SURVEY OF DANISH FREE LIVING OTTERS *LUTRA LUTRA*. A CONSECUTIVE COLLECTION AND NECROPSY OF DEAD BODIES.

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ABSTRACT

During 1979-1993, 194 dead Danish otters Lutra lutra were received. Of these, 145 were necropsied and the cause of death, sex, age and body condition determined. Traffic mortality (45.4%) and drowning (32.5%) constituted the major cause of death. Shot gun lead pellets were detected in 5% of the otters. Inclusion bodies indicating distemper virus infection were found for the first time in a free living otter population. Angistrongylus vasorum larvae were found in the lungs of free living otters for the first time. No ectoparasites were found. Infectious agents were detected in 22.1% of the otters although only few individuals appeared to have died from infections. The age distribution was not significantly different between sexes. Body condition for otters which died violently in Denmark was comparable to findings in Shetland, where thriving populations exist. The results showed a considerable decrease in number of otters found drowned in fish traps coinciding with the introduction of stop grids in fish traps in 1986. The results suggest that the existing otter population in Denmark is healthy and in good condition but it cannot be excluded that the large number of otters killed by traffic threatens the continued expansion of the species.

1 INTRODUCTION

The Eurasian otter *Lutra lutra*, is a highly vulnerable mammal in Denmark as well as in much of Europe (MACDONALD and MASON, 1994). In 1996 a national survey (HAMMERSHØJ *et al.,* 1996) concluded that the species occured in the northern part of Jutland; in the counties of Nordjylland, Viborg, Ringkøbing, Århus, Ribe and Vejle. On Zealand, in the county of Vestsjælland, no signs were found in the national otter survey, but in a more detailed survey undertaken parallel to the national survey (LETH and BYRNAK, 1996), signs of otters were found at two sites (Figure 1).

It has been claimed that contaminants such as the organochlorine pesticide dieldrin, polychlorinated biphenyls (PCBs), and heavy metals, in particular mercury, have been responsible for the rapid decline in otter populations in Europe (MACDONALD and MASON, 1994). Decreasing otter population in Denmark was thought to be due mainly to river regulation, wetland destruction, drowning in fish traps, and intensified traffic (MADSEN, 1991).

Otter carcasses have been collected annually in several European countries. In Germany e.g. more than 50 otters were found dead each year, but only a small number were necropsied (ZOGALL and REUTHER, 1992). Likewise only 24 of 113 dead otters collected in Shetland were necropsied (KRUUK and CONROY, 1991). In south-west England, 77 wild otters were examined postmortem (SIMPSON, 1997).

In this paper a comprehensive necropsy results of 145 carcasses submitted from a population of free living otters are evaluated to assess current threats to otters.

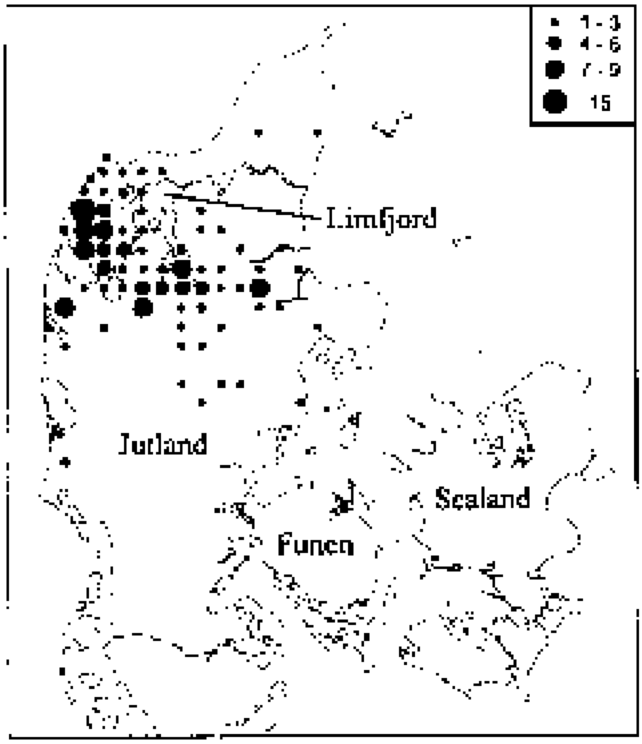

Figure 1. Geographical distribution of dead otters (N=193).

Figure 1. Geographical distribution of dead otters (N=193). the origin of one otter is unknown.

2 MATERIALS AND METHODS

Dead otters were received from hunters, motorists, anglers, forestmen etc. The otters were usually followed by written information about circumstantial evidence like killed on a road, died in a fish trap etc. Carcasses were frozen immediately upon arrival and stored at -18^0 C until necropsy was performed.

2.1 Necropsy

After thawing, the length (nose to tail) and weight were recorded. The animals were pelted followed by a routine necropsy procedure, including a search of the subcutis for lead pellets. Otters were aged as juveniles (less than about 5 months old) if tooth replacement was incomplete, as subadults (5-18 months) if the epiphyseal closure of humerus and femur at their proximal and distal ends was not complete or as adults (older than about 18 months). In males the length of the os penis was also used in ageing (VAN BREE, JENSEN and KLEIJN, 1966). The craniums were cleaned from muscles etc. and the upper and lower jaw was inspected by a dentist.

2.2 Laboratory tests

Lungs and gut contents were examined for parasites, eggs and larvae from parasites using McMaster and modified Baerman techniques (HENRIKSEN, 1965; HENRIKSEN and KORSHOLM, 1984). Scrapings of epithelial lining from trachea, lungs, and urinary bladder from otters necropsied later than 1988 were examined for viral inclusion bodies using S3-staining and a routine immunohistochemical method to detect distemper virus. Bacteriological examinations (Aerobic cultures on blood agar), were performed on material from the digestive tract, lungs and kidneys.

The body condition (K) of otters was calculated using the equation $K = W/(a \times L^n)$ where W = weight (kg) and L = total length (m) according to LE CREN (1951). The constants were those calculated by KRUUK, CONROY and MOORHOUSE, (1987) viz. a = 5.02 for females and 5.87 for males; n = 2.33 for females and 2.39 for males.

3 RESULTS

Of the 194 otters received, 145 were necropsied and 52 X-rayed. For some of the animals complete data were not received. Therefore, the number of individuals in the various examinations is inconsistent (Table 1).

Table 1. Salient data and the number of animals included.

Type of data presented	Number of animals
Total received	194
Origin stated	193
Sex determined	192
Age determined	178
Length and weight determined	158
Necropsied	145
X - rayed	52

The geographical origins and densities of the otters are given in Figure 1. The vast majority came from the Limfjord area. One individual found in 1979 came from the island of Funen. Half of the otters were found in or close to marine habitats. The annual number of carcasses received varied from two in 1979 to 31 in 1993 (Figure 2). Major causes of death were identified as traffic mortality (88 = 45.4 %) and drowning (63 = 32.5 %).

No significant difference was found in age distribution between the two sexes, ($\chi^2 = 0.43$; d.f. = 2:n.s.) (Table 2). Considerably more males (113) than females (79) were received during the survey. The weight and length of adult males were significantly higher than for adult females (weight - t = 9.60; d.f. = 65: $p < 0.001$, length - t = 20.35; d.f. = 67: $p < 0.001$). The condition index (K) of the otters had an overall mean value of 1.12, animals that died violently (traffic accidents and fish traps) had a value of 1.16 (Table 3).

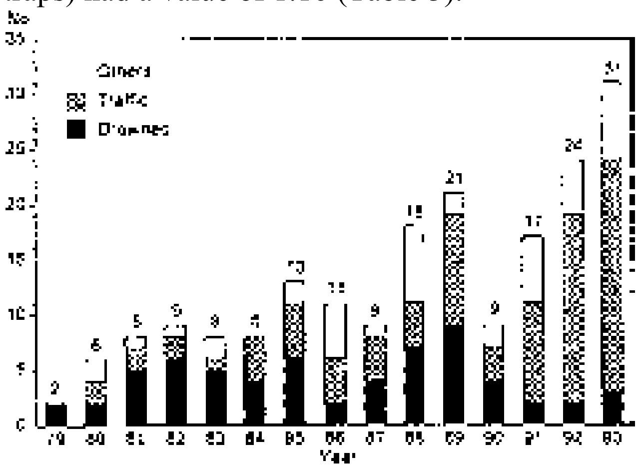

Figure 2. Annual number of dead otters and cause of death (N=194).

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Table 2. Sex and age distribution of dead otters.

	Females	Males	Unknown	Total
Juvenile	8	12	-	20 (10.3%)
Subadult	30	48	-	78 (40.2%)
Adult	34	44	2	80 (41.2%)
Unknown	7	9	-	16 (8.3%)
Total	79	113 (58.2%)	2 (1.0%)	194 (100%)

Table 3. Weight and length of adult otters and calculated condition indices (K).

	X	s.d.	range	n
Weight (kg)				
Male	9.07	1.35	5.45-11.40	37
Female	6.02	1.17	3.36-7.60	30
Length (cm)				
Male	112.9	5.06	90.0-130.0	36
Female	103.0	3.17	95.5-110.0	33
Condition index (K)				
Non-violent	0.94	0.18		
Violent	1.16	0.16		30
Sum	1.12	0.18		154

The results of necropsy and the corresponding pathological findings are detailed in Table 4. No ectoparasites were found. Signs of endoparasites were found in only five individuals viz. two with one egg of *Ascaridae* per gram in the intestinal tract, one with one egg of *Strongylidae* per gram in the intestinal tract and one with *Angiostrongylus vasorum* larvae in the lungs. Two tapeworm *Cestedae* eggs per gram were found in the intestinal tract of one individual.

Table 4. Numbers and types of pathological findings recorded at necropsy of dead otters (N=145).

Pathological findings	Number of animals
Parodontal disease	11 (7.6%)
Endoparasites	5(3.4%)
- Ascaridae	2
- Strongylidae	1
- Angiostrongylus vasorum	1
- Cestedae	1
Viral infections	6 (4.1%)
- distemper virus	6
Bacterial diseases	7(4.8%)
- pneumonia	5
- peritonitis	1
- Streptococcus sp.	1
Kidneystone	3(2.1%)
Gallstone/enlarged gall bladder	2(1.4%)
Hepatitis	2(1.4%)
Hypertrophied suprarenal gland	2(1.4%)
Tumour in spleen/enlarged spleen	2(1.4%)
Tumour in the small intestine	1(0.7%)
Umbilical hernia	1(0.7%)
Blindness	1(0.7%)
Total	43(29.7%)

Inclusion bodies were found in six individuals, three females and three males of

different age. These otters were all collected in the Limfjord area. The six otters were not believed to have suffered from clinical distemper.

Due to often severe decomposition bacteriological examination could only be applied to eight otters. Pneumonia due to bacterial infection was found in five individuals, four females and one male, of which two were juveniles. One abandoned juvenile died from bacterial peritonitis two weeks after being taken into captivity. Local infection with *Streptococcus sp.* was recorded in one animal.

Kidney stones consisting of ammonium urate were found in three adults, two males and one female, and two otters had a gall bladder enlarged by gall stones. Two otters showed hypertrophy of the suprarenal glands. A small intestinal tumour possibly a leiomyoma (severe decomposition) and a minor umbilical hernia was seen in two otters, respectively. The eyes of one adult, male otter were completely opaque, probably causing total blindness.

Lead pellets were found in nine otters (5%) in numbers from one to five pellets except for one individual carrying 14 pellets. The lead pellets were generally found in the pelt or subcutaneously and none were found in or close to vital organs. Parodontal disease was detected in 11 otters indicating a relatively high proportion of diseased animals.

4 DISCUSSION

Based on condition (K) of violent death otters there was no significant difference between otters from Denmark (Table 3) and from Shetland (KRUUK and CONROY, 1991) ($K = 1.08 \mp 0.15$ s.d. $n = 49$), ($t = 2.99$; d.f. = 171: n.s.) where thriving populations exist. The results agree with condition indices estimated by the authors from Danish data collected by JENSEN (1964) ($K = 1.13 \mp 0.16$ s.d. $n = 81$).

The increase in the annual numbers of submitted otters during the survey period (Figure 2) might indicate an expanding population of otters (MADSEN, CHRISTENSEN and JACOBSEN, 1992) but a greater public awareness of otters cannot be excluded as the underlying cause of the increasing number of submissions.

The present results show that males achieve a larger overall size than females. MASON and MACDONALD (1986) classified animals weighing more than 4kg as adults. In our study adults were classified as individuals with fully developed growth. One female with pneumonia but no emaciation weighed as little as 3.36kg confirming that the weigth and length alone may not be used as an indicator of age.

No ectoparasites and only small numbers of endoparasites were found. This indicates that in the present situation the otter is not parasitized very often, probably due to their solitary living and the relative scarcity of the species. However, decaying before collecting the dead otters combined with freezing might have disintegrated some parasites and larvae.

