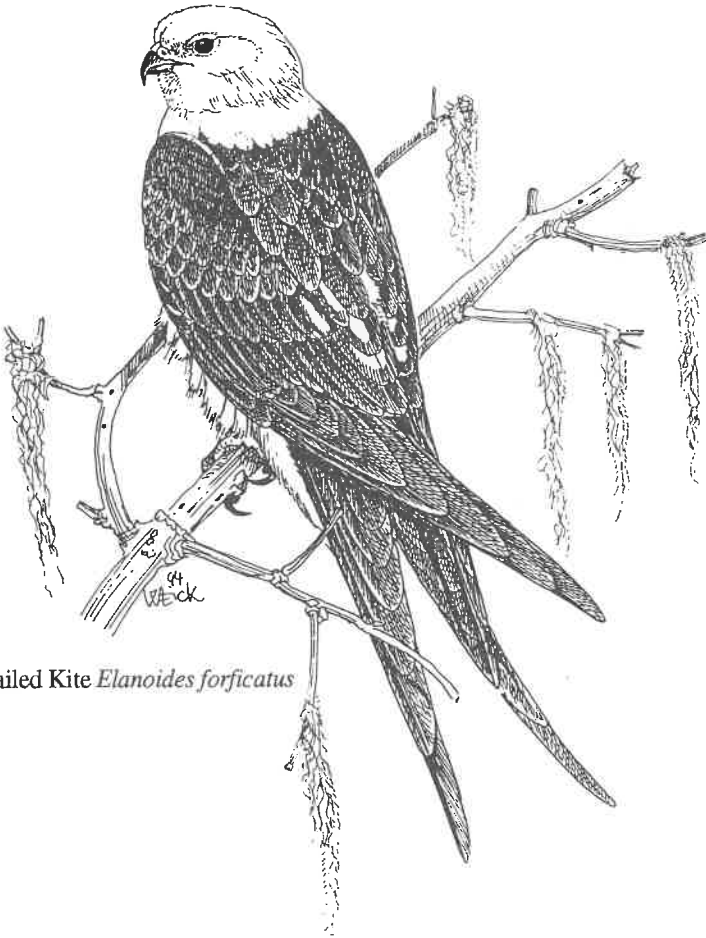


Part 10

Environmental Contaminants and Raptors



Swallow-tailed Kite *Elanoides forficatus*

Convener:

Dr. R. W. Risebrough

Toxic Chemicals and Birds of Prey: Discussions in Berlin in 1992

R.W. Risebrough

The structure of the Contaminants Session at the Ivth World Conference on Birds of Prey in Berlin, as well as of this overview paper, has followed with one major exception that of the IIIrd World Conference in Eilat and of its summary chapter (Risebrough 1989). The exception has derived from a need to incorporate a discussion of current research on the PCBs and of their current ecological relevance to birds of prey.

The principal contaminant issues of concern for the protection of birds of prey appeared in 1987 to fall within six principal considerations: 1) A re-interpretation of past events, in the light of recently acquired knowledge and understanding. This is of evident scientific interest, and responds to a continuing need to provide a technically sound basis for conservation policies that would protect birds of prey from adverse effects of the use of biocidal chemicals in all parts of the world; 2) Documentation of now-familiar effects of toxic chemicals on birds of prey, such as eggshell thinning and depressed productivity, in areas of the world that have hitherto received relatively little attention; 3) Priorities for future research and projects; 4) New chemicals or new uses of chemicals; 5) Re-introduction of raptors into polluted environments; 6) Implementation of national and international strategies for reducing harmful effects of toxic chemicals on birds of prey.

Although the papers presented in Berlin provided a very uneven coverage of these topics, the same format is followed in this overview chapter, except that the sixth topic is not discussed in the absence of any suggested new approaches. The review of current relevance of the PCBs is included with an assessment of their role in contributing to the population declines of the past. To reduce the demands on space, all references for the papers in the Contaminants Session are combined into a common bibliography.

One editorial inconsistency deserves mention. The name "White-tailed Eagle" is used for *Haliaeetus albicilla* in the papers by North American authors, but there appears to be no justifiable reason to apply the dictates of the Committee on Classification and Nomenclature of the American Ornithologists Union to the papers of European authors, in which the traditional name "White-tailed Sea Eagle" continues to be used.

