ELSEVIER

0269-7491(95)00099-2

Environmental Pollution, Vol. 92, No. 2, pp. 165–171, 1996

0269-7491/96 $15.00+0.00

CONCENTRATIONS OF SOME ORGANOCHLORINES IN OTTERS (*LUTRA LUTRA* L.) IN SCOTLAND: IMPLICATIONS FOR POPULATIONS

H. Kruuk & J. W. H. Conroy

Institute of Terrestrial Ecology, Banchory AB31 4BY, UK

(Received 5 July 1995; accepted 12 October 1995)

Abstract

*The concentrations of polychlorinated biphenyls (PCBs) and residues of organochlorine pesticides (HEOD and DDE) were measured in livers of otters (*Lutra lutra*) from different areas in Scotland. Whilst HEOD and DDE occurred at very low concentrations only, PCBs were present in high levels in some areas, the highest in Shetland (geom mean 2.05 ppm wet wt), related probably to high levels in sediments in the north-east Atlantic. PCBs were a mixture of congeners in which higher-chlorinated ones predominated, especially 138, 153, 170 and 180, a pattern comparable to that observed in otters from continental Europe. Individual values of total PCB reached levels of over 14 ppm wet weight, even in otters in good condition in thriving populations.*

Otter population densities were known in some of the study areas; in Shetland numbers increased over the study period, and the density was relatively high also elsewhere in Scotland. There was a strong negative correlation between PCB and body condition, but no correlation between PCB concentration and age of otters (mean age = 4.1 years), which suggests that PCBs do not accumulate substantially in otters in the long term. The observations cast doubt on the significance of published 'critical levels' of PCBs to otter populations, based on data obtained from captive mink. Copyright © 1996 Elsevier Science Ltd.

INTRODUCTION

In this contribution we describe the concentration of some potentially important pollutants in the tissues of Eurasian otters *Lutra lutra* from Scotland, in relation to the animals' age, condition and area of occurrence, during 1987–92. Some of the populations were of known density. The otters were analysed for concentrations of *p,p'*-DDE (the main metabolite of DDT), HEOD (ingredient of dieldrin), and several congeners of PCBs (polychlorinated biphenyls). The main purpose of the analysis was to assess whether otter populations are at present likely to be affected by these contaminants, and how the potential for such effects is distributed within Scotland. The data also provide a baseline for future observations and for comparisons with otters elsewhere.

Otter populations in Britain have declined sharply in the last few decades, probably due to the effects of contamination by organochlorines (Mason & Macdonald, 1986; Jefferies, 1989). A similar decline took place in many other areas in Europe, and in several countries the species is now extinct (Foster-Turley *et al.*, 1990). The exact causal agent is still uncertain (Mason, 1989), but the various compounds which have been suggested (e.g. dieldrin, DDT, PCBs) are now all, or almost all, officially out of use (Newton *et al.*, 1993). As a result, in parts of England, Wales and Central Scotland there is evidence that otters are returning in areas from where they had disappeared, expanding from surviving populations elsewhere (Strachan *et al.*, 1990; Green & Green, 1984; Andrews & Crawford, 1986; Andrews *et al.*, 1992). Even if this recovery is sustained, however, it is slow and is taking considerably longer than that of other species affected by pollutants, particularly birds of prey (Newton, 1979; Newton & Wyllie, 1992).

During the period of the population crash in Britain, otters remained common in most of Scotland (Green & Green, 1980), and questions arise about current levels of contaminants in areas where populations appear to be doing well, or populations in areas bordering regions where declines occurred. Furthermore, no account has been given of the accumulation of contaminants with age, in individual otters; it is expected that unless animals are able to excrete or metabolize the various compounds involved, these would accumulate with age. Otters can be aged accurately (Heggberget, 1984), and the availability of tissues from numbers of dead otters provided the opportunity to study the rate at which contaminants accumulate in individuals, by comparing their presence in animals of different age (Kruuk & Conroy, 1991). Similarly, a method to assess relative body condition is available (Kruuk *et al.*, 1987; Kruuk & Conroy, 1991), and this provides further important background information for the comparisons of concentrations of the different compounds in this obviously sensitive species (Mason, 1989).

One problem which has plagued earlier research on pollution in otters is the lack of information on population parameters: whether densities are high or low, whether numbers are stable, decreasing or increasing.