Except for the larvae of *Angiostrongylus vasorum* all other endoparasites recorded have been described earlier to occur in otters (JEFFERIES, HANSON and HARRIS, 1990; SCHIERHORN *et al.,* 1991; WEBER, 1991). Otters forage on frogs which might act not only as paratenic but also as intermediate hosts for *A. vasorum* (BOLT *et al.,* 1993, 1995). None of the parasites recorded were considered to have influenced the health status of Danish otters.

Distemper virus in captive Eurasian otters was described by GEISEL (1979) and STEINHAGEN and NEBEL (1985). Our study is the first to record distemper virus in a free living population of otters. The fact that the infected otters were collected from the Limfjord area in a period when distemper virus was present both in the common seal *Phoca vitulina* (BLIXENKRONE-MØLLER *et al.*, 1989) and in major outbreaks of distemper in farmed mink in this area indicates a wide range of host species for dis-temper virus. Negative findings in the remaining material may indicate a low propagatory rate of the virus in the population, but may also relate to the solitary life of otters and hence a low contact between animals.

Two cases of hepatitis probably causing severe health problems were seen. Pneumonic changes were found in five of 145 necropsied free living Danish otters. This corresponds to the findings of KRUUK and CONROY (1991) who found one case among 24 necropsied otters. Pneumonia has not hitherto been recorded in captive animals (ROGOSCHIK and BRANDES, 1991). One individual was recorded as blind in our study. WILLIAMS (1989) also reported blind otters from Britain during the period 1957-80.

Based on our study we would argue that only the two animals with hepatitis, and the five animals with pneumonia were likely to have died because of the diseases detected. In addition, one animal with peritonitis definitely died from this disease.

Since 1967, the Danish otters have been protected by law. During the period 1967-1982, fish farmers could be granted a special permission to kill otters at fish ponds but this exemption was terminated in 1982. However, this study shows that totally protected animals are still shot. To the less experienced hunter an otter may be mistaken for a free living mink of which more than 8,000 are shot annually in Denmark (ASFERG, 1999).

The level of PCBs in otters from Denmark (MASON and MADSEN, 1993) is at the same as found in 1988 among young common seals in the Limfjord area (STORR-HANSEN and SPLIID, 1993) and much lower than the 50mg/kg which causes reproductive failure among mink in laboratory studies and which is assumed by some to be a critical level for otters (KEYMER *et al.,* 1988; SMIT *et al.,* 1994).

It is seen (Figure 2) that the number of otters dying in fish traps has decreased. It is believed that this is the successful effect of a 1986 compulsory use of stop grids in fish traps for fishermen (MADSEN and SØGAARD, 1994). It should be noted that traffic mortality constitutes 45% of the total mortality (males as well as females, young as well as adults) indicating the need for preventive measures where roads are crossing rivers in Denmark.

In conclusion, our results suggest that the population of otters seems healthy and in good reproductive condition (ELMEROS and MADSEN, 1999), although traffic mortality may constitute a threat to the spread of the population.

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OCULAR PATHOLOGY IN WILD OTTERS

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1 INTRODUCTION

Simpson and colleagues have previously reported their investigation of disease in wild otters presented predominantly as carcasses after road traffic accidents (SIMPSON, 1997). A small number of captive otters were also investigated after death by this group. An important part of that study was that tissues were stored both fixed for further histopathological study and frozen for measurement of levels of vitamin A and of pollutants such as the polychlorobiphenyl toxins. Among tissues collected were eyes, allowing study of pathological lesions and correlation with vitamin A and pollutant levels. The findings reported here concern eyes from the first 82 otters examined.

2 GROSS PATHOLOGICAL FINDINGS

Few eyes showed abnormalities on gross examination of the entire or hemitransected globe and measurements of globe diameter showed no correlation with intraocular pathology or with vitamin A and pollutant levels.

3 HISTOPATHOLOGICAL FINDINGS

The particularly striking finding of this study was that of retinal folding and rosetting characteristic of retinal dysplasia. This was seen in around 40% of the eyes. This finding of abnormalities in retinal development such as retinal folding and rosetting is complicated by concurrent artefactual changes. Retinal detachment occurs when eyes are fixed, this being particularly marked with the use of formalin as fixative. It is sometimes difficult to differentiate the lesions of true retinal dysplasia from those of artefactual detachment but the finding of rosettes is indicative of abnormal retinal development. Thus although around 15% of the eyes had detachments which could not be differentiated from artefactual change, a significant number had retinal folds or rosettes which could not have occurred artefactually. A number of eyes had changes that could have been artefactual but were most probably true retinal dysplasia.

Eyes in which rosettes, involving one or more retinal layers, were obvious, were deemed to be clearly dysplastic (Figure 1). Eyes with retinal folds alone were considered possibly dysplastic, especially where there was adhesion between arms of the fold. Eyes with gross retinal detachment, which could have occurred as part of the pathological spectrum of dysplasia but was more likely to be fixation artefact, were considered to be artefactually damaged and not dysplastic.

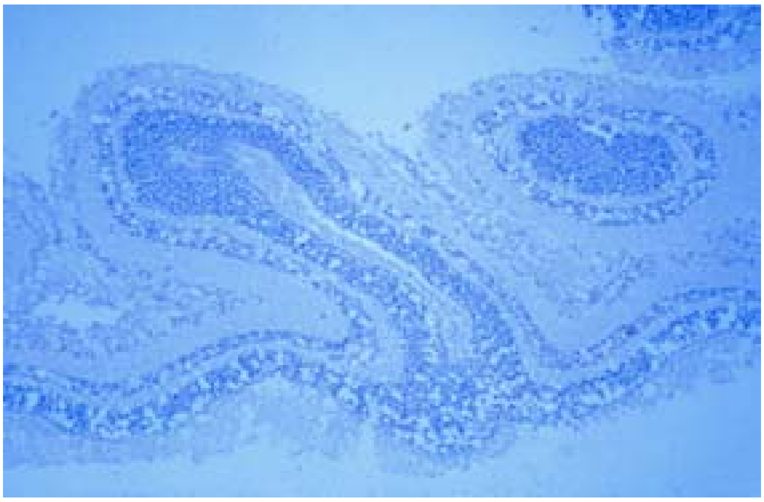

Figure 1

Given this categorisation, 32% of eyes had apparently artefactual posterior segment change. Twenty-five per cent were normal without significant retinal abnormality. Thirteen per cent had unmistakable dysplastic changes not complicated by any artefactual changes. A further 25% of eyes displayed potentially dysplastic lesions complicated by artefactual changes. Eleven per cent had other non-dysplastic pathology of the ocular surface, anterior or posterior segments. These included lesions such as lymphoid aggregates in the cornea and periocular multinucleate giant cell granulomas with protozoal cysts in periocular fat. A small number of eyes had degenerative changes in the lens and vitreal liquefaction while isolated cases had neurectodermal proliferation and choroidal thickening.

4 CORRELATION WITH VITAMIN A AND POLLUTANT LEVELS

Attempts to correlate retinal pathology with levels of various pollutants yielded significant results only in the case of dieldrin in which otters with dysplastic retinas had over three times the concentration of tissue dieldrin than in otters with normal eyes, this significant at $p = 0.028$. Comparison of liver vitamin A levels in animals with and without retinal changes also demonstrated statistically significant differences. Animals with retinal dysplasia had a statistically lower level of Vitamin A than those with normal retinas at $p = 0.023$.

5 DISCUSSION

The documentation of retinal dysplasia is the major finding in this study and one that may have significant implications as a sign of developmental abnormality in these otters. The discussion following will thus focus on this condition. Other abnormalities such as focal accumulations of leucocytes in the corneal stroma in some individuals and the finding of protozoal cysts in the extraocular muscles of one animal were seen in specific individuals and not as a finding over a number of otters in the group. While these are interesting findings, discussion of these individual changes must take a secondary place in this preliminary report.

Retinal dysplasia literally means maldevelopment of the retina but has been used specifically to denote changes involving retinal rosettes, folds and gliosis since first described at the end of the last century (BERGHEIMER, 1894). In human ophthalmology the term has been used more specifically for babies with these retinal changes associated with anomalies of the central nervous system but also to cover retinal maldevelopment with numerous aetiologies (SILVERSTEIN, OSBURN and PRENDERGAST, 1971; LAHAV, ALBERT and WYLAND, 1973). In the veterinary sphere retinal dysplasia has been used to describe similar retinal changes inherited in a number of dog breeds, sometimes as multifocal vermiform fundus lesions (ASHTON, BARNETT and SACHS, 1968), sometimes as larger geographic areas of retinal detachment (BEDFORD, 1982; O'TOOLE *et al.*, 1983) and sometimes as complete retinal detachment (RUBIN, 1968; CARRIG *et al.*, 1977). Viral infections such as herpes virus in the dog (PERCY, 1971) and bluetongue virus in sheep (SILVERSTEIN *et al.*, 1971) can both give rise to dysplastic retinal lesions. These are probably caused by aberrant development after retinal inflammation. In both animal models and human babies, a number of other factors have given rise to retinal dysplasia including radiation (GORTHY, 1979), cytotoxic drugs (SHIMADA *et al.*, 1973; PERCY and DANYLCHUK, 1977) and other drugs such as LSD (CHAN, FISHMAN and EGHBERT, 1978). The ocular teratogenic influence of vitamin A deficiency has been reported to cause signs of retinal dysplasia (PALLUDAN, 1961; VAN DER LUGT and PROZESKY, 1989). Any microphthalmic eye with multiple congenital anomalies may be characterised by dysplastic areas of retina, as one of us has previously reported (WILLIAMS and BARNETT, 1993).

Thus the finding of dysplastic retinal lesions in these otters could be inherited, nutritionally or environmentally induced, or toxic. The first of these must be considered unlikely in out-bred wild animals, although given the substantial reduction in the size of the wild otter population in the UK over the last decades this cannot be excluded from consideration. The correlation of retinal pathology with hepatic levels of vitamin A suggests that this nutritional cause is important although, as will be discussed below, it is abnormalities in maternal vitamin A that is likely to be the important contributing factor in the development of retinal pathology.

The mechanism of retinal dysplastic change thus remains unclear. The hypothesis that retinal necrosis is the primary cause of dysplastic change has been supported by several models (SHIVELEY *et al.*, 1970) although not by others (O'TOOLE *et al.*, 1983) and not by the findings in the present study. O'Toole suggested that focal areas of dysplasia may be caused by developmental abnormalities in retinal or vitreal vasculature or anomalies of Muller cells in the locality of retinal change.

The possibility of retinal folding as an artefactual change caused by fixation must be considered in a study such as this one. SZCZECH, PURMALIS and CARLSON (1976) for example, reported artefactual retinal folding in rat foetuses produced by 70% alcohol fixation. Alcohol produces considerable tissue shrinkage and as there is only firm adhesion between retina and underlying choroids and sclera at the optic disc and ora serrata, retinal detachment and formation of large folds may occur with alcohol fixation in such cases. Globe fixation in 4% buffered formaldehyde can be seen to cause retinal detachment in adult eyes through osmotic effects (MARGO and LEE, 1995). While the 70% alcohol immersion, used in this study for post-fixation scleral hardening, is unlikely to have caused retinal artefacts, the use of formalin as an initial fixative may well have caused a proportion of the total retinal detachments observed here. The retinal rosettes detected cannot, however, be explained as a fixation artefact and are thus taken as clear evidence of retinal dysplasia.

The most likely cause of the retinal dysplasia reported here, in these authors' opinion, is either hypovitaminosis A, given that otters with retinal lesions had a significantly lower level of vitamin A than those with normal retinas. While no obvious correlation could be made between levels of any one environmental pollutant and retinal pathology, there is a correlation between toxins such as dieldrin and vitamin A levels and thus these pollutants are likely to have had an indirect effect resulting in retinal pathology. One problem with the data as presented here is that the pathological effect of hypovitaminosis A on the retina occurs during the period of retinal development, that is to say before birth or as a cub suckling from the mother. Thus hepatic levels of vitamin A may not have much relevance to developmental defects such as the retinal dysplasia noted here. The statistical significance reported here does suggest that there is a meaningful relationship between vitamin levels and retinal pathology but does not, of course, prove a causative link between the two. Considerable further work is required to correlate disease severity with tissue levels of vitamin A and potentially teratogenic pollutants that may cause abnormal retinal development.

6 CONCLUSION

The finding of retinal dysplasia in these otters is surprising and probably points to an environmental toxin acting either directly by teratogenis or more likely through the agency of causing hypovitaminosis A. Considerable further work is required to substantial this and to rule out other possible explanations. These findings may signal that examination of retinal pathology in other wild species may show similar developmental abnormalities: the retinal may indeed be an exquisitely sensitive tissue to demonstrate toxic effects of pollutants.

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POLLUTION AND ITS EFFECTS ON OTTER POPULATIONS IN SOUTH-WESTERN EUROPE

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1 INTRODUCTION

The case of the otter (*Lutra lutra*) has been one of the clearest example~~s~~ of effort to research and conserve an animal species in the 20th century. This resulted from the dramatic decline in numbers and restriction of range that occurred during the second half of the 20th century (Figure 1). During the 1970s and 1980s the disappearance of the species was confirmed in most of Western Europe (REUTHER, 1980; FOSTER-TURLEY, MACDONALD and MASON, 1990; MACDONALD and MASON, 1994).