1. RE-INTERPRETATION OF THE PAST

At the Eilat Conference in 1987 a major effort was made to sort out the relative contributions of DDE and of dieldrin in contaminant-induced population declines of several raptor species, particularly the Peregrine Falcon, *Falco peregrinus*. The need for this had become apparent at the Second World Conference in Thessaloniki in 1982: the British contingent was unanimous in believing that adult mortalities caused by dieldrin had been the major factor in the declines of Peregrine Falcons and Sparrowhawks, *Accipiter nisus*, in Britain; most of the North Americans

considered that depressed productivity associated with DDE-induced shell thinning was the major factor in the disappearance of the Peregrine from eastern North America and the population declines of other species such as the Osprey, *Pandion haliaetus*, and Bald Eagle, *Haliaeetus leucocephalus*. A mixed message indicating a fundamental uncertainty about the real cause(s) of the population declines was coming from the community with the greatest interest in protecting birds of prey. Academically, this situation was unacceptable, inconsistent with the need to establish an historical accuracy. Moreover, an uncontested argument from either side could be used to justify continuing use of these or comparable biocidal chemicals in several parts of the world.

In the discussions of dieldrin-DDE effects, nothing, or almost nothing, was said about the PCBs and their possible role in population declines. Over the past five years there have been major advances in the understanding of the physiological and biochemical mechanisms of action of PCBs. It is clearly time to assess the role of PCBs in the population declines of raptor populations in the past, and their current role in causing or contributing to the depressed productivity in the several raptor populations still experiencing such difficulties. This was the principal theme of the Contaminants Session, and is the principal focus of this overview chapter.

Dieldrin-DDE

My own conclusions from the discussions in Eilat, which followed those on the same theme in the proceedings of The Peregrine Fund Conference in Sacramento (Cade *et al.* 1988; Nisbet 1988; Risebrough & Peakall 1988) are that depressed productivity caused by DDE could have caused the disappearance of the Peregrine from eastern North America over a 20-year period and that adult mortalities caused by the DDT compounds without any contribution from other contaminants made extinction inevitable; adult mortalities caused by dieldrin and other organochlorine biocides undoubtedly had a major additional impact on the population, or would have had if Peregrines were still surviving. In Britain, the use of dieldrin came at a time when the Peregrine population had been stressed by DDE-induced low productivity over a decade. The number of non-breeding birds without territories is therefore expected to have been abnormally low. Normally, a bird without a territory would quickly replace a member of a pair that has died such that an increase in the adult mortality rate might not be immediately detected; any increase in mortality rate, even a relatively small one, would much more likely be detected if the floating population without territories was depressed in its numbers (Risebrough & Peakall 1988; Risebrough 1989). This factor could explain why the decline of breeding Peregrines in Britain coincided more closely with the use pattern of dieldrin rather than that of DDT.

Newton *et al.* (1992) have compiled data on the percentages of mortalities of Sparrowhawks and Kestrels, *Falco tinnunculus*, that were caused by dieldrin in Britain over the period 1963-90. Population changes were correlated with the changes in dieldrin-induced mortalities. Incorporation of these data, together with productivity data, into a population model might provide further insight into the importance of adult mortalities vs. depressed productivity as causes of the population declines in these and other species affected by organochlorines. Mentioned in the discussion of the papers at the Eilat symposium (Risebrough 1989) but not adequately developed was the importance in assessing the relative importance of dieldrin (principally adult mortalities) and DDE (principally depressed productivity) of applying population parameters on a species-by-species basis to population models such as those developed by Grier (1979, 1980, 1991). A model applied at this symposium to Bald Eagles in northern California (Jenkins, this volume) was based on population parameters of adult survival, juvenile mortality and productivity compiled over a ten-year period in a sub-population. Over the period 1966-1972, autopsies of 145 Bald Eagles found dead in the U.S. indicated that 12.4 % had died of dieldrin poisoning (Mulhern *et al.* 1970; Belisle *et al.* 1972; Cromartie *et al.* 1975). A comparable 11% increase of adult mortality, without any depression of productivity, would have only a small effect in the northern California population; yearly adult survival would be reduced from 94.6% to 94%.