This is now being redressed in at least some areas (Kruuk *et al.*, 1989, 1993; Conroy & Kruuk, 1995). It has been argued convincingly that the effects of pollution on a species should be studied in the context of populations, not just as effects on individuals (Newton, 1988); populations may be seriously affected at relatively low mean tissue concentrations in individuals.

In this paper we address the following hypotheses:

(1) In otter populations at carrying capacity (or known 'thriving' populations) pollutant levels are generally well below levels demonstrated to cause mortality or reproductive failure.
(2) Concentrations of organochlorine contaminants increase with age of the otter.
(3) Body condition of otters is lower in animals with high burdens of contaminants.
(4) Concentrations of contaminants in otters are higher in areas known or suspected to be affected by specific pollutants, and lower in areas far from potential sources of contamination.

MATERIALS AND METHODS

Carcasses of otters ($N=116$) were collected between June 1987 and July 1992. The large majority of the animals had been killed by traffic, they were deep frozen, sent to the Institute of Terrestrial Ecology (ITE) laboratory in Banchory where they were measured, dissected and autopsied.

The carcasses originated from different areas in Scotland (Fig. 1). Areas from which no carcasses were received were those where the species is

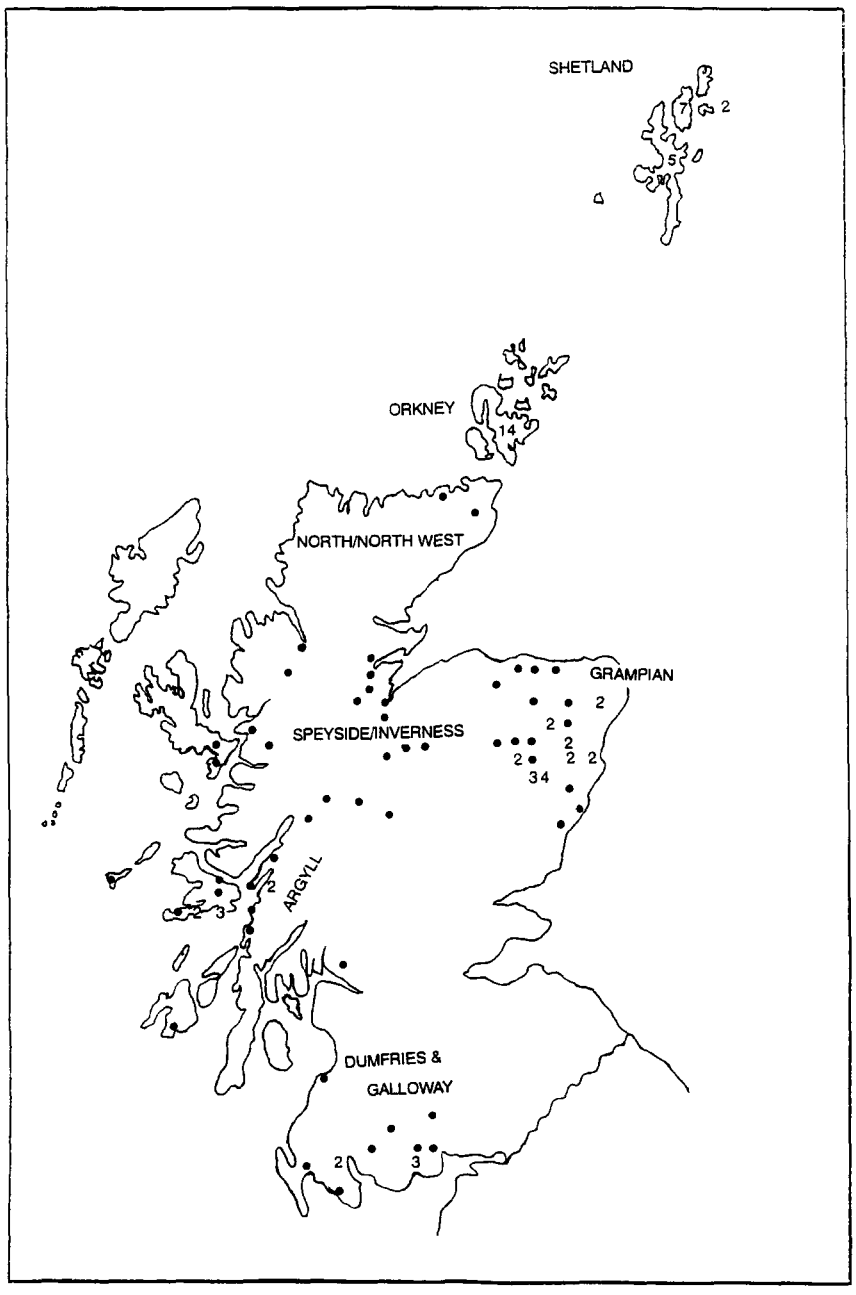

Fig. 1. Areas of origin of otter samples from Scotland.

