ENDOCRINE DISRUPTORS AND ECOTOXICOLOGY

HALOGENATED ENDOCRINE DISRUPTORS IN WILDLIFE AND THE AQUATIC ENVIRONMENT: IS THERE (STILL) A PROBLEM?

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Introduction

In the second part of the 20th century the introduction of halogenated pesticides, flame-retardants and the formation of their by-products, collectively known as organohalogens (OCs), caused significant problems for many wildlife species, especially those from or depending on the aquatic ecosystem. Since the 1960's numerous reproductive and developmental effects have been described in wildlife species, which have been linked or associated with OC exposure. Many of the compounds strongly biomagnify in species at the top of the aquatic foodchain, such as seals and fish-eating birds. Consequently, these species were found to be most sensitive to effects from exposure to high contaminant levels. An increasing awareness of the human and environmental risks associated with the exposure to OCs has resulted in a strong decline in production of these compounds. Subsequently, a measurable decrease in biotic levels, including those in humans, has been observed during the last decades. However, the question arises to what extent this decrease in environmental contaminant levels has led to a reduction in adverse health effects in wildlife. In this presentation a number of case studies, exemplified by animal species from different parts of the world, will be used to evaluate if organohalogen compounds are still a significant threat to wildlife populations. In addition, the discussion considers the relative impact of these compounds on wildlife compared with other anthropogenic factors, such as habitat destruction and overpopulation.

Albatrosses on Midway Island as a case example for a marine and pristine environment

During the last decade the populations and reproductive success of the Laysan Albatross (*Diomedea immutabilis*) and Black-footed Albatross (*Diomedea nigriceps*) have been studied in detail on Midway Island, an isolated atoll in the Pacific. Although both albatross species spend all their life at sea in relatively pristine environments, detectable levels of PCBs, PCDDs, PCDFs, DDT and several persistent metabolites have been found in their tissues. The information about contaminant levels, life cycle, annual reproductive success, and mortality caused by fishery by-catch, allowed a calculation of the relative risk of these organochlorines to both species[1]. Based on field studies it was observed that hatching success was lower and the crushing of eggs greater in the Black-footed Albatross than in the Laysan Albatross. Adult Black-footed Albatross retain these contaminants in their bodies, and thus appear to be at a slight risk from eggshell thinning. Subtle embryonic effects that could increase the risk of terata and decrease the viability of eggs

can also be expected. It was calculated that the observed levels of organochlorines probably reduced the reproductive success of the Black-footed Albatross by 2 to 3 %. No adverse reproductive effects were expected to be caused by these compounds in the Laysan Albatross. Since the 1980's, population surveys in breeding colonies have indicated a decrease in the Black-footed Albatross population. In contrast, the Laysan Albatross population has slowly increased. The question arises to what extent OCs are responsible for the decrease of the Black-footed Albatross population compared to the effects on mortality caused by the fishery by-catch. The effect of the fishery by-catch on adult loss and widowing was estimated to result in a decrease in reproduction of 17.6 % for the Laysan Albatross and 27.2 % for the Black-footed Albatross, respectively. Based on differences in life-style and behavior of both species, it is expected that there are no significant effects of organohalogen compounds and by-catch on theLaysan Albatross population. In contrast, the population of the Black-footed Albatross is slowly decreasing. The impact of by-catch on this decrease is about one order of magnitude greater than that of the organohalogen compounds [1].

Common Terns and Great Cormorants in the estuaries of the Rhine and Meuse as a case example for Western Europe