The uncompleted task is to compile the best available current estimates of the relevant natural population parameters of all species that have been affected by contaminants, and from a model predict the effects of increases in adult and juvenile mortalities and depressions in productivities, taking into account the existence of a non-breeding component without territories, density-dependent factors, age-dependent mortality factors, etc. This is therefore proposed as the priority topic for the Contaminants Session at the Vth World Conference in 1997.

Moreover, further clarification of the dieldrin-DDE and the increased mortality-depressed productivity factors will be a pre-requisite in resolving a new controversy that is developing - on the relevance and importance of past and present impacts of PCBs.

The Avian Toxicology of PCBs

When PCBs were first recognized as global contaminants, similar to the DDT compounds in their environmental behavior and whose local abundance was frequently as high or higher than that of the DDTs, a role in the population declines of birds of prey appeared likely (Jensen *et al.* 1969; Koeman *et al.* 1969; Risebrough *et al.* 1968). David Peakall's experiments showed that PCBs, like DDT, DDE, and dieldrin, induced mixed-function oxidase enzymes in the liver that degraded the female sex hormone and other steroid hormones into polar metabolites (Peakall 1967; Risebrough *et al.* 1968; Lincer & Peakall 1970). Estrogen was known to be linked with the deposition of medullary bone, the principal source of eggshell calcium (Simkiss 1967), and a link with shell thinning appeared plausible. This hypothesis was soon disproved: PCBs were found not to be the major cause of shell thinning (Peakall 1971; Heath *et al.* 1972; McLane & Hughes 1980), and the absence of adverse effects except in the oviduct in populations highly contaminated with DDE such as the California Brown Pelicans, *Pelecanus occidentalis*, that were laying very thin-shelled eggs, pointed to a mechanism of action that was local and confined to the oviduct (Risebrough *et al.* 1970).

PCBs have a relatively low acute toxicity to most avian species (Stickel *et al.* 1984) and only in those cases when body burdens were particularly high would they be expected to have contributed to adult mortalities (e.g. Koeman *et al.* 1973). The domestic chicken is an exception and is particularly sensitive to PCB toxicity (e.g. Flick *et al.* 1965; Vos & Koeman 1970; Bush *et al.* 1974), just as the mink, *Mustela vison*, is among mammals (Aulerich & Ringer 1977).

A number of experimental studies have demonstrated PCB effects on behavior (e.g. Peakall & Peakall 1973; Kreitzer & Heinz 1974; Tori & Peterle 1983; Koval *et al.* 1987). Comparable effects have been shown to be caused by DDE (Haegele & Hudson 1973, 1977; Richie & Peterle 1979), dieldrin (Gesell *et al.* 1979; Kreitzer & Heinz 1974), toxaphene (Kreitzer 1980), endrin (Kreitzer & Heinz 1974; Kreitzer 1980), and chlordane (Kreitzer & Heinz 1974), suggesting a common mechanism of action. In the field a lower intensity of nest defense behavior by Merlins, *Falco columbarius*, was associated with higher concentrations in the eggs of DDE and PCBs and with shell thinning (Fyfe *et al.* 1976; Fox & Donald 1980). Nest defense and other behavior patterns of Herring Gulls, *Larus argentatus*, was altered in colonies with higher levels or organochlorine contamination (Fox *et al.* 1978). A mixture of DDE, PCBs, mirex and photomirex in the proportions encountered in the Lake Ontario ecosystem produced alterations in the behavior of Ringed Turtle-Doves, *Streptopelia risoria*, under experimental conditions (McArthur *et al.* 1983). Ratcliffe's earlier observations of apparent abnormal behavior of Peregrines (i.e. Ratcliffe 1970) are recalled. The bulk of the evidence indicates therefore that PCBs, and the other organochlorines, had a significant role in depressing productivity through modifications of behavior.

Although induction of the mixed-function oxidase enzymes results in degradation of steroid hormones, feed-back mechanisms maintained levels of plasma progesterone in Mourning Doves, *Zenaida macroura* (Koval *et al.* 1987), Ringed Turtle-Doves (McArthur *et al.* 1983) and Japanese Quail, *Coturnix* (Biessmann 1982). In each of these studies, however, significant deviations in the timing of hormone response resulted in effects that would cause a general decrease in reproductive efficiency with ecological significance. The corrective action worked, but did not provide the fine-tuning necessary to maintain reproductive efficiency.

