks has been reported to be approximately 70 ng TCDD-EQ/kg, ww, in their eggs (48). If this value is divided by an application factor of 10, the estimated NOAEC for eggs is approximately 7 ng TCDD-EQ/kg. Alternatively, based on an LOAEC value of 21 pg TCDD/g for the effects of TCDD-EQ on wood ducks under field conditions (48), an NOAEC value of 2.1 pg/g TCDD-EQ in the egg can be estimated by using the standard 10 times 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 on the toxicity of TCDD to avian species.

The NOAEC value used in our HI is similar, but not identical, to those suggested by other workers. Our value is approximately 16-fold less than that derived by the U.S. EPA in their guidance document for hazard assessments of the effects of 2,3,7,8-TCDD on wildlife, but is in the range of values predicted from studies of other species (42). The U.S. EPA determined that a concentration of 6 ng TCDD/kg in fish would be associated with little hazard to fish-eating birds, based on assumptions about the proportions of fish in the diet and the BMF values and the NOAEC value of 1.0 ng 2,3,7,8-TCDD/kg in the ring-necked pheasant (Phasianus colchicus) eggs (49). Based on the BMF value of 19 that we have used in our study, this would be equivalent to approximately 114 ng/kg in the eggs of bald eagles. In a hazard assessment of the effects of TCDD-EQ on bald eagles, Kubiak and Best (9) used an NOAEC value of 20 ng TCDD-EQ/kg in the egg that was estimated from the effects of 2,3,7,8-TCDD on the white leghorn chicken (50). This value is three times greater than the value we have 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 Setina (48) 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 value of 1.5 ng TCDD-EQ/kg in the egg has been suggested to protect sensitive avian species (40). By predicting the NOAEC in eggs using 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 from 1 to 114 ng/kg in the egg. The value we have used in our hazard assessment is greater than the value derived from a simple application of the results with the chicken, a very sensitive species, but less than values based on the pheasant, which seems to be one of the more tolerant species. The value we have chosen to use is similar to that predicted for the kestrel and similar to that derived for several other species. Implicit in our choice of an NOAEC value is the assumption that bald eagles are more sensitive to the effects of TCDD-EQ than pheasants but are less sensitive than white leghorn chickens. The value we have chosen is approximately in the middle of the range of NOAEC values observed in bird eggs: 10 times less than the more tolerant species and 10 times greater than the least tolerant. Thus, our value can be assumed to be conservative and protective of most species, and there would seem to be no need to apply a safety factor to our NOAEC value 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 Giesy et al. (10) for the accumulation of PCDD and PCDF from Great Lakes fishes to the eggs of fish-eating colonial water birds indicate that the BMF value was approximately 21. We have used the consensus BMF value of 19 reported for the accumulation of TCDD-EQ from fish to bald eagle eggs (9). If this biomagnification factor is applied, a value of approximately 0.37 ng TCDD-EQ/kg, ww, is obtained for the dietary NOAEC value (Table 1). Even if the NOAEC value 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 is due to the relative proportion of fish in the diet. We derived weighted average dietary content of fishes in the diet that were based on measurements of the relative proportion of each species of fish in the diet at each of the locations for which an HI was calculated. The predicted concentration of TCDD-EQ in the eggs can be underestimated if the eagles ate a great number of other fish-eating birds such as gulls because there is an additional trophic magnification step. Bald eagles can also take less contaminated mammals in the diet, which results in an overestimate of exposure. We did not correct for either of these eventualities.

Concentrations of TCDD-EQ in eggs of bald eagles as great as 1650 ng TCDD-EQ/kg, ww, have been measured in eggs of bald eagles living on the shoreline of Lake Huron (9). This is approximately 236 times greater than the NOAEC values that we used in our hazard assessment, 16.5 times greater than the NOAEC values for pheasants (49), and approximately 165 times greater than the NOAEC values in white leghorn chicken eggs (30). Since these values were determined with the H41IE assay used in our studies (25), the values are directly comparable to those reported here. Thus, current concentrations of TCDD-EQ are sufficient to cause the observed reduction in productivity of bald eagles living along the shores of the Great Lakes or anadromous-accessible rivers (25). This substantiates the hazard assessment conducted for consumption of fishes that TCDD-EQ is the primary cause of the observed adverse effects in populations of bald eagles that consume fishes from the Great Lakes (10) (Figure 7). The observed exceedance of 230 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. 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 (25).