Table 1. Summary of values of the main variables analysed in 116 carcasses

	Mean	S.E.	Min.	Max.
Age	4.07	3.82–4.33	0.5	15.0
Condition	1.01	0.99–1.04	0.48	1.43
PCB (w)	0.797	0.712–0.892	0	14.404
PCB (l)	27.402	24.288–30.915	0	595.923
DDE (w)	0.115	0.102–0.130	0	2.806
DDE (l)	3.988	3.523–4.511	0	116.776
HEOD (w)	0.084	0.080–0.089	0.024	0.277
HEOD (l)	2.954	2.790–3.128	0.672	14.961

For the chemical analyses 'mean' and 'S.E.' are geometric mean ± one geometric standard error.
PCB (w) = total PCB (concentration in wet liver tissue), PCB (l) = total PCB (in lipids in liver), etc.

uncommon, except for some islands (Outer Hebrides) where otters are plentiful but we were unable to obtain specimens.

The age of each otter was estimated by counting incremental growth rings in the dentine of a canine or incisor tooth [details in Heggberget (1984)]. The body condition index K was calculated as $K = W/a \cdot L^n$, where W = body mass in kg, L = total length (straight line from nose to tip of tail) in m, $a = 5.02$ and $n = 2.33$ for females, $a = 5.87$ and $n = 2.39$ for males (Kruuk *et al.*, 1987).

Samples of liver of each otter were sent to the ITE laboratory at Monks Wood, where they were analysed for the presence of organochlorines using procedures and safeguards detailed by Johnstone *et al.* (1991), Newton *et al.* (1993) and Simmonds *et al.* (1994). Analysis was carried out on a Varian 3400 GC with electron capture detection using a 30-m DB210 capillary column, at a temperature of 190°C. Identification and quantitation were based on comparisons with a standard pesticide mixture, and with a PCB standard of Aroclor 1254. As quality assurance, analyte recovery was tested with standardized procedures (spiked samples); recovery rates were over 90% in all cases.

Results were expressed in terms of $\mu g\ g^{-1}$ (ppm) wet wt following the arguments by Newton *et al.* (1993) and Smit *et al.* (1994). Additional values in ppm lipid weight are given in Table 1, for comparison with data from some previous authors. There was a linear relationship with a high correlation between our measurements expressed as ppm wet weight and lipid weight [ppm $PCB_{tot(lipid)} = 43.8 * \text{ppm } PCB_{tot(wet)} - 8.7$; $r^2 = 0.94$].

The otters were analysed for concentrations of *p,p'*–DDE, HEOD, for the total concentration of all PCBs (PCB_{tot}), and separately for the following PCB congeners: Nos 8, 18, 28, 31, 52, 77, 101, 118, 126, 128, 138, 149, 153, 169, 170 and 180.

Results of chemical analyses are expressed as geometric means rather than arithmetic means, following arguments by Newton (1988), Newton *et al.* (1993) and Smit *et al.* (1994). This removes disproportionate effects of outlying values, and gives a value close to that of the median but with greater possibilities for statistical evaluation.

RESULTS

Age and condition

The overall mean age at death of the otters was 4.07 years (±2.71 S.D., $n = 113$; Table 1). The mean age at death of females was 3.76 years (±2.62 S.D.; $n = 51$), of males 4.32 years (±2.78; $n = 62$); the difference was statistically not significant (Kruskal–Wallis $\chi^2 = 1.38$, d.f. = 1, $p < 0.24$).

To allow for possible differences in pollutant burden related to age structure of otter populations in different areas, otters were grouped into eight areas, and mean age per area was calculated. In the seven areas where sample size was ten or more, mean age varied between 3.1 (NW Scotland) and 5.3 years (Argyll), but the within-sample variation was large and differences were statistically not significant (Kruskal–Wallis $\chi^2 = 9.97$, d.f. = 6, $p < 0.13$). There did not appear to be an overall pattern in otter age, e.g. related to latitude or habitat (fresh water or coastal), and for further analysis any possible differences in age between areas were ignored.