Another mechanism whereby PCBs can affect behavior has its most dramatic example in a depression of learning ability by human children exposed to PCBs in the womb (Jacobson *et al.* 1985, 1990a, 1990b). The effect has been confirmed experimentally with monkeys (Schantz *et al.* 1991) and may be related to a reduction of brain levels of dopamine (Seegal *et al.* 1990, 1991a). In birds, depression of both dopamine and norepinephrine in the brains of Ringed Turtle-Doves followed feeding of 10 or 100 ppm Aroclor 1254, 20 or 200 ppm DDE, or 4 or 16 ppm dieldrin, but not by 1 ppm Aroclor 1254, 2 ppm DDE, or 1 ppm dieldrin (Heinz *et al.* 1980). The effect has been associated in monkeys with lightly-chlorinated congeners with a chlorine in the *ortho* position (Seegal *et al.* 1990, 1991b; Shain *et al.* 1991).

Very different kind of PCB effects provide the background for the papers by Henny *et al.* and

Falandysz *et al.* (this volume). Unlike the effects on behavior, these do not share a common mechanism of action with the organochlorine biocides; rather, biochemical effects are shared with the chlorinated dibenzofurans and dioxins. The history of this fascinating chapter in Pollution Ecology has recently been summarized by Gilbertson *et al.* (1991). Briefly, a disease in chickens that first appeared in 1957 and that was characterized by edemas (Schmittle *et al.* 1958) was traced to contamination of the feed fats by a chlorinated dioxin (Cantrell *et al.* 1969). Among the dioxins, the 2,3,7,8-tetrachlorodibenzo-*p*-dioxin (TCDD) had the highest capability to cause the disease (Higginbotham *et al.* 1968). PCBs produced comparable effects at higher doses; symptoms included abnormalities of both the bill and the feet (Flick *et al.* 1965; Kohanawa *et al.* 1969; Bush *et al.* 1974). The finding of comparable abnormalities in chicks of Common Terns, *Sterna hirundo*, and Roseate Terns, *S. dougallii*, in Long Island Sound (Hays & Riscbrough 1972) and in several species of water-birds in the Great Lakes (Gilbertson *et al.* 1976) prompted concern that dioxin-like compounds were present in the environment.

Initially, the dioxin-type effect of PCBs was attributed to the presence as contaminants in commercial PCB mixtures of chlorinated dibenzofurans, structurally similar to the dioxins but somewhat less toxic (Vos *et al.* 1970; Bowes *et al.* 1975). Experiments with Mallards, *Anas platyrhynchos*, fed PCBs with a known dibenzofuran composition indicated, however, that the dibenzofurans were accumulated to a much lower extent than the PCBs (Norstrom *et al.* 1976), indicating a low potential for ecological effects. A major achievement in PCB research over the past several years has been the identification of several PCB congeners, those that resemble the dioxins in their structure, as the cause of the majority of the dioxin-type effects observed in the environment, including among birds embryonic mortalities and deformities (Kubiak *et al.* 1989; Gilbertson *et al.* 1991; Tillitt *et al.* 1992) similar to those produced by PCBs in chickens (Cecil *et al.* 1974). These PCB congeners have been termed "coplanars" because the absence of chlorines in the *ortho* positions permits a planar configuration resembling that of the dioxins and benzofurans (see the papers by Henny *et al.* and Falandysz *et al.* in this volume).

In assessing the relative importance of these effects in relation to other contaminant-induced effects on raptors both in the present time and in the past, this session was particularly fortunate in receiving contributions from two laboratories that have made major contributions in the development of analytical techniques for the identification and measurement of the coplanars in environmental samples, the laboratories of Drs. Yamashita, Tanabe, and Tatsukawa in Japan, and that of the U.S. Fish and Wildlife Service in Columbia, Missouri.

Ecological Relevance of Dioxin-type Effects

Several investigators of the distribution of PCBs in Great Lakes ecosystems (e.g. Heinz *et al.* 1984) have remarked that if fish-eating birds were as sensitive to PCBs as are chickens, many Great Lakes populations would long ago have been decimated. One of the coplanars, PCB-77, had no effects on embryos of several duck species, Herring Gulls, and Black-headed Gulls, *Larus ridibundus*, at doses 1-3 orders of magnitude higher than the LD₅₀ to chick embryos; turkeys and pheasants are also less sensitive to the effects of coplanar PCBs than are chickens (Brunström 1988; Brunström & Reutergårdh 1986). This high magnitude of interspecific variation precludes any rigorous comparison of effects between raptors and other groups or among raptor species based on concentrations of the coplanars, or of their toxicities expressed as equivalencies (TEQs) to the toxicity of TCDD, the most toxic of the dioxins.