The causes of this decline in most of these territories were: (1) direct persecution (MASON and MACDONALD, 1986), although the otter also disappeared in places where there was little persecution, (2) the transformation of its habitats (resting sites, breeding and sheltering places), and (3) also changes in the availability of the species upon which the otter fed (see summary in MASON and MACDONALD, 1986 and FOSTER-TURLEY, MACDONALD and MASON, 1990). However, this did not explain why this mustelid had also disappeared from catchments where the habitats had apparently not been modified and, in most cases, it neither included prey populations that had been affected such as those of our study areas. These include, for example, regions of both sides of a large area of the Pyrenean mountains, the Alps, the Sierra Nevada and coastal wetlands such as the Ebro and Po Deltas, the Camargue or Albufera of Valencia.

The otter was not the only animal to be drastically affected in those years. In Western Europe, some other predators also showed sudden declines and fell to their lowest distribution levels. The first indications of decline were found in species like the peregrine falcon (*Falco peregrinus*) (NEWTON, 1979). The most affected animals were, however, insectivorous and fish-eating species such as the osprey (*Pandion haliaetus*), and various species of *Ardeidae* (*Ardea purpurea,* for example). The otter's decline, as well as that of these other species, was considered to be the result of a group of cumulative or synergistically-acting causes.

Pollution and epizootic occurrences were suggested as possible causes of decline (OLSSON and SANDEGREN, 1983; MASON, 1989), as they were the latest and the most consistent with the simultaneous disappearance of otter populations and other species.

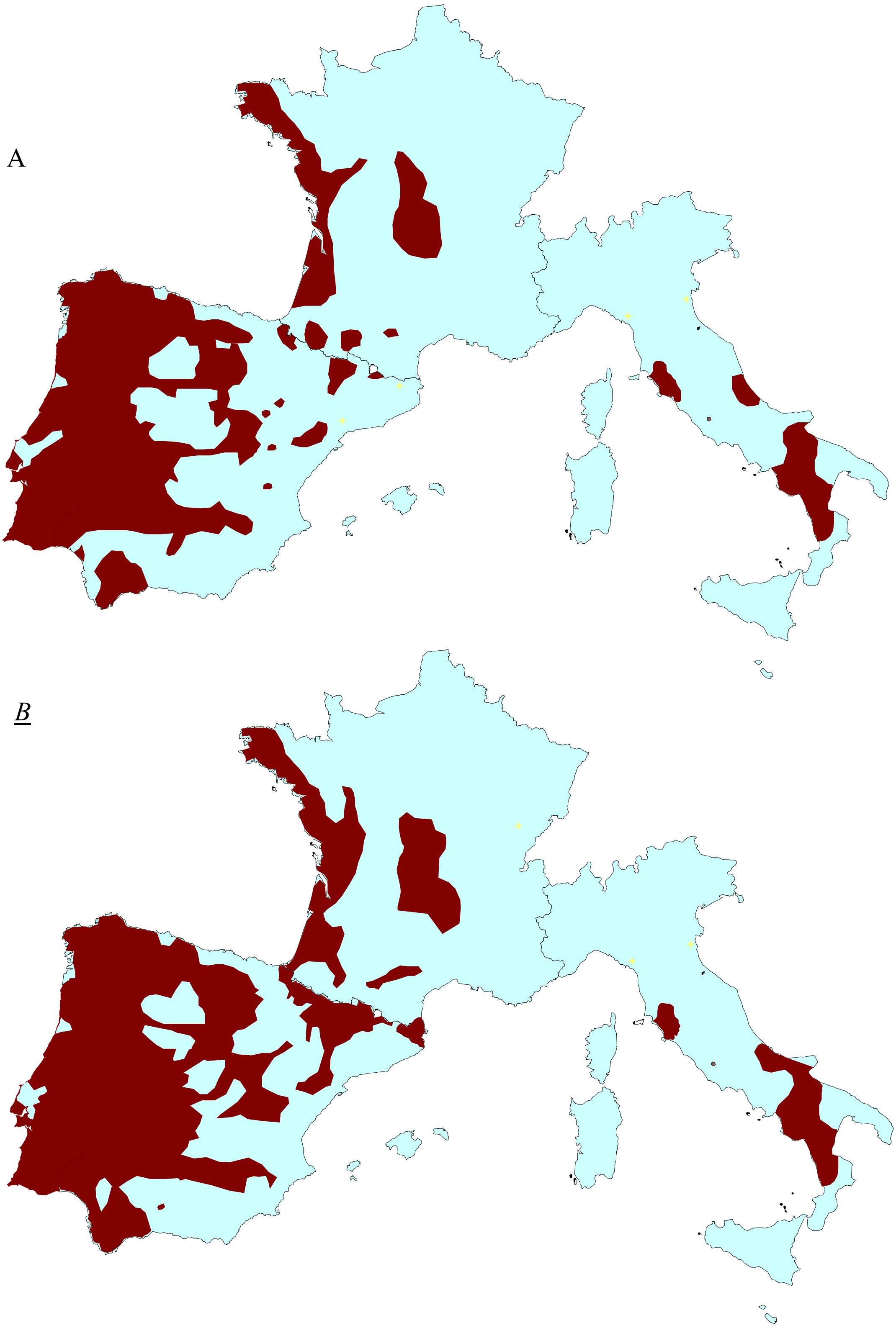


Figure 1. Distribution of the otter in SW Europe: a) 1980-85 and b) 1995-2000 (GREEN & GREEN, 1981; MACDONALD & MASON, 1982, 1983; ELLIOT, 1983; SANTOS-REIS, 1983; DELIBES & CALLEJO, 1983; BOUCHARDY, 1986; CASSOLA, 1986; DELIBES, 1990; ROSOUX *et al.*, 1996; PRIGNIONI, 1997; RUIZ-OLMO & DELIBES, 1998; TRINIDADE *et al.*, 1998).

In order to assess the impact of pollution and to explain the decline of the otter populations, as well as endowing managers with conservation measures, several research projects were begun in the 1980s. This paper is a synthesis of the current knowledge on the effect of pollution on otter populations and their principal prey in South-west Europe. At the same time, an analysis of the effect that pollution dynamics and its control could have on the evolution in the distribution of this species was carried out.

2 STUDY AREA

The zone under consideration includes France, Italy, Portugal, Spain, Andorra, Monaco and San Marino. In the remaining countries of continental Western Europe (Netherlands, Belgium, Luxembourg, Switzerland and West Germany) otters disappeared, with only small populations surviving in the north of Germany (MACDONALD and MASON, 1994). The area covered in this paper is nearly 1,418,450km^2, and is bounded by the Atlantic Ocean and the Mediterranean Sea. This determines the climate notably, with a different climate in the Mediterranean area in the south (dry, especially in summer), in the Atlantic area to the west and north (wet). There is also a continental trend in the higher zones; the maximum altitudes of which are in the Alps (4,807m), the Pyrenees (3,404m) and the Sierra Nevada (3,478m). This determines a strong altitudinal, microclimatic and biogeographical gradient. The Atlantic and continental hydrographic networks are dense in watercourses with more or less permanent river flow. In the Mediterranean area, watercourses have a tendency to be of low density and of medium or low flow (or even dry), with minima in summer. Such rivers often show catastrophic flows, especially in autumn or spring, and have a great capacity for movement and cleaning of the river bed.

Temperatures are higher in Mediterranean rivers and lakes, especially during the summer. This means that great rates of metabolism could occur that in extreme cases determine the occurrence of anoxy situations.

All these differences could be important in pollution tolerance capacity or could mean its dilution, with the situation in Northern and Central Europe being different from that in Southern Europe, and also with differences in Southern Europe itself.

3 OTTER DISTRIBUTION 1980-85

Formally, the otter was widespread in most of aquatic European ecosystems up to the first half of the 20th century. Between these dates and the beginning of the 1980s a dramatic decline took place in Western Europe, and resulted in this species becoming extinct in some areas and/or restricted to small areas of previously occupied zones (MACDONALD and MASON, 1994). Otter surveys, based on the search for indirect signs of the species' presence (spraints [faeces] and tracks) (MASON and MACDONALD, 1986), allowed for an improvement in the precision and standardisation of the study of its distribution. Thereby, the distribution of the otter in our study areas has been described by several authors (Fig. 1a) (GREEN and GREEN, 1981; MACDONALD and MASON, 1982, 1983; ELLIOT, 1983; SANTOS-REIS, 1983; DELIBES and CALLEJO, 1983; BOUCHARDY, 1986; CASSOLA, 1986; DELIBES, 1990). The otter had disappeared from Andorra, where it was cited before (RUIZ-OLMO and GOSÁLBEZ, 1988), and other small states. In France there were only stable populations in the Departments on the Atlantic Ocean to the south of Normandy, including Normandy and the Massif Central, with isolated cores in the adjacent zones. In Italy, the otter had practically become extinct, having been reduced to some rivers in the south of the country (it was only found in 8.2% of the surveyed points in the southern half), and two rivers to the north of Rome (Fiora and Farma-Merse). There were also some isolated individuals in other zones near these. In

Portugal, the otter was distributed throughout the country (found at 70% of the survey sites). Finally, in Spain the otter was found in 40% of the surveyed sites in 1981 and in 33% in 1984-85. Otters were more abundant in the western half of the country, being absent from the more industrialised zones and the big cities and their surroundings and from the main intensive agriculture areas.

The process of decline has been well reported in various areas, mainly occurring between 1950 and 1985 (RUIZ-OLMO and GOSÁLBEZ, 1988; LODÉ, 1993; ROSOUX, TOURNEBIZE and MAURIN, 1996).

4 OTTER DISTRIBUTION AND POLLUTION: THE OTTER AS A BIOINDICATOR SPECIES

As causes of decline, various factors have been suggested that could affect *L. lutra*. These include: persecution; habitat changes; disturbance; decrease in food and water availability or illness, although in the last 15 years the possible role of pollution has grown in importance as a global factor that could explain this simultaneous decline, in a relatively short period of time (OLSSON and SANDEGREN, 1983; MASON and MACDONALD, 1986; MASON, 1989, 1997).

The effect of the pollution on the otter's distribution in our study areas has been demonstrated by several works (ADRIÁN, WILDEN and DELIBES, 1985; RUIZ-OLMO, 1985; LAFONTAINE, FORTUMEAU and MAINSANT, in press; RUIZ-OLMO, LÓPEZ MARTÍN and DELIBES, 1998). These authors found otters at significantly lower frequencies in many polluted areas.

For this reason, the otter has been suggested as a good bioindicator species of water quality and riparian habitat conservation, because of its sensitivity to pollution, to the transformation of its habitats and to the changes in the availability of prey species. RUIZ-OLMO *et al.* (1998) compared the distribution of the otter with those of the main orders of macroinvertebrates and the most-used indices: BMWP (Biological Monitoring Working Party) and ASPT (Average Score per Taxon) (HELLAWELL, 1978; ARMITAGE, 1983) in order to test if both behave in the same way (macroinvertebrates are commonly used as bioindicators of contamination and of quality of water; see VALENTINE, 1973; WESTMAN, 1978). According to these results, in northeast. Spain, otters act as a pollution bioindicator species. However, they found differences related to the biological, ecological and ethological characteristics between otter and macroinvertebrates. The use of the otter as a bioindicator thus needs revision. Invertebrates, otters and, surely, other species could be used as bioindicators in a complementary fashion.

5 LEVELS OF ORGANOCHLORINES AND HEAVY METALS IN OTTER TISSUES

What types of pollutants are therefore affecting otters?. A wide range of different pollutants could affect them directly or through their prey (MASON, 1989). Among these, organochlrine compounds and heavy metals should be highlighted (OLSSON and SANDEGREN, 1983; MASON, 1989, 1997; MASON and MACDONALD, 1986). MACDONALD (1991) linked pollutant distribution, in particular the polychlorinated biphynels (PCBs), to wind circulation (that transports micropollutants subsequently deposited by rain), the distribution of the main sources of pollution and the distribution of the otter. Otters have healthy populations in the territories next to the Atlantic Ocean or further south, away from these pollutant carrying winds. These tend to be located more to the west (Portugal, Galicia, Extremadura, Western France, Ireland, Scotland, Norway and an area of Denmark).

In Table 1 the levels of organochlorine compounds and heavy metals found in the muscular tissues of otters from France and Spain are shown. The levels of PCBs are the more elevated in all cases. However, high levels of some pesticides were found in Spain (mainly DDTs in the south, an average of nearly 80mg/kg: lipid weight) and also mercury in France (although the geometric mean is not very high, 4.71mg/kg), with a maximum of 41mg/kg, whereas the threshold of 30mg/kg has been proposed, beyond which neuronal damages are produced in mammals (GUTLEB, 1995). Locally, there are high levels of most of the analysed compounds (especially some heavy metals, oxychlordane or lindane), but these are not related to a possible decline of the species on a national/ international scale.

