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White-Tailed Eagles (*Haliaeetus albicilla*) in Schleswig-Holstein No Longer Endangered by Organochlorines

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Populations of Baltic sea eagles (*Haliaeetus albicilla*) decreased in many European countries during the period of 1900 -1960 (Kollmann et al. 2002; Mizera 2003). In addition to anthropogenic disturbances organochlorine compounds like DDT and PCB`s have been determined to be influencing factors. Fortunately, this trend has changed and since the mid 1980`s populations have increased slowly. Conservation programs helped to establish new populations and this success is also well documented in Schleswig-Holstein (Struwe-Juhl 2002). While we have had some knowledge about the contamination status of eagle eggs before 1988, information for the last 15 years, as the eagle population has increased, has not been available. In this paper we document the current status of HCB, DDT and PCB residues in eagle eggs and clarify their role and importance in the population dynamics in the Schleswig-Holstein eagle population.

MATERIALS AND METHODS

Since 1969 eggs from sea eagles *(Haliaeetus albicilla)*, nesting in Schleswig-Holstein, North Germany, that failed to hatch (n=40), have been collected and analyzed for organochlorine compounds (OC). Several authors have presented their results for different time periods: Koeman et al. 1972 (n=8; period: 1969-71), Conrad, 1981 (n=5: 1975-76), Baum (pers. comm.) (n=6: 1980-81) and Rimkus (pers. comm.) (n=9: 1984-88). We analyzed eggs from 1992-2001 (n=12) and compared the following three time periods: 1969-76, 1980-88, and 1992-01. Our method has been published previously (Scharenberg 2001, 1991) and should be mentioned in brief: the content of frozen eggs was dried at 45° C until constant weight was achieved. After homogenization, 0.5 g of dry material from each egg was extracted with 30 ml hexane:acetone (1:1) for 30 min. at 80° C in a microwave oven. The lipid extract was cleaned on a column filled with florisil (7g) and silicagel (1g) using a mixture of hexane/dichlor-methane (4:1). After evaporation, the sample was measured by an Electron Capture Detector using Capillary Gas Chromatography (column SE 54) and was compared to commercial single PCB congeners, to 4,4′-DDT , 4,4′-DDE, 4,4′-DDD (DDE contributed 95% to Σ-DDT) and HCB. The detection limit was 0.01 mg/kg lipid weight.
To compare this investigation to older ones we calculated the Σ-PCB by addition of PCB (IUPAC #) 28, 52, 101, 138, 153 and 180 (assuming that the contribution

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of # 28 and 52 is very small) and multiplied this sum ba a factor of 1.6; both findings were demonstrated in a previous investigation (Scharenberg 2001). For the conversion of results from fresh (fw) into lipid weight (lw) we used a factor of 17 (Bezzel and Prinzinger 1990).

The age of the females was determined by moulted wing feathers, ringed birds, and controlling nestsites every year. We define productivity as the number of fledged young per year and nestsite. The eggshell-index (see Ratcliff, 1967) was determined by the following formula: weight (mg) / length * width (mm).
For statistical analysis we used the nonparametric one tailed Mann-Whitney-U-Test or Kruskal-Wallace. „P-values" < 0.05 have been considered statistically significant.

RESULTS AND DISCUSSION

As shown in Figure 1 the mean concentrations of organochlorine compounds (OC) declined from 1969 to 2001, comparing three time periods.

PCB reduction is about 4 times from 1969/76 to 1992/01. The reduction between 1969/76 and 1980/88 was statistically not significant. A maximum was measured in 1975 with a concentration of 2280 mg/kg, while the maxium since 1992 was 10 times lower and residues are quite constant on a level of around 120 mg/kg lw. The last year of PCB-production in Germany was 1982. Today we are still measuring residues, because of the persistence of PCB especially in aquatic systems, probably by sedimentation and resuspension (Scharenberg 2001) and because of existing waste deposits.