The recovery of raptor populations, described in many papers presented at the Sacramento conference in 1986 (Cade *et al.* 1988) has continued. Relatively few populations still experience depressed productivity in North America and Europe, and only in these can current effects of the coplanar PCBs be looked for.

Productivity of the Bald Eagles has generally improved (i.e. Jenkins *et al.* this volume) but the sub-population breeding in the Columbia River Estuary in north-western US continues to have poor reproductive success; levels of TCDD in five eggs were "elevated", with a mean concentration of 32 pg/g (Anthony *et al.* 1993). This value, expressed as a Toxic Equivalent, is less than 1% of the geometric mean of TEQs derived from coplanars in breast muscle of adult female White-tailed Eagles in north-west Poland (Falandysz *et al.* this volume). Other studies have reported that TCDD usually accounts for only a small fraction of the total TEQs (Kubiak *et al.* 1989; Jarman *et al.* in press; Henny *et al.* this volume) such that without data on the other active compounds it is

not possible to evaluate the potential for a dioxin-type effect in this population. The mean of estimated total PCBs (13 ppm) measured in the eggs was higher than the mean DDE concentration (10 ppm). Since depressed productivity was related to shell thinning in this population (Anthony *et al.* 1993), DDE alone would appear to be sufficient to account for the low productivity (Wiemeyer *et al.* 1984, 1993; Nisbet & Risebrough this volume).

Depressed productivity of Bald Eagles breeding on the shores of several of the Great Lakes is discussed in the papers by Best *et al.* and Bowerman *et al.* (this volume). A coplanar contribution is suggested from observations of chicks with beak abnormalities. The DDE alone, however, in the eggs from Lake Michigan and Huron (Bowerman *et al.* this volume) could account here also for the depressed productivity without a contribution from a coplanar effect or other PCB effects. Since concentrations of PCBs are highly correlated with those of DDE, it would be difficult to demonstrate conclusively, as in previous studies (Wiemeyer *et al.* 1984, 1993; Nisbet 1988) a separate contribution of the PCBs. Other local species such as the Double-crested Cormorant, *Phalacrocorax auritus*, almost disappeared during the years of high DDE contamination (Anderson & Hamerstrom 1967) but numbers have greatly increased with the decline in environmental levels of DDE (e.g. Sharf & Shugart 1981; Price & Weseloh 1986). PCB effects can now be distinguished from those of DDE (i.e. Tillitt *et al.* 1992). A further decline in DDE levels might therefore be necessary before a separate PCB effect on the local Bald Eagles can be distinguished in the Great Lakes area. Needed are analyses of the causes of failure (natural or non-contaminant causes, embryonic mortality not explained by non-contaminant events, infertility or very early embryonic mortality, etc.) and correlations with levels of total PCBs, the coplanars, and DDE.

In a sample of 85 unhatched eggs of White-tailed Eagles obtained in 1965-1978, infertile addled eggs contained significantly higher levels of both DDE and PCBs than fertile unsuccessful eggs (Helander *et al.* 1982). Unless the addled eggs had contained small decomposed embryos, such a pattern is unlike that observed in the fertility and hatchability of eggs of chickens on PCB diets. In these, fertility was not affected, but embryonic mortality occurred earlier in the more contaminated eggs (Tumasonis *et al.* 1973; Bush *et al.* 1974). The Sea Eagle pattern is also unlike that observed a colony of Double-crested Cormorants in which embryonic mortality was correlated with coplanar concentrations (Tillitt *et al.* 1992). Helander's observations are consistent with a combined DDE/PCB effect on fertility or behavior being a significant factor in depressing productivity; an association between higher total PCB levels and embryonic mortality would be expected if the coplanar PCBs were a major factor in causing depressed productivity in the Sea Eagles.