For these reasons, PCBs should have the greatest effects on otter survival from the studied regions, although locally other pollutants could have more influence and, in general, the effects should be added.

The arithmetic means of PCB levels (between 6.15 – 33.94mg/kg: lipid weight) are lower in three of the four studies than the threshold of 50 mg/kg, a level proposed as an upper limit for the presence and distribution of the otter (see MASON and MACDONALD, 1993). This is based on the results from JENSEN *et al*. (1977) for American mink (*Mustela vison*). They are also lower or similar to the 30mg/kg for the same species proposed by LEONARDS *et al.* (1994) and SMIT *et al*. (1996). The exception concerns data from our Spanish sample collected during the period 1981–1993 and from Doñana (1982–1983). The arithmetic mean of the whole country data (78.30mg/kg: lipid weight) exceeded both thresholds. However, the sample from the whole of Spain is very heterogeneous due to the geographical amplitude and difference in uses between zones; in the second case the sample is too small. In fact, RUIZ-OLMO, LÓPEZ-MARTÍN and DELIBES (1998) highlight important differences between the diverse regions (greater contamination in the southwest than in the north of the country), and even between river basins. In France, there are also differences between basins (LAFONTAINE, 1995). The presentation of results as arithmetic means could mask the real situation, since extreme values (of up to 1005mg/kg in South-west Spain) can affect the average values. For this reason LAFONTAINE (1995), following the arguments of NEWTON and WYLLIE (1992) and SMIT *et al*. (1994), used the geometric mean for his samples from France, while using both means as part of a pan-European comparative synthesis (see also Table 1). In Spain, only 28% of the otters analysed in the period 1981-1993 exceeded the threshold of 50mg/kg (RUIZ-OLMO, LÓPEZ-MARTÍN and DELIBES, 1998), being almost the same (29%) as in French otters from 1987-1995 (according to the sample studied by LAFONTAINE, 1995).

Table 1. Arithmetic ([a]) and geometric ([g])mean levels (mg/kg lipid basis in muscular tissues) of organochlorine compounds and heavy metals (mg/kg dry weight in liver) of otters from France and Spain (the range is given between brackets, except for France b: standard deviation). For Spain a (Doñana), levels are expressed as mg/kg wet weigh), and heavy metals were analysed in muscular tissue. Underlined values are expressed as wet weigh.

	France a (5 regions) (*n* = 8 to 22). Mostly 21 or 22	**France b (Western Marshes, 3 regions) (*n* = 32)**	**Spain a (Doñana) (*n* = 5)**	**Spain b (whole) (*n* = 41: *n* = 19 for heavy metals)**	**Spain c (whole) (*n* = 10)**
PCBs	33.94[a] 13.99[g] (1.24 - 145.31)	26.19[a] (± 21.74)	2.44 [a] (2.40- 2.45)	78.30[a] 25.06 g (1.49 - 1005.59)	6.15 [a] (n.d. – 20.60)
DDTs	1.10[a] 0.38[g] (0.01 - 6.08)	(not detailed)	3.50 [a] (2.25 - 5.62)	14.79[a] 5.78 [g] (0.19 - 82.95)	1.28 [a] (0.38 – 2.97)
Oxychlordane	0.55[a] 0.28[g] (0.002 - 1.95)	-	-	-	-
BHC	0.17[a] 0.08[g] (0.005 - 0.67)	0.08[a] (± 0.07)	-	-	-
HEPO	0.124[a] 0.068[g] (0.009 - 0.546)	0.003[a] (± 0.010)	-	0.26[a] (n.d. - 1.53)	-
Dieldrin	0.70[a] 0.33[g] (0.12 - 2.91)	0.75[a] (± 1.45)	-	-	-
Aldrin	-	0.02[a] (± 0.07)	-	0.24[a] (n.d. - 5.84)	-
Lindane / HCH-γ	0.47[a] 0.10[g] (0.01 - 3.27)	0.18[a] (± 0.46)	0.02 [a] (0.01 -0.02)	1.95[a] 2.50 [g] (0.03 - 9.92)	-
Hg	9.46[a] 4.71[g] (0.50 - 41.00)	-	1.33 [a] (1.25 - 1.41)	0.99[a] (n.d. - 2.80)	-
Pb	0.55[a] 0.42[g] (0.12 - 1.58)	-	0.64 [a] (0.51 - 0.80)	0.09[a] (n.d. - 0.34)	-
Cd	0.35[a] 0.17[g] (0.01 - 2.03)	-	0.13 [a] (0-10 - 0.17)	0.04[a] (n.d. - 0.22)	-
Cu	28.78[a] 26.00[g] (11.00 - 53.00)	-	-	-	-
As	0.070[a] 0.055[g] (0.02 - 0.20)	-	-	-	-
Cr	1.51[a] 0.63[g] (0.07 - 4.90)	-	-	0.17[a] (n.d. - 0.49)	-
Ni	2.80[a] 1.51[g] (0.10 - 4.34)	-	-	-	-
Zn	77.33[a] 77.00[g] (70.50 - 90.70)	-	-	-	-
Reference	Lafontaine, 1995	Tans *et al.*, 1995	Hernández *et al.*, 1985	Ruiz-Olmo *et al.*, 1997	Ruiz-Olmo & López-Martín, unpublished
Period	1987-1995 except fpr one idividual in 1981	1987-1994	1982 -1983	1981-1993	1994-1999

On the other hand, the most recent studies on the effect of the pollutants on the American mink (KIHLSTRÖM *et al*., 1992; LEONARDS *et al*., 1994) have shown that the effect of PCBs on reproduction is progressive and asinthothic. Some American mink can breed with low success even with high pollutant levels in tissues. For this reason it seems appropriate to carefully consider the use of such static thresholds, especially when remembering that resistance to these levels could be variable in each species, according to the individuals and/or the populations (SMIT *et al*., 1994). Organochlorine compounds can also be eliminated by female otters during gestation and weaning periods and by glandular secretions in both sexes. In addition, the body condition and the mobilisation of fat reserves and intoxication of the organism for these lipophilic pollutants must be considered in each case.

RUIZ-OLMO, LÓPEZ-MARTÍN and DELIBES (1998) found no correlation between the levels of any organochlorine compound or heavy metals, and the age of the otters (neither as a whole, nor by sex). This negative result could be affected by the great heterogeneity of the environments and to the great geographical dispersion of samples. LAFONTAINE (1995), found no significant correlation between the age and the levels of several organochlorine compounds (samples also originating from different zones). However, he found a positive significant correlation between the age of the otters and the levels of some heavy metals (mercury: $r^2 = 0.988$: $p < 0.09$ in liver; cadmium: $r^2 = 0.994$: $p < 0.05$ in kidney; lead: $r^2 = 0.976$: $p < 0.13$ in kidney).

6 CORRELATION BETWEEN POLLUTION LEVELS AND OTHER VARIABLES

In Table 2, the functions between the levels of main pollutants are shown. In the case of significant results, the high correlation found, allows us to use, for discussion, a single tissue of the most representative compounds (in this case, the PCBs).

Table 2. Correlation between the body condition index (KRUUK et al., 1987; LAFONTAINE, 1995; RUIZ-OLMO, 1995) and levels of several pollutants in otter tissues from France and Spain (after LAFONTAINE, 1995, and RUIZ-OLMO, LÓPEZ MARTÍN and DELIBES, 1998). (*) significant differences.

		France	Spain
PCBs	**Muscle**	$r = 0.580$ $p < 0.05$*	n.s.
	Liver	$r = -0.501$ $p < 0.05$*	n.s.
HCB	**Muscle**	$r = -0.498$ $p < 0.05$*	n.s.
	Liver	$r = -0.680$ $p < 0.01$*	n.s.
Cu	**Liver**	$r = -0.674$ $p < 0.01$*	n.s.
Hg	**Liver**	n.s.	$r = 0.561$ $p = 0.19$

LAFONTAINE (1995) found a significant correlation between the body condition index, *k* (KRUUK, CONROY and MOORHOUSE, 1987), for French otters (LAFONTAINE, 1995) and the levels of some pollutants (Table 3). However, RUIZ-OLMO, LÓPEZ-MARTÍN and DELIBES (1998) found no similar significant correlation. Again, the geographical dispersion of the samples and their heterogeneity could explain the lack of correlation for this study.

Table 3. Functions, correlation coefficients and signification between the levels of some pollutants in the otter tissues from France and Spain (after LAFONTAINE, 1995 and RUIZ-OLMO, LÓPEZ MARTÍN and DELIBES, 1998). First pollutant is y and second x. (*) significant differences.

y / x	France	Spain
PCBs liver / PCBs muscle	log y = 0.967 log x + 0.012 $r = 0.57$: $p < 0.01$	log y = 0.758 log x + 0.441 $r = 0.81$: $p < 0.0001$
DDTs liver / DDTs muscle	log y = 0.426 log x + 1.729 $r = 0.50$: $p < 0.05$	log y = 0.667 log x + 0.585 $r = 0.65$: $p < 0.0001$
PCBs / DDTs (muscle)	$p > 0.25$	log y = 0.475 log x + 1.026 $r = 0.55$: $p < 0.0003$
PCBs / DDTs (liver)	$p > 0.25$	-
DDTs (muscle) / Hg (liver)	log y = 0.599 log x + 2.136 $r = 0.44$: $p < 0.05$	$p < 0.05$
DDTs (liver) / Hg (liver)	$p > 0.25$	$p < 0.05$

LAFONTAINE (1995) also found a positive correlation ($p < 0.02$) between the levels of lindane (γ-HCH) in the muscle of individual otters from Brittany, France, and the rate of maize culture (on a local scale), where lindane was widely used. A similar case could be found in South-west Spain, with a large agricultural area and higher use of some pesticides and where the levels of some pesticides (DDTs, lindane, etc.) were greater than those used in the north (RUIZ-OLMO, LÓPEZ-MARTÍN and DELIBES, 1998).

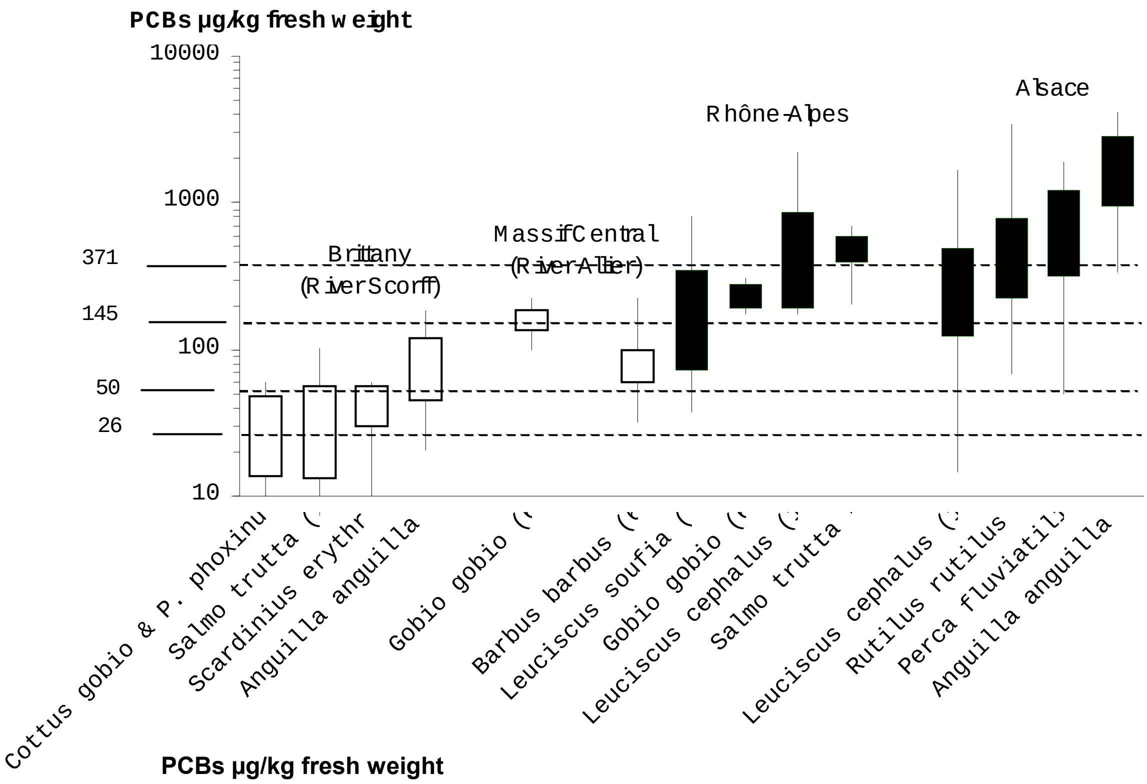
7 LEVELS OF ORGANOCHLORINES AND HEAVY METALS IN TISSUES OF FISH AND CRAYFISH AS OTTER PREY