The body condition index K of individual otter carcasses in our sample varied between 0.48 and 1.43. Mean K was the same for males and females (for males $K = 1.02 \pm 0.22$ S.D., $n = 57$; for females $K = 1.01 \pm 0.20$ S.D., $n = 48$). There were substantial differences in the condition of otters from different areas, with K varying from a mean of 0.85 in south-west Scotland to 1.18 in Orkney; other areas were intermediate. This area effect was statistically significant (Kruskal–Wallis $\chi^2 = 21.21$, d.f. = 6, $p < 0.002$), but there were no obvious overall geographical, habitat-linked or latitudinal trends.

Contamination in otters overall, and in different areas

Table 1 summarizes the presence of the analysed organochlorines in the otter livers as geometrical means ±1 S.E., and individual minimum and maximum values.

DDE was found only in small concentrations (mean 0.12, maximum 2.81 ppm), with values much lower than those found by Mason in the early 1980s throughout Britain (Mason, 1989), and considerably lower also than those found in herons, kingfishers and raptors (Newton *et al.*, 1993). Our results were similar to those obtained by Skaren (1988) for otters from central Finland.

HEOD concentrations were also low (mean 0.08 ppm, and maximum value of 0.28 ppm), compared with previously published early values for Britain of 0.5 ppm (arithmetic mean) and 13.9 ppm maximum (Jefferies *et al.*, 1974). In lipid weight our mean and maximum values were 2.8 and 15.0 ppm, compared with previously published arithmetic mean values for Britain in the 1980s of 5.3 and 66.4 ppm (Mason, 1989). As with DDE, the concentrations of HEOD in otters were far lower than those in various fish-eating birds and raptors (Newton *et al.*, 1993). There was just the occasional high value in our data.

There was a close and highly significant correlation between the concentrations of DDE and HEOD in our otter tissues ($r = 0.59$). There were no significant correlations between PCBs and DDE or HEOD (all samples for all areas), nor was any of these compounds correlated with mercury (Kruuk & Conroy, unpublished data).

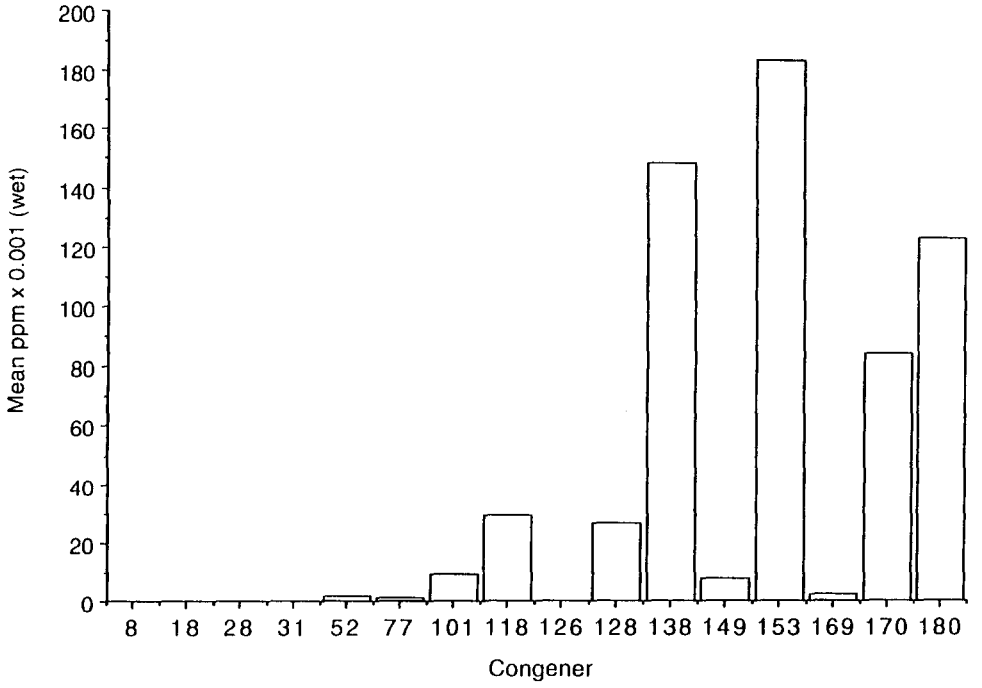

Fig. 2. Profile of PCB congeners: mean of all samples.