Mercury. It is difficult to establish an NOAEC value for the adverse effects of mercury on bald eagles. A theoretical NOAEC value of 0.5 mg Hg/kg in the eggs of bald eagles has been proposed (6). This value was derived from a study in which mallards (Anas platyrhynchos) were fed a dietary dose of 0.5 mg Hg/kg (51). The concentration of p,p'-DDE contained in eggs of bald eagles from studies reported by Wiemeyer et al. (6), in which mercury concentrations were above 0.5 ppm, were greater than the p,p'-DDE concentration associated with greater than a 50% decline 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 (52). A theoretical concentration for effect in eggs was given as 1.0 mg Hg/kg in eggs, although no direct link to adverse effects was noted (52). When concentrations of Hg in feathers of white-tailed sea eagles of the Baltic ranged from 40 to 65 mg Hg/kg, eggs from these areas were observed to seldom hatch (53). It should be noted that no organochlorine pesticide analysis had been completed at the time of publication for those data. It is likely that the observed effects on hatchability of white-tailed sea eagle eggs were due to the effects of organochlorine compounds. Subsequent reports refute the Hg/reproduction theory of Berg et al. (53) and link white-tailed sea eagle reproductive problems primarily to p,p'-DDE and PCBs (52,54). The effects of Hg on wild populations of nesting bald eagles is difficult to assess because there are nearly always organochlorine compounds present (55). This is also true in the Great Lakes Basin where p,p'-DDE and PCBs have been correlated with reproductive effects in bald eagles (4).

Mercury in fishes upstream of the dams represents a greater hazard to bald eagles than at the downstream locations. The HIs, based on a conservative estimate of the NOAEC value, were not very great (Figure 7). 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 are, exclusively, several of the species of fish studied, with yellow perch (Perca flavescens) and walleye (Stizostedion vitreum) having the greatest Hg concentrations and presumably posing the greatest risk to eagles. However, the relative proportion of these two fish species in the diet of bald eagles along the three streams was small (<3% of total diet). Because these fishes are not a large part of the diet of bald eagles, it is not likely that current concentrations of Hg are the cause of any population-level effects.

Implications for Continued Recovery

Models used to predict bald eagle population dynamics predict that decreases in productivity are less critical than increases in adult mortality for declines in population (43). This is unique to long-lived species with delayed adult maturity.
Modeling efforts have failed to use the scenario of both increased adult mortality and depressed productivity occurring simultaneously in a region with significant immigration. This appears to be the scenario that is currently occurring among eagles who nest along the shorelines of the Great Lakes and anadromous-accessible rivers. Eagles nesting along the Great Lakes continue to show chronic effects of organochlorine pollutants, primarily p,p'-DDE and PCBs. These effects include impaired productivity, adult mortality, and teratogenicity (4,26,56). Eagles nesting in interior regions are less contaminated and exhibit generally greater productivity. In areas of great density such as the Chippewa National Forest, density-dependent declines in reproductive success have been noted. These interior areas are the source of relatively uncontaminated bald eagles that are supplying the "population sink" along the Great Lakes. The Great Lakes have been the last area where bald eagle recovery has occurred; however, the potential for occupancy along the shorelines and anadromous-accessible rivers is great, based on the availability of unoccupied nesting habitat. In Michigan, as the numbers of breeding areas have increased from 85 breeding areas in 1977 to 246 in 1993, the percentage of Great Lakes shoreline and anadromous-accessible river breeding areas has increased from 14 (n = 12) to 35% (n = 86) during the same time period. It is important that essential habitat be protected in the interior areas and managed to control human activities near breeding areas during critical periods of eagle nesting. This is necessary to maintain the greater productivity of these interior areas so that the influence of the Great Lakes "population sink" will not jeopardize the regional recovery of this species.

The importance of a vulnerable, relatively uncontaminated forage base for bald eagles during the breeding season is imperative for successful reproduction. Effects of environmental contaminants on bald eagle productivity are well known (4,6,11,26,30), but other aspects of habitat requirements need to be considered. Management techniques that control populations of prey species used by bald eagles need to take into account the effect that increases or decreases in utilization of contaminated species will have on the bald eagle's reproductive success. The need to maintain populations of primarily warmwater fish in interior foraging areas for inland eagles in the Midwest is imperative for maintaining the continuing recovery of this species.

The fact that concentrations of PCBs and DDT remain at concentrations that are still associated with lesser average productivities presents continuing management issues, even though production of these compounds has ceased in North America, and concentrations of most halogenated hydrocarbons in the prey of eagles are decreasing in the Great Lakes Region (25). Current concentrations of both PCBs, p,p'-DDE and TCDD-EQ are sufficiently great to cause adverse effects in nesting bald eagles feeding on the Great Lakes food web (6,10,25,57) (Figure 7). Our results verify that poor productivity of eagles is inversely correlated with exposure to PCBs, TCDD-EQ, and p,p'-DDE but not with mercury (11). Furthermore, we have observed congenital deformities in bald eagle nestlings (56). Developmental deformities have been observed in the populations in which the greatest concentrations of PCBs have been found in the blood of nesting eagles. The results of laboratory and field studies indicate that the lethality of and deformities in embryos of colonial, fish-eating water birds of the Great Lakes are due to the toxic effects of multiple compounds, primarily TCDD-EQ contributed by coplanar PCBs, PCDDs, and PCDFs. These compounds express their effects through a common mode of action, the Ah receptor (25). The concentration of total PCBs and TCDD-EQ (58), converted from congener specific data, in two added bald eagle eggs collected near Lakes Michigan and Huron were 83 and 98 µg/g total PCBs and 21,369 and 30,894 pg/g as TCDD-EQ, respectively (58). Currently, TCDD-EQ, contributed primarily from coplanar PCBs, seems to be the critical toxicant limiting bald eagle reproduction. Concentrations of TCDD-EQ in bald eagle eggs exceed known effect levels in poultry experiments, either by total PCB concentration or by conversion of individual PCB congeners (9,37,58).