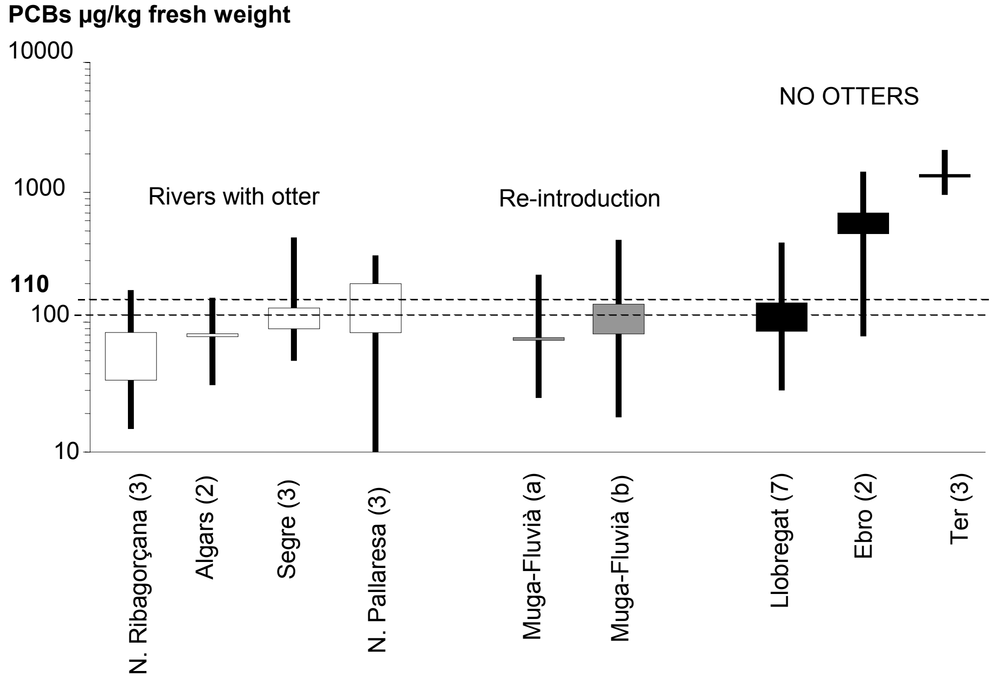
The accumulation of organochlorines in otter tissues could only come from food, specifically fish, the main diet of the species. RUIZ-OLMO and LÓPEZ-MARTÍN (1994) found that the populations of otters from Catalonia (North-east Spain) were distributed in zones with less than the average level (arithmetic mean) of 0.1mg/kg: wet weight of PCBs in the muscle of fish (referring to the group of the fish consumed by the otter in each site).

LAFONTAINE and DE ALENCASTRO (2000) found quite a close relationship between the levels of PCBs in fish from different regions of France (Brittany, Massif Central, Rhône-Alpes and Alsace) and otter occurrence (Figure 2) (also see MICHELOT *et al*., 1998).

After these findings, the critical thresholds for contamination effects on fish could be between 0.15-0.20mg/kg: fresh weight of PCBs. In most areas, the average levels of PCBs in fish tissues from sites used by otters were under 0.10mg/kg, and are similar to those found in the previous work, and are comparable to the 0.145mg/kg proposed by LEONARDS *et al*. (1994); however, we need to keep in mind that there are differences between levels in muscle and the whole fish. They are higher than 0.026 - 0.05mg/kg levels presented by MACDONALD and MASON (1994), for the whole data, and for eels, respectively.

Figure 2. a) Synthesis of the average values, ranges ans interquartiles of PCBs in 11 fish species from four French regions, showing samples coming from the otter range (grey) and outside (white) (after LAFONTAINE & DE ALENCASTRO, 2000). b) Average values, ranges and interquartiles of PCBs levels in fish muscle from nine basins in Catalonia (N.E. Spain), showing samples coming from the otter range (grey), outside (white) and from the re-introduction area (recalculated data from LÓPEZ-MARTíN *et al*., 1995, and MATEO *et al*., 1999).





In North-east Spain, levels found by LÓPEZ-MARTÍN, RUIZ-OLMO and BORRELL, (1995), from the different basins (unpublished data), and plotted in Figure 2, coincide again with the results of LAFONTAINE and DE ALENCASTRO (2000), and with all otter populations in sites under 0.14 mg/kg: fresh weight of PCBs in fish muscle. In rivers not used by otters, average levels were often over 0.2mg/kg, reaching up to 1.34mg/kg. We must highlight the basins of the Muga and Fluvià Rivers, where, despite the fact that the otter became extinct toward the end of the 1970s (RUIZ-OLMO and GOSÁLBEZ, 1988), levels in fish tissues found at the beginning of the 1990s were compatible with the presence of the mustelid. This has allowed for the development of a reintroduction project (SAAVEDRA and SARGATAL, 1998) in which nearly 50 individuals have been relocated.

Results from France and Spain confirm those from Italy, since levels of PCBs within the otter range are much lower that those from zones without otters (VIVIANI *et al*., 1974; GALASSI and GANDOLFI, 1981; GALASSI, GANDOLFI and PACCHETTI, 1981; CANTONI, CATTANEO and FABRIS, 1985; COZZANI and PIETROGIACOMO, 1985; TURSI, CONSTANTINO and MATARRESE, 1989). In the Ticino River, where there is a reintroduction project, PCBs levels seem to be appropriate for their return (NARDI *et al*., 1993), while they could be less compatible with those of DDTs.

RUIZ and LLORENTE (1991), in the 1980s, found high levels of PCBs and DDTs in tissues of fish from Ebro delta (Northeast Spain), where otters had become extinct in the 1970s.

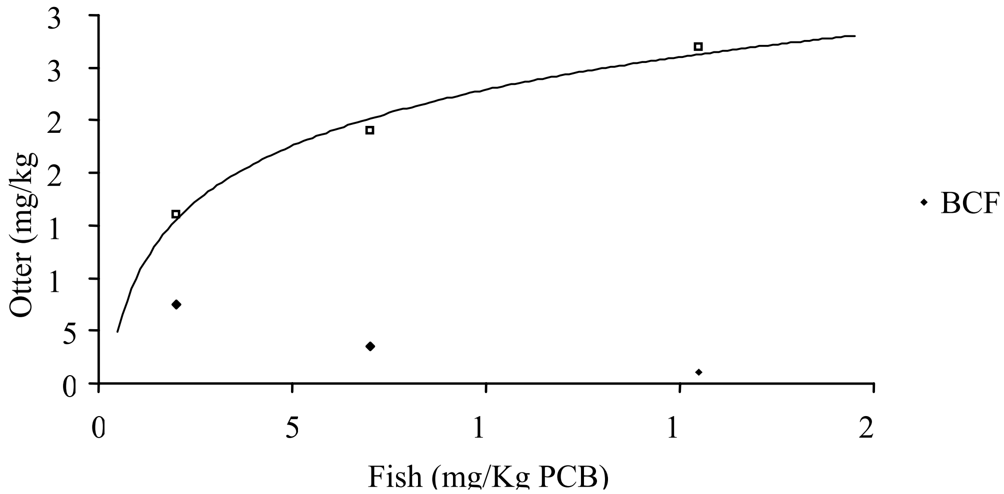
There are several studies on heavy metal levels in fish tissues from Italy; TURSI, CONSTANTINO and MATARESSE (1989) and FUMAGALLI and PRIGNIONI (1991) found very low levels in fish from Italian rivers with otters; the higher levels were mercury, being in almost all cases below the 0.2mg/kg: fresh weight. In other Italian rivers, with no otters, mercury levels were lower (BEGLIMONDI, FRAVOLINI and MOROZZI, 1975; LOCHT *et al.,* 1981). In Spain, RALDUA and PEDROCCHI (1996) found average mercury values between 0.74 and 2.80mg/kg: wet weight, in fish from Huesca (Aragon, Northeast Spain), outside the otter range or on the border of the otter distribution towards 1993. In Portugal, SANTOS-REIS, AFONSO and FREITAS, (1995), found high mercury levels in American crayfish, *Procambarus clarkii*, from the Tejo basin, this being the main otter prey in many areas; levels were highest in Almonda River (mean: 0.21mg/kg) and Server River (mean: 0.29mg/kg).

In some rivers, lead, chromium or nickel levels were very high (CAGGINO, 1982), and could have contributed in an important way to the decline of the otter.

8 BIOMAGNIFICATION FACTORS

In Spain, most of our studied otter populations feed mainly on fish (in the zones where we have studied bioconcentration factors: 90-95%; RUIZ-OLMO and PALAZÓN, 1997). This allowed us a better approach to the bioconcentration factors (considered as the ratio: levels in otter tissue/levels in fish tissue), than in other otter populations with a more complex diet. The only bioconcentration factors to PCBs and DDTs on the whole (LÓPEZ-MARTÍN and RUIZ-OLMO, 1996; RUIZ-OLMO, LÓPEZ-MARTÍN and DELIBES, 1998). These authors base their data on the analysis of the most-caught fish (the most abundant in the environment) and its importance in otter diet. Levels are drawn in Figure 3.

Figure 3. Biomagnification factors for otter and fish tissues in otters from NE Spain (RUIZ-OLMO *et al.,* 1995; LAFONTAINE & DE ALENCASTRO, 2000).



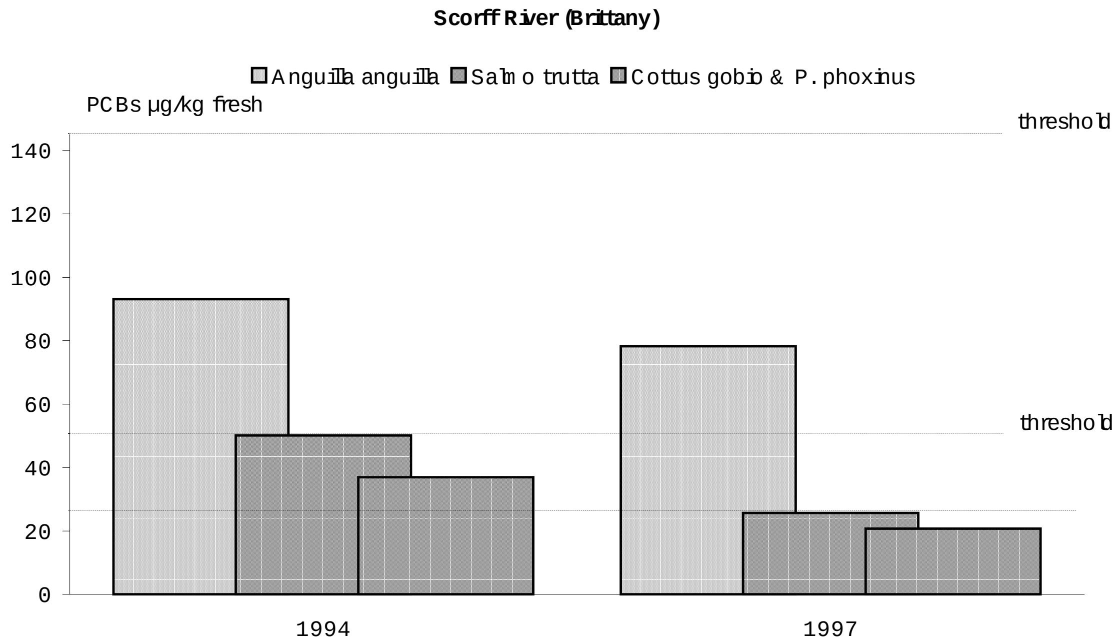
Biomagnification factors in PCBs were double those of DDTs (i.e., they accumulated double). For PCBs, a logarithmic function is followed (r = 0.99: p = 0.021), very similar to the results from the Canadian river otter *Lutra canadensis* (FOLEY *et al*., 1988). Bioconcentration values ranged between 2.2 and 8.2 in the case of PCBs (lipid weight), and between 0.9 and 4.8 in the case of DDTs.

For the River Scorff, Brittany (North-west France), from 1994 to 1997, LAFONTAINE and DE ALENCASTRO (2000), found PCBs biomagnification factors varying from 22.2 to 44.9 (wet weight) for otter tissues *vs* all the fish species together, and from 584 to 3236 (wet weight) for otter tissues *vs* sediments.

RUIZ-OLMO, LÓPEZ-MARTÍN and DELIBES, (1998) found PCB congeners No.138, 153, 170, 180, 194, 195, 196/ 203 and 201, in greater proportion in otter tissues than in fish, there being a significant bioaccumulation among fish in congeners No. 101, 138, 141, 153, 170, 174 and 180. LAFONTAINE (1995) found greater prevalence of congeners 138, 153, 170 180, for the 12 analysed, the data agree with the Spanish results. MASON and RATFORD (1994) found that the congeners No 138, 153, 163, 180 and 187 are the most prevalent in British otters and Nos 118, 138, 153, 163 and 180 in Danish otters. For otters from Denmark, SMIT *et al*. (1996) found a greater bioconcentration of the congeners Nos. 138, 180, 156, 157, 189, 126 and 169. On consideration that they are the most bioacumulated congeners and could have greater weight in the toxicity, they utilise an index that includes these seven congeners only (∑ 7 PCBS), or with the Toxicity Equivalent Concentration (TEQ) (SAFE, 1994; LEONARDS, 1997). These methods allow the real power of toxicity for each concentration of PCB congeners, but differences between studies must be explained.