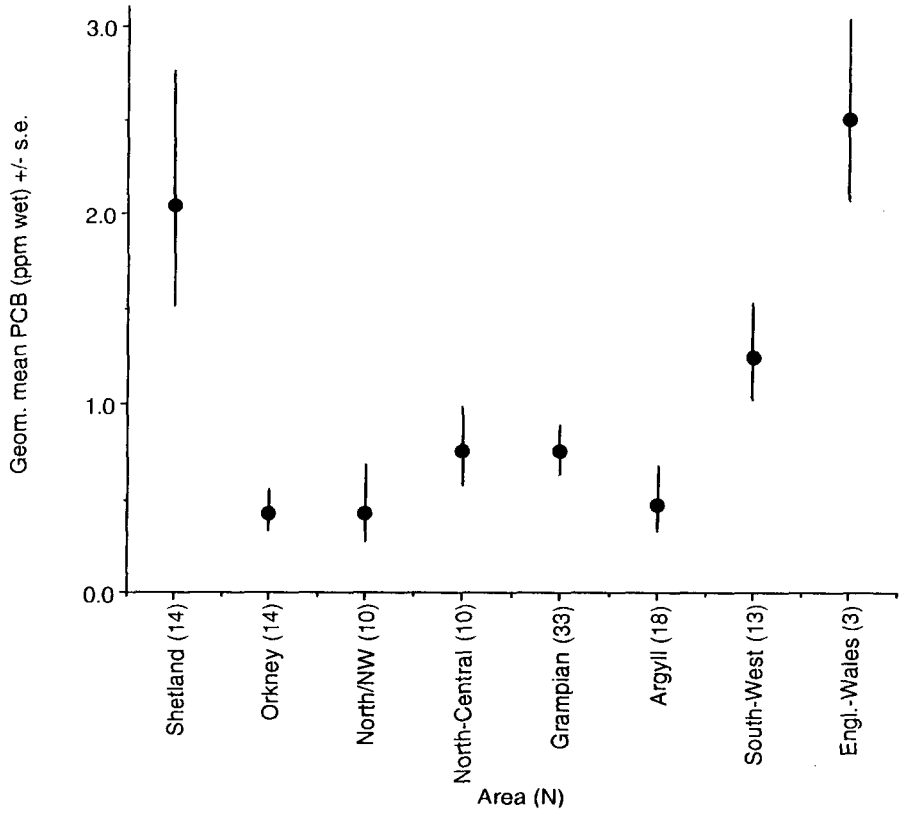

Fig. 3. Total PCBs in otter livers from different areas, mostly in Scotland.

The most prominent congeners amongst the PCBs tested were the higher chlorinated compounds 138, 153, 170 and 180 (Fig. 2); together they constituted 37.7% of the total PCB load of otters (PCB_{tot}). This congener-profile was rather similar to that in results from the Netherlands and Germany (Broekhuizen, 1989; Smit *et al.*, 1994), and Denmark, SW England and Ireland (Mason & Ratford, 1994).

The occurrence of the most common PCB congeners in our samples was strongly inter-correlated. There were also highly significant correlations between the measurement of all PCBs together (PCB_{tot}) and of the congeners 118 ($r=0.81$), 128 ($r=0.72$), 138 ($r=0.92$), 153 ($r=0.97$), 170 ($r=0.94$) and 180 ($r=0.96$). For further analyses and discussion, therefore, we will refer to the total PCB concentration (PCB_{tot}, wet wt).

The body condition K was highly significantly correlated with the concentrations of PCBs in the liver: $r=-0.43$ for PCB_{tot} (wet wt; $n=105$, $p<0.0001$) or $r=-0.45$ (lipid weight; $n=105$, $p<0.0001$). Correlations between K and the other compounds were non-significant ($r=-0.13$ and $r=0.03$ for ppm DDE and HEOD, respectively).

The data were analysed for correlations between age of the otters and concentrations of the various compounds, first jointly for all areas. There were no significant overall correlations between age and DDE ($r=-0.06$), HEOD ($r=-0.07$), and PCB_{tot} ($r=0.11$).

However, the PCB concentration varied significantly between areas (Fig. 3; Kruskal–Wallis $\chi^2=29.39$, d.f.$=7$, $p<0.001$), with the highest Scottish values found in Shetland, and only the few samples from south of the Scottish border were higher still. However, there was no obvious general geographical or habitat trend.