Our results suggest that exposure of bald eagles to Great Lakes fishes should be minimized. It would be inappropriate to use hacking programs to reestablish populations of eagles or improve their genetic diversity along the Great Lakes shoreline, especially for Lakes Erie and Ontario, with the current concentrations of organochlorine contaminants in their potential prey. Management practices that increase the potential exposure of eagles to chlorinated hydrocarbons in Great Lakes fishes, e.g., passage of fishes around dams on tributaries to Lakes Michigan, Huron, and Erie, could have adverse effects on productivity of bald eagles in regions that currently produce sufficient numbers to act as a source of eagles to colonize other areas. Only by maintaining a readily available, relatively uncontaminated food source for eagles during the breeding season can we continue to experience the population recovery of this species in the Midwest.

Summary

By far the most important factor in controlling current bald eagle reproduction along the shorelines of the Great Lakes is the influence of environmental contaminants. We have shown that p,p'-DDE and TCDD-EQ from PCBs are correlated with impaired reproductive success of eagles along the shorelines of Lakes Superior, Michigan, Huron, and Erie, as well as at Voyageurs National Park. We have shown further that concentrations of mercury are not correlated with bald eagle productivity in the Great Lakes region.

The results of the hazard assessment indicate that current concentrations of DDE, total PCBs, and TCDD-EQ in fishes continue to have adverse effects on bald eagles living below the barrier dams on the three rivers but should not have adverse effects on bald eagles living in interior areas above these dams. While there might be some effects of DDE on bald eagle productivity, it is not at this time the critical contaminant. Concentrations of both total PCBs and TCDD-EQ in fishes below the dams currently represent a 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, currently TCDD-EQ is the more critical. Even though the majority of the TCDD-EQ found in Great Lakes fishes is contributed by the planar PCB congeners, there are additional sources of TCDD-EQ that result in more TCDD-EQ than would be expected from technical Aroclors alone. Weathering of Aroclor mixtures results in an enrichment of the non-ortho-substituted congeners and 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 toxics, total PCBs and TCDD-EQ, are both occurring at sufficient concentrations in fishes and in bald eagle eggs to explain the poorer productivity observed in eagles that nest along the shorelines of the Great Lakes and along anadromous-accessible rivers, without the need to invoke other causes such as weather, food.
availability, or other, as yet undefined, contaminants. The assimilative capacity of the Great Lakes has been exceeded, and no additional loadings of compounds that can contribute to the total concentrations of TCDD-EQ should be allowed.

The results of the hazard assessment are supported by the observed productivities of bald eagles in the upstream and downstream areas. Bald eagle populations throughout the Midwest have experienced a steady increase in breeding pairs throughout 1977 to 1993. However, productivity has not been uniform throughout the study area. Bald eagles nesting along the Great Lakes shoreline and at Voyageurs National Park were significantly less productive than those from interior areas of Michigan and the Chippewa and Superior National Forests in Minnesota.

Availability of physical habitat does not seem to be limiting expansion of the bald eagle population along the upper Great Lakes shorelines. Bald eagles are restricted from some areas due to human disturbance or physical structure of the habitat. There are still areas deemed to be suitable nesting habitat, which are currently unoccupied by bald eagles. This is especially true of the northern forested regions that are less populated by humans. Suitable habitat along Lake Erie is scarce and may be a limiting factor in the near future.

As the bald eagle continues to reoccupy areas where they were extirpated during the 1950s and 1960s, differential effects of productivity could become even more pronounced. Density-dependent factors will continue to cause eagles from the more interior areas, where more eagles are fledged than is necessary to maintain a stable age distribution, to reoccupy the Great Lakes shorelines. This is already occurring, as the Great Lakes subpopulation has the greatest growth rate in terms of numbers of new breeding areas established. Additional investigation into the dynamics of these populations is needed to monitor the recovery of this species and to compare areas with exposure to greater concentrations of organochlorine compounds with more pristine areas. The effect of differential adult turnover along the Great Lakes shoreline also needs to be understood before a population model of the region can be produced and verified. Although the number of bald eagles in the Great Lakes Basin and adjacent areas has continued to increase as the effects of p,p'DDE have subsided, the carrying capacity of the region is still uncertain.

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