Due to the ban of DDT in Schleswig-Holstein in 1972 the residues in eggs declined between the first two time periods (Figure 1). DDT had been in use in the former German Democatic Republic until 1988 and therefore we can remark a second significant reduction between the last two time periods. A transboundary transport might have taken place between Schleswig-Holstein and Mecklenburg-Vorpommern, where DDT was in use in the agriculture until 1988. As a result, a maxium could be detected for 1975 (1365 mg/kg) while no concentration after 1992 exceeded 60 mg/kg.

The most pronounced reduction was measured for HCB, which declined 23 times between the first and the last compared time periods (Figure 1). Since 1988 the use of HCB as a pesticide has been forbidden in Germany as well as in many other European countries. A maximum was reached in 1969 (60 mg/kg) and since 1988 only in one sample the concentration exceeded 3 mg/kg.

Within clutches (n=9) we could detect a slight variability in the residues, but no tendency related to egg position. In relation to the higher contaminated egg we found in the lower contaminated eggs 78% for HCB, 86% for DDT and 80% for PCB. These differences are low and each egg of a clutch can be accepted as an indicator for the contamination status of this nestsite/female. The variation co-efficient within clutches and between clutches in a study of Helander et al. (2002)

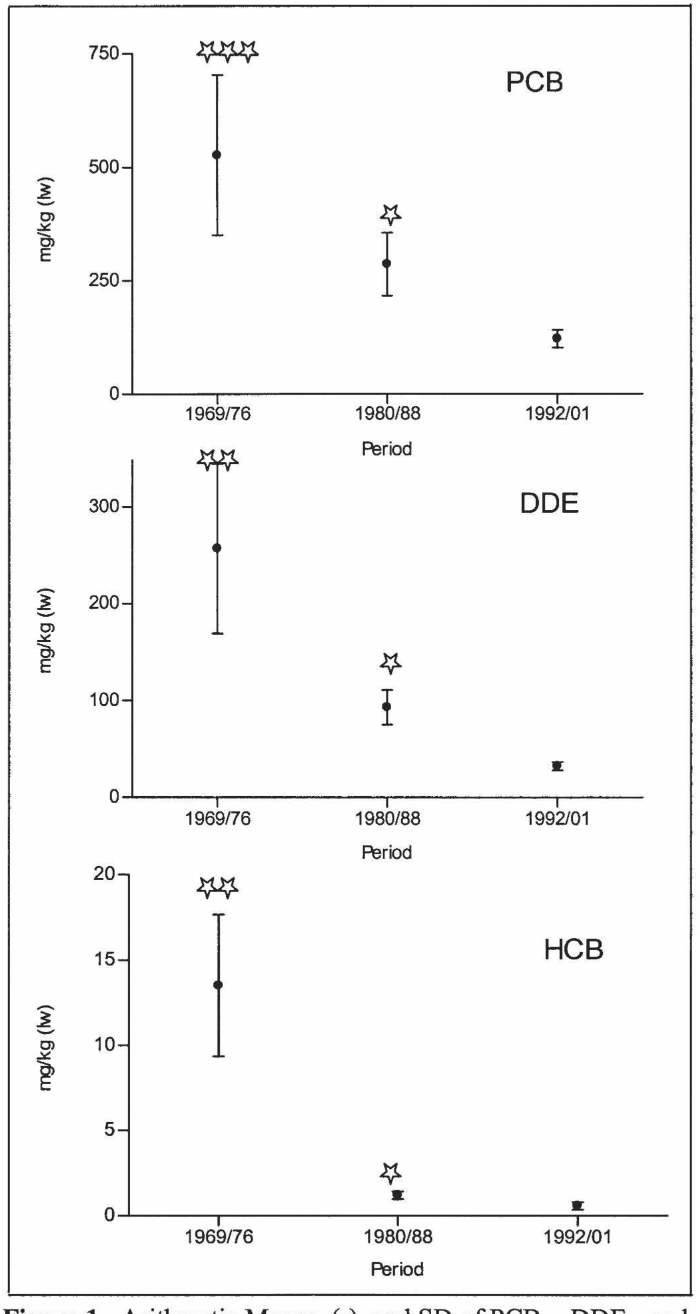

Figure 1. Arithmetic Means (•) and SD of PCB-, DDE-, and HCB-residues in eagle eggs from Schleswig-Holstein, sampled during 3 periods: 1969-76, n = 13; 1980-88, n = 15; 1992-2001, n = 12. Significant differences ($p < 0.05$; see method) between time periods :* = 1980/88 vs 1992/01; ** = 1969/76 vs 1980/88 and 1992/01; *** = 1969/76 vs 1992/01

showed, that one eagle egg can represent the clutch contamination representatively, confirming our findings.