Coplanar contamination of the White-tailed Eagles from coastal areas of Poland is high, accounting for geometric mean TEQs from 3,500 to 11,600 pg/g in breast muscle, depending on the method of estimation (Falandysz *et al.* this volume). Helander (1983) found abnormal toes in a White-tailed Eagle chick that died after pipping and beak abnormalities in 2 of 115 nestlings examined live on the Baltic coast of Sweden, but none in 60 examined in Lapland. Analyses of fish, cod-liver oils, eggs of the Common Murre, *Uria aalge*, and eggs of the White-tailed Eagle have shown no change in PCB levels in the Baltic since the early 1970s but a significant decline in DDE (Olsson 1978; Schneider 1978; Luckas *et al.* 1980a,b; Koivusaari *et al.* 1980; Kannan *et al.* 1992; Helander this volume); simultaneously, productivity of the White-tailed Eagles has increased over the region (Oehme 1990; Oehme & Manowsky 1991; Mizera 1990; Mizera & Szymkiewicz 1991; Koivusaari *et al.* 1980; Helander this volume). These data and observations indicate that DDE, or a combined DDE-PCB effect, was primarily responsible for depressed productivity and that a PCB effect independent of DDE, such as that of the coplanars, has had only a minor role. The minimum estimate of a threshold level for dioxin-type effects at the population level in White-tailed Eagles would therefore be in the range of 4-12 ng/g, higher by 1-2 orders of magnitude than that determined by Tillitt *et al.* (1992) for Double-crested Cormorants.

The Peregrines of central and southern California have been increasing, in part or entirely as a result of an intensive program of manipulation and management (Walton and Thelander 1988). Productivity of many pairs, however, would be low or zero in the absence of a management program (Linthicum *et al.* this volume; Walton this volume). Jarman *et al.* (in press) determined a mean total TEQ of 120 pg/g from dioxins, benzofurans and coplanar PCBs in 7 eggs from 5 sites in 1983-1988 based on the toxic equivalency factors (TEFs) of Kutz *et al.* (1990) and Safe (1990). This is a level at which an increase in embryonic mortality in Double-crested Cormorant eggs can

be detected (Tillitt *et al.* 1992); the total TEQs were determined by bioassay rather than application of TEFs to measured concentrations of active compounds. Peregrines, however, can not be assumed to be as sensitive as chickens or cormorants, or as insensitive as ducks, to the dioxin-like effects. Henny *et al.* (this volume) report much higher levels of total TEQs in eggs of 6 Peregrine clutches from arctic Russia from which a geometric mean of 890 pg/g can be calculated. About half of the TEQs, however, derive from PCB-118, a congener that contains one chlorine in an *ortho* position, with significant contributions from other *ortho* PCBs. Jarman *et al.* included only the 4 coplanar congeners (#s 37, 77, 126 & 169) in their estimate of PCB contributions to a potential dioxin-like effect. Including congeners 118 and 105 in the analysis would increase the estimate of total TEQs in the California eggs by a factor of about 4 to around 500 TEQs; other AHH-active congeners, if present in the California peregrines in about the same relative amounts as in the arctic eggs, would increase the estimate to no more than about 600, a level still below that of Henny *et al.* for a population in arctic Russia that has been increasing. The 890 pg/g mean in the Russian eggs would therefore appear to be a minimum threshold level of effect of dioxin-like compounds for Peregrines. If the eggs with higher TEQ levels in the Russian sample had dead embryos, a coplanar effect might be indicated; an absence of development would suggest other kinds of PCB/DDE effects.

A TEF derived for one species may not, however, apply to another; those used by Henny *et al.*, from Safe (1990), would indicate that PCB-118 is 100x less toxic than PCB-126; Brunström (1989) has reported that the toxicity of PCB-118 is 8000 times less toxic than PCB-126 when injected into chick embryos.

No evidence can be found therefore for a dioxin-type effect at the present time in a raptor population that is ecologically relevant, in the sense that a threat to the population exists. Fish-eating birds in contaminated areas of the Great Lakes that are experiencing dioxin-type effects such as the Caspian Tern, *Hydroprogne caspia*, and Double-crested Cormorants have been increasing with the decline in DDE residues; the incidence of bill abnormalities in the highly-contaminated Cormorant colony of Green Bay, Lake Michigan, has been about 0.5 % (Environment Canada 1991). Although there is a high incidence of local embryonic mortality associated with the coplanars (Tillitt *et al.* 1992), and however deplorable this may be, the population is not affected.