The impact of OCs on the reproductive success of Great Cormorants (*Phalacrocorax carbo*) and Common Tern (*Sterna hirundo*) has been studied in the estuaries of the Rhine and Meuse during the last decades. Comparative studies with both fish-eating bird species in several colonies, and differing in their degree of contamination, showed significant differences in hatching and breeding success. In a Great Cormorant colony situated in the heavily contaminated sedimentation area of The Rhine and Meuse, concentrations of p,p'-DDE were associated with eggshell thinning. In the 90's concentrations of PCBs still caused a significant decrease in hatching and breeding success of this species [2]. Laboratory studies with hatchlings of Great Cormorant also indicated that organohalogens could induce cytochrome P4501A (CYP1A) activity, while respiratory activity of the hatchlings was significantly increased, possibly due to the chemical stress[3]. The Common tern is another fish-eating bird species that has been the subject of research on reproductive success and perinatal development. Comparative studies in eight colonies in The Netherlands and Belgium showed that induction of hepatic CYP1A levels differ four-fold, depending on the TEQ levels in the yolksac. Average TEQ concentrations in chicks that hatched after 23 days of incubation were twice that of those that hatched after 21 days[4]. Effects of PCB 126 and PCB 153 on Common tern chicks were also studied in a postnatal feeding experiment. Relatively uncontaminated eggs were collected and hatched in an incubator. Hatchlings were given diet with concentrations ranging from 0.01 to 1.2 ng (avian) TEQs/g fish ww in which the ratio of PCB 126 and 153 was 1:1000. The estimated lowest observed effect level (LOEL) for induction of CYP1A in the Common tern was approximately 25 ng TEQs/g liver lipid, which was equivalent to 0.6 ng TEQs/g ww fish. At these dietary concentrations a reduction in plasma total thyroxine (T4) and bursa weight were also observed. At present, concentrations of TEQs in fish in The Netherlands are about 0.1 ng TEQs/g ww. Based on these results it is concluded that no overt effects on growth and development in the Common tern are expected during the post-hatching period with current background level of exposure [5]

Fish-eating birds on the Great Lakes as a case example for North America

Numerous studies in areas such as the Great Lakes basin of North America have documented a causal relationship between exposure of fish-eating birds to OCs and decreased reproductive capacity. Exposures to elevated concentrations of OCs in the 1970s and early 1980s have led to decreased populations of the Herring gull (*Larus argentatus*), Double-crested cormorant (*Phalacrocorax auritus*), Common tern, Forster's tern (*Sterna forsteri*) and Bald eagle (*Haliaeetus leucocephalus*) [6]. The overt symptoms of reproductive impairment were low hatchabilities, deformities in embryos and hatchlings, eggshell thinning, and the so-called 'chick-edema disease', which represents a whole suite of adverse effects. The observed eggshell thinning could be ascribed to the presence of p,p'-DDE. Various other toxicities, such as the chick edema disease have been linked to exposures to TCDD and several other toxicologically related chemicals, including other PCDDs, PCDFs and certain PCB conegeners. Strong associations have been documented between exposure to dioxin-like chemicals, the occurrence of chick edema disease and reduced reproductive capacity in several fish-eating bird species in the Great Lakes[7] and in Great blue herons in British Columbia, Canada[8]. Since the ban of DDT in the late 1970s levels of p,p'-DDE have decreased to levels that no longer cause eggshell thinning. Levels of dioxin like OCs have also declined. Concomitant with the observed declines of the concentrations of first DDT and later dioxin- like compounds, populations of several avian species have been restored. Current threats to populations of wild avian species may still be of a chemical nature, as new (agro)chemicals continue to be introduced into the environment, and old OCs continue to be released. However, the ever increasing human population, its associated economic activities and encroachment on areas formerly reserved for wildlife, may presently pose a greater threat.

Seals as a case example for marine mammals

Casual linkages between OC exposure and reproductive impairment at the population level have been reported for several marine mammal species from ecosystems in Europe and North America[9]. OC exposure has been linked to reproductive effects, immune dysfunction and/or greater mortality resulting in population declines of i) Baltic Grey (*Halichoerus grypus*) and Ringed Seals (*Phoca hispida*) suffering from a disease syndrome in the 1980s, ii) Harbour Seals (*Phoca vitulina*) and Stripped Dolphins (*Stenella coeruleoalba*) as a consequence of the 1988 and 1990 morbillivirus epizootics in the North Sea and Mediterranean, respectively, iii) Bottlenose Dolphins (*Tursiops truncatus*) from the U.S. east coast suffering a mass mortality in 1987-89, and iv) Beluga Whales (*Delphinus leucas)* from the St. Lawrence river estuary. In other marine-related and fish-eating mammals, casual associations have been made between OC exposure and masculinization in Spitsbergen Polar Bears (*Ursus maritimus*), reproductive impairment in European Otter (*Lutra lutra*) and Great Lakes Mink (*Mustella vision*) and Otter, premature pupping in California Sea Lions, and reduced testosterone levels in Dall's Porpoise from the northwest Pacific Ocean. The best evidence for direct cause-effect relationships between OC exposure and lower reproductive success and population health resulted from field studies with Baltic gray and ringed seal, and Dutch semi-field studies with captive common and harbour seals feed flatfish from the polluted Wadden Sea. Reproductive success was firmly linked to higher body burdens of OCs such as PCBs, DDTs, PCDDs/Fs and related compounds. Exposure to undetected contaminants and other OCs were also implicated, specifically persistent methylsulfone ($MeSO_2$-) PCB and –DDE metabolites and phenolic compounds and metabolites [10].