Residues in goshawk eggs (*Accipiter gentiles*; Schleswig-Holstein) were lower. Between 1988 – 2002 the average concentration of PCB was 28 mg/kg (lw) (Scharenberg and Looft, 2004). But in comparison to the Swedish coast region the PCB`s in eagle eggs are low (Table 1) . The trend of declining contaminations is evident in other eagle populations, too (see Donaldson et al. 1999).

Representing 9 different years we collected 14 eggs from the same female at one nestsite between 1969 and 1992. Residues declined within this time span. No age accumulation took place, probably because of the overall reduction in PCB availability. The breeding success of this bird was 0.7 fledged young/year during 24 years. The average residues for HCB, DDT and PCB were: 1969-71: 29/55/638 (mg/kg lw) resp. and 1972-1988: 6/110/315 (mg/kg lw). These values were similar to the average values for all birds (see Figure 1). Further correlations between residues and locations or age of females were tested but no significant dependencies could be detected.

In a Baltic coast population of white-tailed eagles in Sweden the PCB residues in eggs declined from the early 1980s to the late 1990s about 60%, from around 1000 mg/kg (lw) to 390 mg/kg (see Table 1). The DDE reduction was 4 times from 400 to 110 mg/kg lw. In contrast to residues the productivity increased (Helander et al. 2002). No such dramaticly declining residues could be detected in eagle eggs from Northern Sweden (see Table 1). Comparably low productivity could be related to human disturbance, weather and food shortage but not to OC.

Struwe-Juhl (2003) discussed the breeding success of the Schleswig-Holstein eagle population. The number of fledged young per pair was much lower between 1965-74 (x = 0.4) than between 1975-99 (x = 1.25). Since 1975 successfull pairs (at least one fledgling per nest) have increased as well as their individual success (Table 2). More older female birds were breeding during 1965-74 and normally older ones are more successful. Therefore we assume that high DDE contamination is the main reason for the low individual success in reproduction during the earlier time period. The overall number of young fledged per year is negatively correlated with declining DDE and PCB residues. We will discuss LOEC (lowest observed effect concentrations) later.

Constant sufficient reproduction to increasing population in a US bald eagle population between 1982-96 was correlated to egg residues of DDE and PCB of 3 mg/kg ww (corresponding to 60 mg/kg lw). This population showed significant lower OC levels than other populations in the US (Donaldson et al. 1999). Increasing productivity of eagles in Finland has been mentioned by Koivusaari et al. (1980); the authors assuming declining DDE residues as a reason, while PCB`s remained constant.

Besides productivity, eggshell parameters can be influenced by OC. The correlation between shell-thickness and shell-index was tested in 11 eggs showing a significant correlation (Pearsons $r = 0.81$; $p = 0.002$ and $R^2 = 0.66$) in

Table 1. Residues of DDE and PCB in white-tailed eagle eggs from Sweden as well as productivity in the investigated populations.

Region	*Period*	*DDE*	*PCB*	*Productivity*[1)]	*Author*
Sweden: Baltic coast	1980/84 (n = 18)	400	1000	0,35 (n = 255)	Helander et al. (2002)
	1985/89 (n =31)	250	740	0,53 (n = 292)	
	1990/94 (n = 17)	170	540	0,82 (n = 411)	
	1995/97 (n = 5)	110	390	1,10 n. m.	
Sweden: Lapland	1985/89 (n = 5)	33	130	0,68 (n = 127)	
	1990/94 (n = 10)	20	76	0,69 (n = 155)	