Certainly, continuation of PCB use on a global scale at the rate of increase in production of the late 1960s would have created a disastrous situation for many species, particularly for our own. The efforts in the late 1960s and early 1970s to end the uses of PCBs and the organochlorine biocides were well worth the time. That the eagles and peregrines have come back is our response, after 23 years, to the Editor of NATURE who wrote in 1969 "The fact that many professional scientists have recently been contributing to this misguided assessment of the risks of modern life is reprehensible" (*Nature* 224:1242).

Synergistic Effects of DDE and PCBs?

Jeffrey Lincer's Ph.D. thesis (Lincer 1972) is frequently cited in the literature but is not generally available. We publish here critical data from his thesis that suggest a synergistic effect of PCBs and DDE (Lincer, this volume). The results, however, are not conclusive since the small sample size precludes assignment of statistical significance; incubator technology has since improved. None of the eggs of the group fed DDE and PCBs hatched (Table 2, Lincer this volume). In the only other comparable feeding study we are aware of, Mallards, *Anas platyrhynchos*, fed a combination of DDE and Aroclor 1254 produced fewer eggs and showed a higher incidence of egg-eating than ducks on control, DDE, or PCB diets (Risebrough & Anderson 1975).

Clearly, Lincer's experiment should be repeated, with several dose levels; also, other unanswered questions remain to be answered (see below).

A synergistic DDE-PCB effect would also explain why productive Bald Eagle pairs in California and Arizona frequently produce eggs with higher DDE levels than the 3-5 ppm threshold determined by Wiemeyer *et al.* (1984) and Nisbet (1989) (Jenkins *et al.* this volume; Nisbet & Risebrough this volume). PCB levels in the earlier data set were generally higher. Here also sample sizes as yet preclude assignment of statistical significance.

Water Loss from Thin-Shelled Eggs

One of the unresolved apparent inconsistencies mentioned in Eilat derived from reports in the literature that thin-shelled eggs lost water less rapidly than normal eggs, while observations at the

Santa Cruz Predatory Bird facility indicated that thin-shelled eggs lost water more rapidly. We publish here a report from Santa Cruz (Linthicum *et al.* this volume) that clearly demonstrates that most thinner-shelled eggs brought in from the wild lose water more rapidly than normal; in the absence of corrective action by incubation at higher humidities, the embryos die. Some of the thinner-shelled eggs, however, lose water at the normal rates (Figure 2, Linthicum *et al.*). The Santa Cruz eggs differ from those in the experiments of Peakall *et al.* (1973) and Cooke (1979) in that initially at least they contain living embryos but why the other thin-shelled eggs should have decreased functional pore areas is not explained. Is there a link with the observations (i.e. B.J. Walton, pers. comm.) that egg survival and hatchability is much improved after several days of natural incubation but otherwise might be low? Is the high rate of water loss in some thin-shelled eggs due to the presence of structural abnormalities of the kind recently observed in eggs of Bald Eagles (Bland 1990)?

Unresolved questions remain; these also might be answered if the experiments of Lincer (above) could be repeated and expanded in scope to address questions of water loss, embryo survival, and changes in eggshell structure.

The water loss data from Santa Cruz indicate the importance of water balance to hatchability and could explain why species such as the Prairie Falcon, *Falco mexicanus*, experience DDE-related reproductive failures at a low level of thinning (Risebrough 1989).

Effects of Metals?

Falandysz and Mizera (this volume) provide additional data that feathers are useful monitoring media for heavy metals in local environments. There is no established link, however, with effects on individual birds or on populations. Helander *et al.* (1982) have clearly shown that mercury was not a factor in the population decline of White-tailed Eagles in the Baltic. This is a species, however, that has evolved in an environment in which a principal food source contains relatively high mercury levels. At the Eilat Conference, Furness *et al.* (1989) demonstrated a link, whether causal or incidental, between mercury levels and depressed productivity of Golden Eagles, *Aquila chrysaetos*, breeding on the island of Rhum in western Scotland. This species evolved in an environment with low mercury levels in the food sources. Is it now more sensitive than White-tailed Eagles to dietary mercury?