For the River Scorff, Brittany (North-west France), from 1994 to 1997, LAFONTAINE and DE ALENCASTRO (2000), analysed for 17 CB congeners, including three coplanars, and found the cumulative contribution (p <0.001) of the most toxic CBs (groups 1A + 1B + 2, according to the typology of MACFARLAND and CLARKE (1989) along the food chain, as follows: sediments > *Scardinius* *erythrophtalmus* > *Salmo trutta* > *Cottus gobio* & *Phoximus phoxinus* > *Anguilla anguilla* > otter spraints > otter tissues (Figure 4).

Figure 4. Evolution of pollution levels in fish tissues from Scorff River (Brittany).



The proportion of congeners in each zone depends on the type of PCB (mixture) that the otters/fish have ingested (with the congeners 138, 153, and 180 found in great proportion in all the studies). The several industrial uses of the zones could characterise the proportion of each PCB in each zone.

9 EVOLUTION OF POLLUTION LEVELS

Unfortunately, otter carcasses were not analysed for contaminants between1950 and 1970 in the countries discussed here. This was the period when many of the organochlorine compounds and heavy metals were being used indiscriminately, widely and without control. The only data available refer to fish from the Po delta, Northern Italy. They show high levels (VIVIANI *et al*., 1974), with maxima up to 12mg/kg: wet weight for PCBs in the liver of *Gobius paganellus* and 6.13mg/kg in the liver of *Squalus acanthias*. If these species were consumed in significant numbers by otters, they could give rise to high concentrations in otter tissues (remember the thresholds for otter presence of 0.05-0.2mg/kg in fish). These authors found high levels of DDTs (up to 9mg/kg in some species) and lindane (up to 1.3mg/kg).

Lack of historical information prevents us from carrying out an analysis of the evolution and dynamics of several compounds in otters and their prey from those decades. Also, analytical methods and accuracies have changed during this time. However, we found a decrease in the levels of PCBs, DDTs and other pollutants in otter tissues (Table 1) and fish tissues (Figure 4: Table 4). These data agree with MASON (1998) in that they demonstrate a decrease in these pollutants.

Table 4.- Comparison between the average PCB levels in fish tissues from some rivers of NE Spain in two different periods (after LÓPEZ-MARTÍN & RUIZ-OLMO, unpublished).

		1990-92		1998-99		
Basin	**Site**	**PCBs**	**DDTs**	**PCBs**	**DDTs**	**Fish species**
N. Pallaresa	Gerri de la Sal	0.145 (0.06-0.30)	0.087 (0.02-0.2)	0.065 (0.04-0.09)	0.030 (0.01-0.05)	*S. trutta*
Segre	Seu d'Urgell (1)	0.207 (0.04-0.45)	0.235 (0.05-0.16)	0.212 (0.191-0.436)	0.128 (0.003-0.253)	*S. trutta* *C. gobbio* *C. toxostoma*
	Balira (1)			0.13 (0.07-0.19)	0.072 (0.037-0.107)	*S. trutta*
	Ponts	0.095 (0.05-0.24)	0.052 (0.02-0.07)	0.040 (0.005-0.08)	0.102 (0.01-0.02)	*S. trutta* *C. carpio* Others

(1) Sampling stations downstream Southern Andorra border

Heavy metal and other organochlorine compound levels found in Spain, France and Italy since the 1980s have been generally low, with the exception of mercury in some areas in France (LAFONTAINE, 1995) and Portugal (SANTOS-REIS, ALFONSO and FREITAS., 1995), in some small rivers of Tejo Basin and in Aveiro Ria, (Nuno Gomes, *pers. comm.*). This decrease in pollution levels (not only of PCBs but in general), would contribute to an explanation of otter recovery in several areas.

10 OTTER DISTRIBUTION 1995-2000: THE RESPONSE OF A BIOINDICATOR SPECIES

Otter distribution has changed since 1985, with a tendency towards increase and recovery in wide areas (ROSOUX, TOURNEBIZE and MAURIN, 1996; PRIGNIONI, 1997; RUIZ-OLMO and DELIBES, 1998; TRINDADE, FARINHA and FLORENCIO, 1998; CONROY and CHANIN, 2001). Figure 1 shows this increase in otter distribution, with an evident spread in several areas of central France and in zones of Spain (Pyrenees, Centre, Andalucia, etc.). Even in Portugal, where the otter was previously widespread, better results were recorded in 1995 (89% of positive sites).

However, the animal has not recovered in other areas of these countries, nor in Andorra or small states. In Andorra, the Balira Basin crosses the country and has high pollution levels which could explain the absence of otter in 1999, when we carried out an otter survey for the new Otter Action Plan (RUIZ-OLMO, unpublished). High contaminant values obtained in 1991 in fish tissues from the Segre River, just beyond the junction with Balirain Spanish stretch (LÓPEZ-MARTÍN, RUIZ-OLMO and BORRELL, 1995), are in accordance with the lack of otters. Even in 1998-99 we found 0.13-0.21 mg/kg: wet weight in muscle of brown trout from the Rivers Segre and Balira, just downstream of Andorra. The Balira River was determined to be the most polluted in the Spanish Pyrenees by both the Water Suitability Index and water analysis (CONFEDERACIÓN HIDROGRÁFICA DEL EBRO, 2000). This could explain the absence of otters.

This otter recovery in wide areas is attributable to assistance in and an improvement of the general conditions for the species, especially pollution levels. On the other hand, the lack of otter recovery in other parts (with a good otter habitat index), could be a result of these conditions not improving enough or even worsening conditions, especially pollution.

11 DISASTERS AND OTHER CONTAMINANTS

Up to now, the effects of more widespread pollution types have been analysed. However, big disasters, producing negative effects on otters and their prey, should be emphasised. In analysing these types of occurrence, we can divide them into two types:

a) <u>Oil spills</u>. They have taken place mainly along the Atlantic coasts after ship accidents. In Spain, those from the *Urquiola* (May 1976) with 28.1 million gallons, and the *Amocco Cadiz* (1992) with 21.9 million gallons, can be highlighted. They affected the coast of Galicia. Recently (2000) an unidentified oil slick impacting several kilometres was found off the east coast of Cadiz (Southern Andalucia). Although in the first two instances, effects on some fish species, invertebrates and plants were important, no dead shore-living otters were found. These types of accidents have also occurred in rivers, like the Tajo River in Central Spain, in August 2000. There are no studies on the effects of these accidents on otter populations. However, years later, otters still inhabit some of these zones.

   A similar disaster occurred in December 1999, along the North-west coast of France (Brittany). The *Erika* oil spill polluted several hundreds of kilometres of sea-shore. Invertebrates, fish and especially several thousands of seabirds were affected. No dead shore-living otter was found however. This does not mean that this oil spill had nor will have no impact on otter survival, and, in terms of accumulation in the food chain, a study is now starting to analyse PAHs in both otter spraints (from coastal areas) and marine mammal tissues (LAFONTAINE and HASSANI, in progress).

b) <u>Break-up of toxic reservoirs</u>. This is especially highlighted by the case of the Aznalcollar mine (May 1998), on the Guadimar and Guadalquivir Rivers, Huelva (Southern Spain). A high quantity of heavy metals was released into these rivers as a result of mineral cleaning. Arsenic was the main metal, with more than 4000 mg/kg in sediments in three of nine sampling sites. Manganese, cadmium, chromium, copper, lead (more than 3000mg/kg at three of the sites), zinc and iron were also found at high levels. A monitoring program on physical and chemical parameter effects and on biotic elements has been carried out (Spanish Ministry of the Environment, in progress). In sediments, a decrease in levels was found in arsenic, iron, manganese and nickel, although no decrease was shown in the remainder. An increase in chromium, copper and lead was registered.

c) <u>Mines</u>. Heavy metals are found in high levels in waters used for mining activities, in Northern Portugal, for example, these affected pH levels and sediments (TRINDADE, FARINHA and FLORENCIO, 1998).

On the other hand, perhaps different pollutants are still active or could well begin to be so in the near future. A study of the effect of dioxines, furans, PVCs and organophosphorates, etc., should be made.

12 CONCLUSIONS

The results show the negative effect of contamination on otter distribution in France, Italy, Portugal, Spain and Andorra. This could explain the great decline during a relatively short period (three decades) and could have been coincidental with other factors such as habitat transformation, decrease of food availability and persecution, and other.

The study area is very extensive and is characterised by a high diversity of weather as well as hydrological, ecological and human characteristics. This is of great importance in understanding how pollution could affect otters. In fact, there are examples of pollution from several sources that have determined the distribution of the otter in some specific zones (organic pollution, acidification of water, detergents, etc.). However, only a limited series of compounds (micropollutants) producing lethal effects at a population level and sublethal (for example, affecting breeding) could have had a more global effect, in accordance with a disappearance on a regional scale (MASON, 1989; MACDONALD, 1991). These, especially, are the organochlorine compounds (from industrial sources and agriculture) and heavy metals. Of these, PCBs seem to be the most widespread in fish and otters, attaining higher levels of bioaccumulation in tissues. However, in specific regions, other biocides (mainly DDTs, oxychlordane, dieldrin and lindane) (CHANIN and JEFFERIES, 1978; MASON, 1989) and heavy metals (mainly mercury) (GUTLEB, 1995; KRUUK and CONROY, 1996; KRUUK, 1997) could have had a more important effect than PCBs.

The differences in toxicity need a more thorough analysis of pollutants, both with reference to the type of mercury (GUTLEB, 1995), and with reference to the congeners that make up PCBs (SMIT *et al*., 1994, 1996; LEONARDS, 1997).

There is increasing evidence of the effect on reproduction, and the pathological effects of these compounds on mammals (DELONG, GILMARTIN and SIMPSON, 1973; TANABE, 1988), and more specifically on otters and American mink (JENSEN *et al*., 1977; AULERICH and RINGER, 1977; KEYMER *et al*., 1988; MASON and O'SULLIVAN, 1992; MASON and MACDONALD, 1993). There is a correlation between PCB levels and the index of body condition, *k*, that has been demonstrated to determine the probability of mortality (KRUUK, CONROY and MOORHOUSE, 1987; LAFONTAINE, 1995). Lately, the effect on vitamin A (and subsequent effects) was shown (SMIT *et al*., 1996; SIMPSON *et al*., 2000).

This approach to PCBs congeners could bring with it some results applicable in the management and real understanding of the effects of these compounds. Our results also contribute to the idea of the existence of some thresholds of pollutant levels in prey, of some 0.1-0.2mg/kg: wet weight for PCBs, although important differences between the fish species exist (LAFONTAINE and DE ALENCASTRO, 2000).

Organochlorine levels have been diminishing world-wide, since preventive measures were undertaken (STOUT, 1986; BINGNERT *et al.,* 1993; NEWTON & WILLEY 1992). This decline has also occurred in the areas studied here, both in pollution levels of otter prey as well as in otter tissue levels. This decrease in levels coincides with a recovery in otter distribution. This fact tends to convince us of the effect of pollutants on otters, and also the bioindicator property of this mustelid. It answers positively to the improvement in water quality, in pollution levels in prey tissues and in the environment in general. However, it is necessary to stay on guard since in many areas this recovery is not happening and, other new compounds or substances could have similar effects to those of organochlorine compounds and heavy metals.

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DISTRIBUTION AND POPULATION DENSITY OF THE OTTER *LUTRA LUTRA* AND POLLUTION OF AQUATIC ECOSYSTEMS IN BELARUS

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1 INTRODUCTION

From the 1960s until the beginning of the 1990s, the otter *Lutra lutra* populations declined in Western Europe (MASON and MACDONALD, 1986; FOSTER-TURLEY, MACDONALD and MASON, 1990; REUTHER, 1993). According to MASON (1989), chemical pollution of aquatic ecosystems played a major role in the deterioration of the stock of water-living prey of the semi-aquatic predator and severely affected its distribution and reproduction. In Europe, monitoring of otter populations appeared to be quite often linked to contamination of aquatic ecosystems. In the last decade, however, otter numbers have been recovering in parts of Western Europe (MACDONALD, 1994; STUBBE and STUBBE, 1994; ROMANOWSKI, GRUBER and BRZEZINSKI, 1997; KRANZ and TOMAN, 2000; CONROY and CHANIN, 2001). The aim of this paper is to present current information on otter distribution in Belarus and discuss these results in connection with data on chemical pollution of aquatic ecosystems there.

2 MATERIALS AND METHODS

Information relating to otter presence at 243 sites along rivers in different parts of Belarus and several neighbouring areas (Eastern Poland, Northern Ukraine, and Pskov Region of Russia) was gathered in 1997-2000. In each place, a minimum 1km of river stretch was inspected. Because there was no financial support for this job, I carried out the otter surveys from time to time while doing another study on vertebrates, which determined the distribution of places inspected for otter presence (Figure 1).