1) Productivity given as mean number of young per year and pair; n is the number of all controlled nests during the mentioned period. (n.m. = not mentioned). Values (geometric means) are given in mg/kg lw.

agreement with findings from Helander et al. (2002). We could determine the shell-index of additional 17 eggs (1975-99) to be an average of 2.85 ± 0.18. The lowest index was 2.42 (1984) whereas the highest showed 3.15 (1988). Helander et al. (2002) noted index values for the Swedish Baltic subpopulation of ~ 2.8 (1975-97) and for the Lapland subpopulation 2.96 (1985-97), one hundred years before comparably higher indices in Sweden had been determined with 3.2 (1865-1935) for the Baltic coast and 3.18 (1867-1909) for Lapland. White-tailed eagles in Schleswig-Holstein are producing eggs with a shell thickness which is obviously strong enough for breeding success as Helander et al. (2002) declared for Swedish eagle eggs, too. But the shell thickness from the 19th century is not accomplished today.

For eagles no correlations between shell-index and residues, years or age of females were obvious. This is different from the results we could find in goshawks (*Accipiter gentilis*) from Schleswig-Holstein (Scharenberg and Looft, 2004). Obviously, eagles are less sensitive to this parameter than goshawks.

Values for risk assessment (like LOEC) have been calculated by several authors. Those, developed for eagles, are of interest. Calculated values for depressed

productivity (mean number of young produced per year) for DDE and PCB in eggs were: 120 mg/kg (lw) resp. 500 mg/kg (Helander et al., 2002). The population which had been monitored lived on the Swedish Baltic Sea coast and should be very similar to the population of Schleswig-Holstein with regard to

Table 2. Development of the breeding population of sea eagles in Schleswig-Holstein.

period	*years*	*controlled breeding pairs*	*pairs with success**	*% successful pairs*
1969-76	**8**	38	10	26
1980-88	**9**	36	27	75
1992-01	**10**	167	125	75

*) success=pair with at least one fledged nestling.

physiology, behaviour an genetic aspects. Helander et al. (1982) have reported a critical LOEC for DDE and PCB in a range of 500 mg/kg lw and 800 mg/kg lw resp.. Olsson et al. (1998) have compared residues in undeveloped eggs with those in eggs containing embryos which died before fledging and suggest that 300 mg PCB/kg (lw) were responsible for effects (LOEC; dying embryos). Nevertheless individual females can reproduce with considerable higher egg concentrations. 100% reproduction failure for bald eagles (*Haliaeetus leucocephalus*) correlated with 15 mg DDE/kg (ww) (300 mg/kg lw) while shell thickness reduction occurs at concentrations of 100 mg/kg (lw) (Wiemeyer et al., 1984). Elliot et al. (1996) collected incubated eagle eggs, hatched them and sacrificed fledged birds. After determining several physiological parameters and PCB they suggested a NOEC measuring TEQ (toxicity equivalent factors) in eggs of 100 ng/kg (ww) corresponding to 2 mg/kg (lw) and a LOEC also measuring TEQ of 250 ng/kg (ww) corresponding to 5 mg/kg (lw). Converting TEQ into PCB residue values leads to a concentration of around 300 mg/kg (lw) (Olsson 1998), similar to the above mentioned cooncentrations.

Since 1980 we could not detect PCB concentrations above 300 mg/kg lw while in former years higher concentrations could be measured. The situation concerning LOEC for DDE is comparable: before 1980 several eggs contained residues higher than 120 mg/kg lw, while no egg after 1980 exceeded this concentration. Therefore we are not expecting any further reproduction stress caused by OC for eagles in Schleswig-Holstein. Only the eggshell index might still be influenced by OC. As Helander et al. (2002) suggested, the LOEC for productivity for DDE (around 100 mg/kg (lw)) can cause shell thinning without negative reproduction effects.

In summary, we can state that OC residues in eggs of white-tailed eagles from Schleswig-Holstein declined significantly during the last 40 years. While OC still persist residues are measurable but lower than risk values cited in literature. Finally, we find increasing reproduction success since 1980, the shell index today having not reached the value measurable before the introduction of OC.

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