2. FAMILIAR EFFECTS IN NEW PLACES

Unfortunately we have only one contribution from an area of the world where biocide effects on raptors had not yet been investigated, that of Basili *et al.* from Venezuela. The preliminary data do not suggest an organochlorine problem. Observations of unregulated unrestricted use of other biocides indicate, however, the potential for other kinds of ecological disruptions.

Needed are studies in Asia, including India and China where DDT and HCH (hexachlorocyclohexanes, including lindane) and probably other organochlorines continue in widespread use.

3. PRIORITIES FOR FUTURE RESEARCH AND PROJECTS

There remain many unanswered questions on the immediate reasons for the reproductive failures and of their relative importance, - egg breakage, abnormal gas exchange, embryonic mortalities caused by toxins, behavioral abnormalities, etc., and of their physiological and biochemical basis. Even the mechanism of eggshell thinning is not adequately explained (i.e. Bland 1990; Lowe & Stendell 1991). Although no longer as relevant as they once were to a conservation policy, these questions nevertheless retain their scientific interest and importance.

As indicated above, in those areas of the world where DDT and other organochlorine biocides are still being used, particularly India, China, South-east Asia and areas of Africa and Latin America, studies of the status of those species potentially at risk, combined with laboratory analyses of unhatched eggs and other available samples, remain a priority.

Of equal immediate priority and of much greater longer-term priority would appear to be studies that determine effects of biocide use on local biodiversity and the abundance of prey species at all levels of the food webs. Many papers in the late 1950s and 1960s, e.g. Hickey & Hunt (1960a,

1860b), documented massive avian mortalities following applications of DDT, but evidence for longer-term population changes is generally lacking, e.g. Henny (1972). A longer-term field study of White-headed Black Chats, *Thamnolaea arnotti*, has shown, however, population reductions in areas of recurring DDT use in NW Zimbabwe (Douthwaite 1992). A report from Spain indicates depressed productivity of Blue Tits, *Parus caeruleus*, resulting from a shortage of prey following sprayings with a synthetic pyrethroid (Pascual & Peris 1992). Applications of any biocide will inevitably reduce populations of some components of food webs at least temporarily; continued applications could reduce the prey base of local raptors.

Population declines of frogs and other amphibians are occurring on a massive scale (e.g. Wake 1991; Pechmann *et al.* 1991); there is an analogy with the raptor declines of the 1960s, except that there appear to be no or few common factors that would explain the majority of the reported local disappearances or population reductions of the amphibians. How many other species are disappearing with them, and in the context of this symposium, how would such changes affect raptors? More than ever before is there a need for long-term, in-depth, ecological investigations.

4. NEW CHEMICALS; NEW USES OF OLD CHEMICALS

In their discussion of the potential hazards of the organophosphate fenthion, Lacombe *et al.* (this volume) review also effects of other less persistent biocides on raptors and their prey base. Additional support is therefore provided for the recommendations for the long-term research programs proposed above. Similarly, the observations of Peeters (this volume) indicate the need for field monitoring of the effects of rodenticide applications.

5. RE-INTRODUCTION OF RAPTORS TO POLLUTED ENVIRONMENTS

At the Eilat meeting, Garcelon *et al.* (1989) discussed problems arising from the re-introduction of Bald Eagles to Santa Catalina Island in Southern California; eggs with weak shells broke in the nest or failed to hatch during artificial incubation (Jenkins *et al.* this volume). The principal cause of their disappearance about 30 years ago, environmental contamination by DDE, was still active.

Three of the last California Condors, *Gymnogyps californianus*, in the wild died after ingesting lethal levels of lead (Janssen *et al.* 1986). Lead poisoning from bullet fragments was therefore probably a significant factor in the population decline of the species over the past century. Lead bullets are still widely used in the areas where captive-reared condors are now being released, and pose a potential hazard once the reintroduced birds begin to range more widely. The introduction to the market of all-copper bullets with superior ballistic properties is a possible solution of this problem. But nothing is known about the toxicity of copper to condors; the hypothesis that copper poisoning from ingested bullet fragments contributed to the earlier deaths of condors can not be excluded. Waterfowl deaths from copper poisoning have been documented (Henderson & Winterfield 1975; Kobayashi *et al.* 1991). Feeding experiments with a surrogate species such as the Turkey Vulture, *Cathartes aura*, would likely yield information useful for a future conservation strategy.

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