Figure 1. Distribution of otter signs in catchments of rivers in Belarus, 1997-2000.

Multiannual (1984-2000) dynamics of otter density in the four rivers chosen as model aquatic ecosystems for long-term monitoring of the otter population were recorded. I did a census of otters on a small and medium-sized river in a protected area (respectively Volka and Zapadnaya Berezina Rivers, Naloboky Reserve, Grodno and Minsk Regions) and a hunting area (respectively Ahonka and Nischa Rivers in Vitebsk Region). A pronounced habitat-related difference in otter density was found (SIDOROVICH, 1992, 1997), and according to these results, both the small and medium-sized rivers chosen for the otter monitoring are characterised by a similar quality of habitat conditions. The rivers are moderately flowing and with moderately swamped floodplains, and the small river is a tributary of the medium-sized one. So, it is correct to compare otter distribution at similarly sized rivers in protected and hunting areas. In respect to several questions of otter distribution related pollution, other data on density of otters obtained in 1984-1996 in different regions of Belarus (SIDOROVICH and LAUZHEL, 1996; SIDOROVICH, 1997) were involved.

Each winter, on each river, more than 20km of the river bank were inspected. Otters were censused along the rivers during winter by searching the banks, ice covers and floodplains for their tracks. We tried to count the number of otters living on surveyed stretches of the rivers. To do this, we applied the following criteria. According to the methodological results related otter census (SIDOROVICH, 1992), the prints of hind feet were measured. In our field survey, differences in measurements of footprints of single otters consistently >1cm were accepted as criterion for differentiating individuals. Fresh marking places (with urine and faeces) were examined to determine sex. In relation to the prints of hind feet, males mainly leave urine marks on snow in front of a scat, whereas females defecate and urinate in the same place or urine marks are sprinkled behind the scat.

As is well known, the main contaminants are heavy metals, organochlorine pesticides, polychlorinated biphenyls (PCBs) and acid deposition. In addition to chemical pollution, many aquatic ecosystems in Belarus were strongly contaminated by the Chernobyl radionuclide fallout. All available data from both literature (KUZNETSOV and DOVNAR, 1984; JAKUSHKO *et al*., 1988; SAVCHENKO, 1992 *a* & *b*) as well as our own and joint studies (SAVCHENKO and SIDOROVICH, 1994; SIDOROVICH, SAVCHENKO and DENISOVA, 1996; SIDOROVICH, 1997) relating to the contamination of otters and aquatic ecosystems in Belarus were compared with information on otter distribution to determine the possible impacts of habitat pollution on otters. Data on the concentration of heavy metals (Pb, Cd, Zn, Cu, Mo, Cr, Ni, Ag, Sn, V, Co, Be, Ba, Mn, Ti, Zr, Y, Yb, Nb, Sc), acid deposition, organochlorine compounds (DDE, DDD, DDT, α-BHC, γ-BHC, PCBs), and radionuclides (Cs ^{134} and Cs ^{137}) were found in the literature or obtained in own and joint studies. Places sampled for any pollutants in either habitats (water and bottom sediment) or otters (mostly in muscle, liver and kidney) are shown in Figure 2.

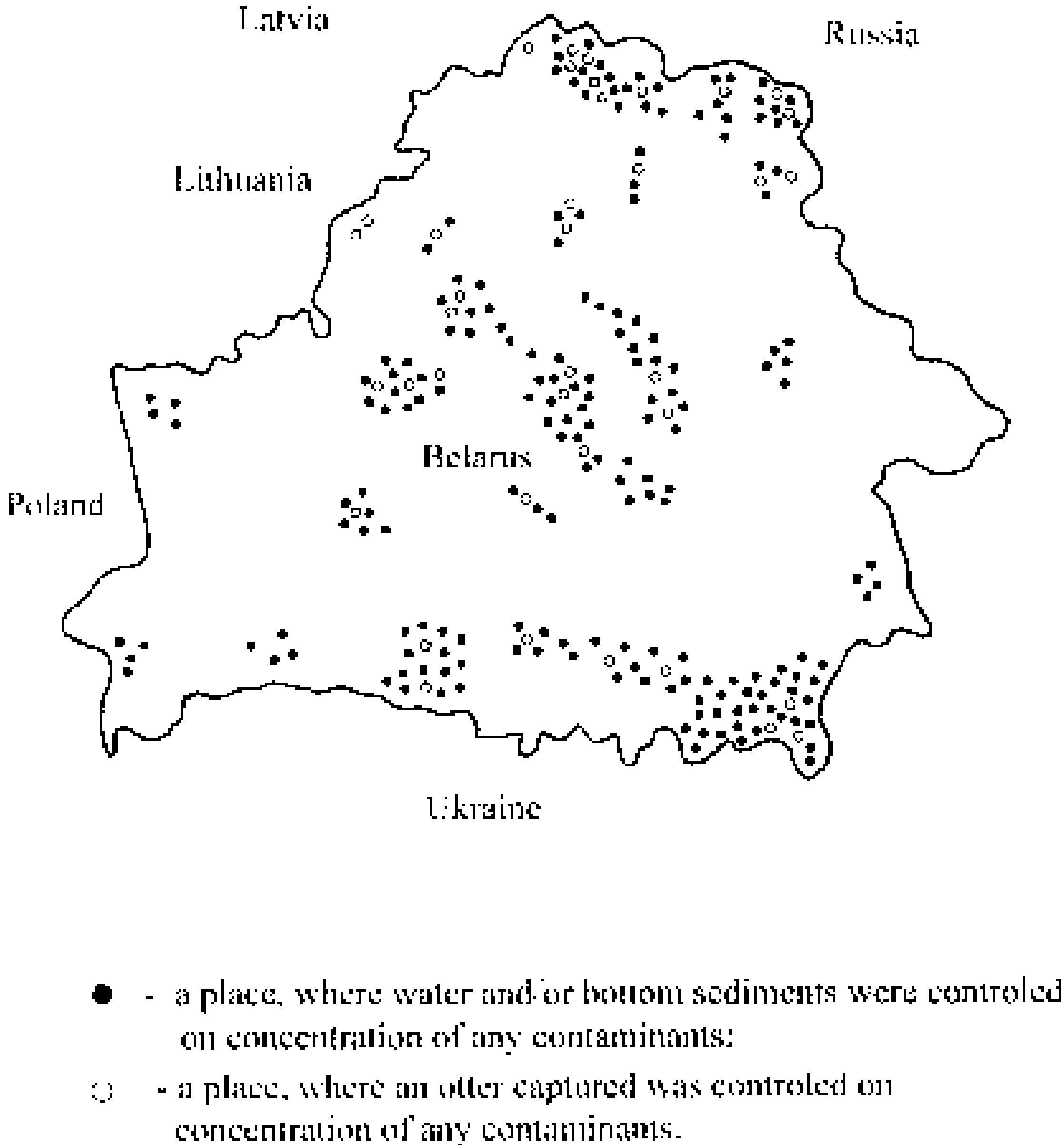

Figure 2. **Sites where otters, water and sediment samples were collected for pollutant analysis in Belarus between 1997 and 2000.**

Determination of organochlorine pesticides and PCBs were carried out by the method of gas-liquid chromatography (ROVINSKY *et al.,* 1990). The detection limit of DDE was 0.2µg/kg wet weight, DDD - 0.3, DDT - 0.4, α-BHC and γ-BHC - 0.1µg/kg: wet weight, PCBs in the water - 0.04 µg/l and bottom sediments - 0.01mg/kg: wet weight. The methods used to measure Cs134 and Cs137were described in detail by SUSCHENYA *et al.* (1995). To determine trace element contents, the samples were dried to constant weight at a temperature of 105°C. and then burnt in a muffle furnace at a temperature of 450°C, using the method described by NIKANOROV and ZHULIDOV (1991). The contents of Cu, Mn, Ti, Cr, Ni, Pb, Zn, Ag, Mo, Co, and V were measured in ash by means of atomic emission spectroscopy.

3 RESULTS

3.1 Otter distribution and population density

Data on otter presence gathered in 1997-2000 from 243 places located in river valleys of different part of Belarus and several neighbouring areas (Figure 1) suggest that otters seem to be largely distributed in the country and surrounding regions. During this otter survey, tracks and/or spraints were found at 229 places (94.2% of the total places investigated). Only in the Minsk urban area (mainly in its south-eastern part at the Svisloch River draining the city) was the otter considered a rare species, there no tracks were recorded in 14 out of the 17 sites inspected (73.7%).

The pronounced habitat-related difference in density of otters that was found, had been studied in detail before (SIDOROVICH, 1992, 1997), and, taking into account the aim of this paper, I will only pay some attention to that. According to the results, in Belarus, in both protected and hunting areas, otter density increased depending on the following favourable factors: natural riverbed, forested bank-side, bigger size of stream, faster flowing rate, intensive beaver construction activity, and abundance of meanders and old riverbeds.

Multi-annual (1984-2000) dynamics of otter population density in the four rivers chosen as model aquatic ecosystems for a long-term monitoring of the otter population in both protected and hunting areas are given in Figures 3 and 4. The density of otters was lower in the hunting areas: on average 7.2 versus 3.7 individuals per 10km of river stretch in medium-sized rivers ($t = 11.2$: $p < 0.001$) and 3.2 versus 1.8 individuals/10km of river stretch in small rivers ($t = 10.1$: $p < 0.001$). The density variation was substantially higher in the exploited otter population than in its protected population (22.2-32.4% versus 8.3-12.5%). These results and many other data obtained in 1984-1996 (SIDOROVICH, 1992, 1997) revealed a marked difference in the mean otter density and its year-to-year variation between protected and hunting areas. The main cause probably was an overexploitation i.e. quite often a year's hunting bag exceeds the yearly production of young. The otter is a protected species in Belarus, and individuals are usually killed by poachers and beaver trappers. Nevertheless, kills of otters in Belarus are still common. Moreover, in a part of the hunting areas, the abnormal population structure (i.e. low density and a predomination of females) is aggravated by the excessive selection for hunting males which directly, and possibly indirectly, negatively affected reproduction (SIDOROVICH, 1997). The direct impact appeared, because otter mothers were killed by hunters. The indirect influence may be a difficulty in mating in conditions of the altered population structure. The lowest otter density was registered in 1994-1996, when hunting of otters was particularly intensive. Then fur wearing declined in Belarus, and the otter population began to grow in number (Figures 3 and 4).

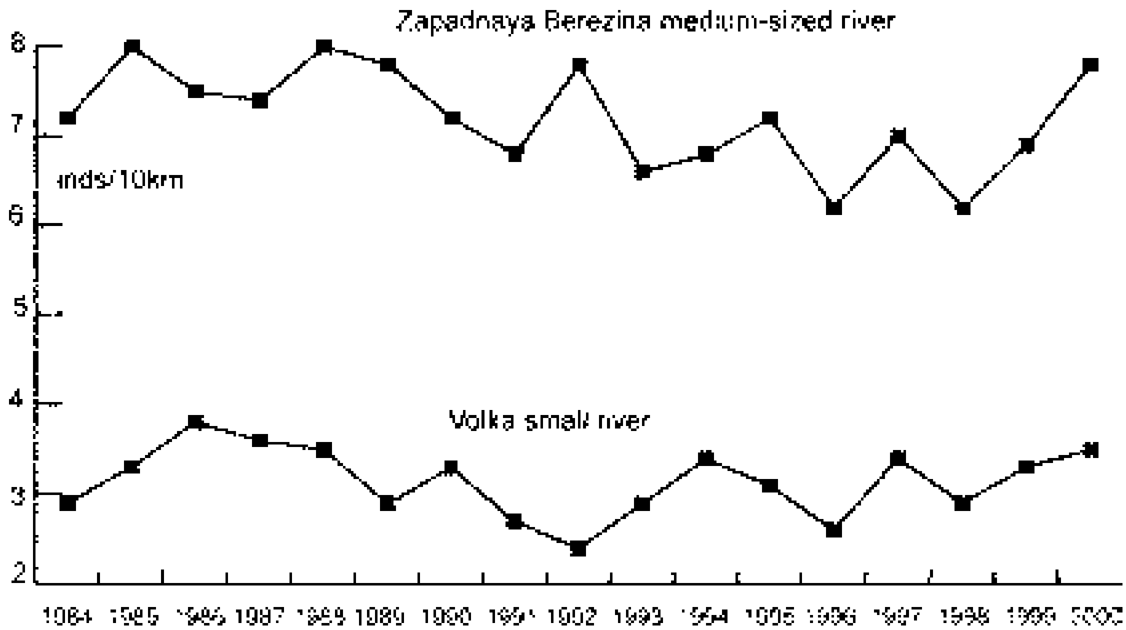

Figure 3. The estimated number of otters per 10 kilometres of river bank in protected areas of Belarus.