Studies so far on marine mammals suggest that the influence of OCs on reproductive success and fertility is a global phenomenon depending on the exposure level, which can potentially affect

wild populations in many industrialized areas of the world. One should consider this generalization with caution. Over the last 2 decades, decreasing body burdens of OCs in Baltic seals have exemplified a general decline in the OC body burden in marine mammals, most notably, from more polluted marine ecosystems. The OC decreases in Baltic seals have coincided with an improvement in reproductive status. However, seals from the Baltic, and even the more remote arctic marine environments continue to possess substantial body burdens of OCs. This may suggest that the majority of seal populations and communities are still affected by chronic and insidious, rather than acute OC-induced impacts on reproduction and population health. However, biological, physiological and other environmental stresses such as ingested waste, dwindling food resources and habitat loss are additional and perhaps more significant threats to the health of marine mammal populations.

Conclusion

During the last decades a general decline in OC levels has been observed in wildlife and in the environment as a whole. At present, the situation for wildlife appears to be more favourable due to the decreased impact of the current levels of OCs. Nevertheless, effects on reproductive success and perinatal development can still be observed in locations with high OC levels in the sediment. If the present impact of OCs is evaluated in a global perspective, habitat destruction and overpopulation are more serious threats for the existence and populations of wildlife species.

References

1. Ludwig, J.P., Summer, C.L., Auman, H.J., Gauger, V., Bromley, D.,Giesy, J.P., Rolland, R. and T. Colborn. 1997. In: Albatross Biology and Conservation (Eds. I.G.Robinson and R. Gales) p.225.

2. Dirksen S., Boudewijn T.J., Slager L.K., Mes R.G., Van Schaick M.J., De Voogt P. 1995. *Environ Pollut* 88,119

3. Van den Berg M., Craane L.H.J., Sinnige T., Van Mourik S., Dirksen S., Boudewijn T., Van der Gaag M., Lutke-Schipholt I.J., Spenkelink B., Brouwer A. 1994. *Environ Toxicol Chem.* 13, 803.

4. Bosveld ATC, Gradener J, Murk AJ, Brouwer A, Van Kampen M, Evers EHG, Van den Berg M. 1995. *Environ Toxicol Chem* 14,99.

5. Bosveld A. T.C., Nieboer R., de Bont A., Mennen J., Murk A J., Feyk L. A., Giesy J. P. and Van den Berg, M. 2000. Env. Tox. Chem. 19, 719

6. Gilbertson, M., Kubiak, T. and Ludwig, J. 1991. J.Toxicol. Environ. Health 33, 455 (and references therein).

7. Giesy, J.P., Ludwig, J.P. and Tillitt, D.E., (1994). Environ. Sci. Technol 28, 128A (and references therein).

8. Sanderson, J.T. and Van den Berg, M. 1999. Human Ecol. Risk Assess. 5, 43 (and references therein).

9. Vos, J.G., Dybing, E., Greim, H.A., Ladefoged, O., Lambré, C., Tarazona, J.V., Brandt, I. And Vethaak, A.D., 2000. *Crit. Rev. Toxicol.* 30,71(and references therein).

10. Letcher, R.J., Klasson-Wehler, E. and Bergman, Å, 1999. In *The Handbook of Environment Chemistry*; vol. 3, Part K: *New Types of Persistent Halogenated Compounds*, Ed. J. Passivita, Springer-Verlag , Heidelberg, pp.315-360 (and references therein).