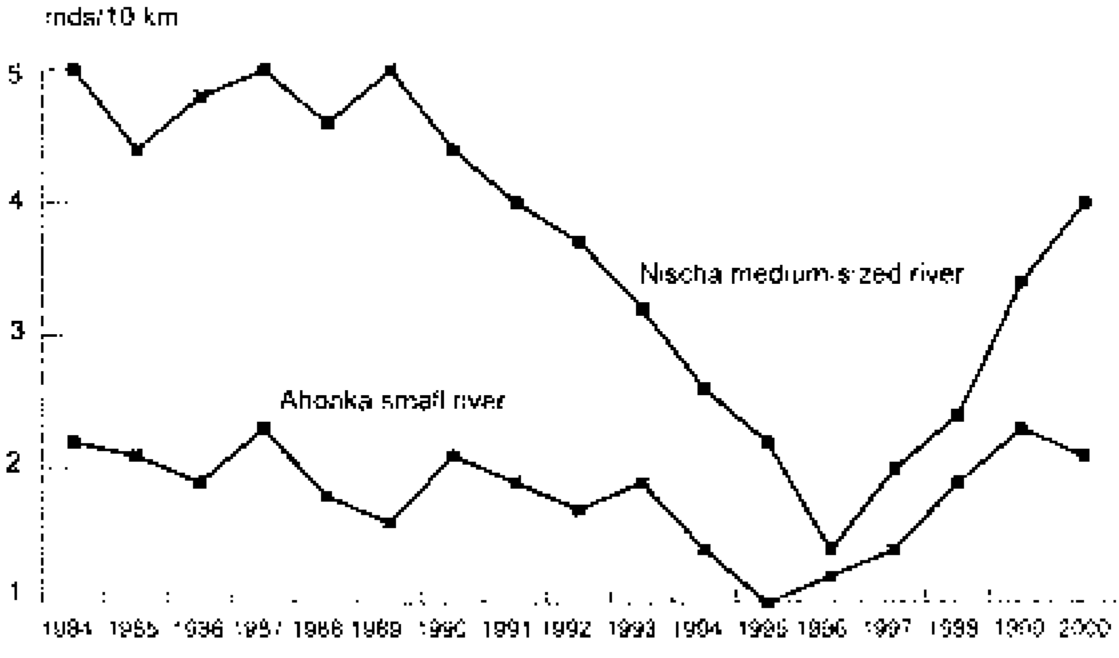

Figure 4. The estimated number of otters per 10 kilometres of river bank in areas of Belarus where hunting is permitted.

3.2 Pollution of aquatic ecosystems compared to the otter distribution

Acid deposition remains an unimportant pollution in Belarus. Only a small number of glacial lakes is characterised by naturally somewhat acidic water (JAKUSHKO *et al.*, 1988).

According to the published data by SIDOROVICH, SAVCHENKOV and DENISOVA (1996), bottom sediments of the majority of the studied rivers (51 out of 58, 87.9%) located in different parts of Belarus contained natural values of trace element concentrations (5-28mg of Pb per kg dry weight, 5-45 of Cr, 2-33 of Cu, 3-53 of V, 4-27 of Ni, 120-4300 of Mn, 190-860 of Ba). The same levels of the trace element concentration occurred in the Berezina River and its tributaries draining the undisturbed forested area in the central part of Belarus (SAVCHENKO, 1992*b*) as well as in the Nischa and Drissa Rivers located in the large forest of central North Belarus (KUZNETSOV and DOVNAR, 1984; SAVCHENKO and SIDOROVICH, 1994) and the Lovat River draining its north-eastern part (SIDOROVICH, 1997). All these rivers were fairly densely populated by otters (SIDOROVICH, 1992, 1997). Some contamination by heavy metals was registered on seven rivers draining cities (SAVCHENKO, 1992*a*; SIDOROVICH, SAVCHENKOV and DENISOVA, 1996). In particular, the Svisloch River draining the City of Minsk was highly polluted by the town's sewage. The main heavy metal pollutants were Cr, Cu, Ni, Zn, Ag, Mo, found both in solid and liquid states in the river. The concentrations of heavy metal in water and bottom sediments were 2-160 times higher than the natural values. The mean concentration of zinc in shallow silts was 3500 mg/kg dry weight, Cr - 2400, Cu - 1400, Ni - 360, Sn - 170, Ag - 13, Cd - 24, Pb - 73, V - 100, Ba - 1200, Mn - 1300, Mo - 25, As - 6, W - 16, Sb - 6, Au - 2, Se - 0.034, La - 49, and Nb - 13. The Svisloch River ecosystem was also polluted by organochlorine pesticides. In the water of this river the concentration of DDE varied up to 0.02 μg/l, DDD - up to 0.89, DDT - 0.055-0.087, α-BHC - 0.004-0.023; γ-BHC - 0.007-0.033 μg/l, while in the water of the five other rivers already mentioned - Lovat, Drissa, Nischa, Volka and Zapadnaya Berezina Rivers, these contaminants were not detected. Nevertheless, the sample size (i.e. number of rivers controlled) is small, and therefore it is hard to say precisely, that aquatic ecosystems in Belarus are only slightly polluted with organochlorine pesticides.

Long-term studies on otter distribution in the Minsk urban area drained by the strongly polluted Svisloch River revealed that normally otters were rarely observed on the first 90km of the river downstream of the city. The otter density over the next 70km of this polluted river varied between 0.5 and 2.0, and on average was about one individual per 10km of river stretch. The stream and bank structure is similar in those parts of the Svisloch River, and this polluted river and many other medium-sized rivers densely populated by otters are also characterised by similar habitat structure. These allow the conclusion, that the rare presence of otters in the first 90km of the contaminated Svisloch River downstream of the City of Minsk appeared because possibly otters were severely affected by the pollution. The adult female otter caught in this polluted habitat was characterised by several morpho-physiological peculiarities. The uterus had a large cyst (4 x 1.5 x 0.8cm) with a long stalk (about 6cm). The liver weight was abnormally high. The ratio of liver weight to body weight (multiplied by 1000) of this female was 67.9, whereas normally in Belarus this index for an adult female otter (n = 33) ranged from 30.2 to 60.7, and on average was 46.5. The kidney weight was also fairly high. The ratio of kidney weight to body weight (multiplied by 1000) of this female was 15.9, whereas normally in Belarus this index for an adult female otter (n = 33) ranged from 6.9 to 17.2, and on average was 10.0.

PCBs were not found in both water and bottom sediments of seven rivers located in central and Northern Belarus including the most contaminated Svisloch River. The detection limit of the method used was 0.04μg/l for water and 0.01mg/kg of wet weight for bottom sediments. Nevertheless, the seven rivers sampled is too small a sample size, and it is too early to summarise, that Belarus is an area relatively unpolluted by PCBs. However, the rather healthy population of otters and the data on PCB concentrations in the rivers investigated suggest this conclusion.

The radioactivity level of 80 out of 83 (96.4%) river ecosystems sampled was close to the natural value, 8-12μR/h for river watercourse, 9-34μR/h for banks and 9-68μ R/h for the floodplains. In 1990-1994 in the area of Chernobyl strongly contaminated by radionuclides, three rivers were investigated. There, levels ranged from 9 to 20μ R/h for the watercourses, from 33 to 42μR/h (average 11.98) on the banks (to 2m from a stream edge), from 26 to 486μR/h (average 51-89μR/h) on banks (from 2 to 10m); and from 29 to 67μR/h (average 44μR/h) in beaver burrows used by otters. In the floodplains, the radioactivity level was markedly higher and varied from 24 to 6231μR/h. Nevertheless, the otter density at the Rivers (Pripyat, Slovechna and Zhelon) in this area polluted by radionucludes reached 5.0, and on average was 2.5 individuals per 10km of river. These values were typical for the otter population in areas of Belarus, which were slightly contaminated by radionuclides (SIDOROVICH, 1997). Also before the Chernobyl radionuclide fallout, the density of otters on the similarly sized rivers in the Pripjat Catchment ranged between 0.7 and 5.4 individuals/10km (SIDOROVICH, 1988). Overall, no negative correlation between the radioactivity level and density of otters was found.

3.3 Contents of pollutants in otters

Out of the 11 heavy metals investigated, only levels of vanadium and cobalt in otters examined (n = 9) were below the limits of detection (respectively 1 and 3mg/kg of ash). Vanadium was measured in muscle and kidneys of only a single otter (0.16-0.35mg/kg: dry weight). Other heavy metal concentrations in muscle of otters (used as a basic tissue) may be arranged in the following decreasing order (SIDOROVICH, SAVCHENKO and DENISOVA, 1996).

Zn -- Cu -- Mn -- Ti -- Mo -- Cr -- Ni -- Pb -- Ag
10^{-2} ------- 10^{-3} ------- 10^{-4} ------------------- 10^{-5} % dry weight.

A similar order might be formed in respect of otter liver and kidney. On average, the otter muscle contained 70 mg of Zn per kg of dry weight, 24 of Cu, 3.3 of Mn, 1.6 of Ti, 0.22 of Ag, 0.45 of Cr, 0.27 of Ni and 0.29 of Pb. The mean concentrations of the trace element in the otter livers and kidneys examined were as follows: 95 and 49 mg of Zn per kg dry weight, Cu - 32 and 25, Mo - 0.82 and 0.27, Mn - 13 and 4.5, Ti - 0.36 and 0.33, Ag - 0.84 and 0.022, respectively.

The levels of PCBs in all otters investigated (liver, fat and muscle from 16 individuals) was less than 0.01mg/kg wet weight (detection limit of the method used). The mean concentrations of organochlorine pesticides in the livers of the four otters studied was as follows: DDE - 22μg/kg: wet weight, DDD - 14, DDT - 14, α -BHC - 7.2, γ-BHC - 13.

In Belarus, the sampling of radioactivity in otter tissues had not been carried out before the Chernobyl accident in April 1986. Testes from five males and embryos from four pregnant females collected in 1983-1985 have only recently been measured for radioactive caesium. In the testes the concentrations of Cs^{137}ranged from 129 to 409Bq/kg (average 287) and in the embryos from 124 to 208Bq/kg (mean 178). After the Chernobyl radionuclide fallout, radioactive caesium in both otter testes (n = 14) and embryos (n = 4: pregnant females) was in markedly higher concentrations. In the areas where the radioactivity level was less than 20μR/h, the mean concentration of

^{137}Cs was 381Bq/kg (range 271-586) in the testes and 384Bq/kg (284-501) in the embryos; the mean concentrations of Cs^{134} was142Bq/kg (range 110-172) in the testes and 136Bq/kg (126-144) in the embryos examined. The increase of Cs^{137} concentrations in otter embryos is statistically significant (t = 3.61: p = 0.02). Substantially higher levels of radioactive caesium were found in the areas highly contaminated by radionuclides. In testes (n = 4) concentrations of Cs^{137} varied from 478 to 995Bq/kg (average 679Bq/kg) and in the single embryo examined it was 731Bq/kg. Concentrations of Cs^{134} were 130-276Bq/kg (average 194) in the testes and 232Bq/kg in the embryo. The differences between non-contaminated areas after the Chernobyl accident and before, compared with the areas highly polluted by radionuclide were statistically significant (t = 2.25-3.32: p = 0.01-0.05).

Also, various tissues and organs from a further six otters (three from the Chernobyl area and three otters from areas relatively non-contaminated by radionuclides) were measured for concentration of radioactive caesium. The content of Cs^{137} in otters that inhabited unpolluted rivers varied from 154 to 569Bq/kg. In contrast, the otters caught in the Chernobyl area contained 1.5-4.6 times higher concentrations of Cs^{137} (429-1122Bq/kg). The concentrations of Cs^{134} measured in otters caught in the Chernobyl area varied between 161 and 681Bq/kg. The concentration of Cs^{137} in otter faeces collected in the Chernobyl area varied from 324 to 6783 (average 2016Bq/kg) (n = 19). In rivers of Northern Belarus largely unpolluted by radioactivity, the concentration of Cs^{137} in the otter faeces examined (n = 10) varied from 84 to 201 (average 140)Bq/kg.

4 CONCLUSION

These data on concentrations of heavy metals and organochlorine compounds in aquatic ecosystems and otters, as well as information relating to acid deposition in otter habitats in different parts of Belarus are compared with levels in highly polluted regions of Western Europe (FORSTNER and WITTMANN, 1979; DISSANAYOKE, TOBSCHALL and PALME, 1983; MOORE and RAMAMOORTHY, 1984; WHITTON, 1984; HERNANDEZ, GONZALEZ and RICO, 1985; MASON and O'SULLIVAN, 1993; SMIT *et al*., 1994 and references therein). The comparison suggests that otter habitats in Belarus are relatively unpolluted by the above mentioned contaminants. Plausibly, this still allows for a healthy population of otters there. Only in the restricted urban area of the City of Minsk does the combined contamination by heavy metals and organochlorine pesticides seem to lead to the local otter decline recorded. Concerning radioactivity pollution, this seems not to have affected otters enough to be the cause of population density decrease, even in the Chernobyl area.

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