Some of the geometric mean values for PCB concentrations in individual areas were relatively high (Mason, 1989), especially in the oft-quoted comparison with the concentration causing reproductive failure in mink *Mustela vison* [50 ppm lipid, or 1.24 ppm wet wt; Jensen *et al.* (1977)]. The regional geometric mean for Shetland was well above this (2.05 ppm), and for the south-west of Scotland it was just above, at 1.25 ppm. Individual values, often in otters in excellent body condition, were much more excessive still. For instance, one lactating female shot in 1987 near Aboyne (Grampian; not included in this sample) contained 25.24 ppm (wet, or 1096.80 ppm lipid) in the liver, and the highest value in the present sample was 14.40 ppm (596 ppm lipid), from Shetland. Of 15 other, lactating females in the present sample the most contaminated contained 1.20, 1.20, 0.87 and 0.39 ppm.

The geometric mean PCB concentration in males from all areas was 0.98 ppm ($n=60$), in females 0.59 ($n=51$), but although large the difference was statistically not significant, also when analysed separately by area.

Within areas the relative variance in the concentrations of PCBs was high, though in general unrelated to the age of otters. In only one area (south-west Scotland) a significant correlation between age and PCBs was found ($r=0.61$, $p<0.05$). Taken separately by sex, for females there was a non-significant negative correlation between PCBs and age in three of the seven areas, a non-significant positive correlation in the rest; for males all areas showed a positive correlation, which was significant in two (at $p<0.02$ and $p<0.01$). The large variation between individual otters within areas has been observed elsewhere (Smit *et al.*, 1994), and suggests large individual differences in uptake, or rate of excretion or metabolizing of the compounds.

DISCUSSION

Newton (1988), in a seminal paper on effects of pollution in birds of prey, argues that the (geometric) mean level of concentration of contaminants in a population of animals which is required to cause a population decline, the LPD, may bear no relationship to LD_{50} levels. The reason for this is compensatory mortality or reproduction; in consequence, some populations may suffer high mortality through pollution but still maintain their density (possibly limited by availability of some resource). Other populations may decline at even very low levels of contamination, because of the loss of the few individuals with high levels of pollutants. Newton (1988) argues that, in general, patterns of mortality and reproduction do not vary sufficiently within species to have a large effect on differences in LPD between areas, for any one polluting compound. There are, of course, large differences between species.

This argument is relevant when comparing the effects of pollutants on different species of mustelid, and especially when comparing laboratory studies with wild populations. Thus, extrapolations from levels of PCBs causing reproductive failure in laboratory mink [50 ppm lipid, or about 1.24 ppm wet wt in liver tissue; Jensen *et al.* (1977)] to wild otter populations (Mason, 1989) are speculative, and may be counter-productive in terms of conservation management. These extrapolations are the basis for the hypothesis that 50 ppm (lipid) PCBs in otter populations is 'critical' for their survival (Mason *et al.*, 1992). However, even if individual otters would be equally sensitive to the effects of PCBs as are mink, nevertheless for some otter *populations* a critical level may be lower or much higher. In fact, mink are known to be particularly sensitive to PCBs (Aulerich & Ringer, 1977; Bleavins *et al.*, 1980; Aulerich *et al.*, 1985).

The need for caution in generalizations to otter populations from studies on mink in the laboratory is underlined by results reported here. The population of otters on Shetland is relatively high and dense, much more so than that in neighbouring Orkney (Kruuk, 1995). During the present study the Shetland population increased significantly from an estimated 700–900 adults to 800–1000 (Kruuk *et al.*, 1989; Conroy & Kruuk, 1995). Thus, it can be described as thriving, possibly limited by habitat and food availability (Kruuk & Moorhouse, 1990; Kruuk *et al.*, 1989). The mean level of contamination with DDE and HEOD was low in the

Shetland otters, as it was everywhere else in Scotland, but mean PCB concentration in Shetland was much higher than in Orkney and elsewhere in Scotland, much higher also than the level causing reproductive failure in laboratory mink.

Similarly, otter populations in freshwater in Grampian have been studied in detail during the time samples were collected for the present study. The animals were present in relatively large numbers (Kruuk *et al.*, 1993), with casual observations suggesting that there were no substantial long-term changes in density over many years. PCB levels there are such that they would be called 'of concern' when compared with the mink studies (Smit *et al.*, 1994), with some very high values, though clearly this had little, if any, effect on density. Similar results were obtained in Spain (Ruiz-Olmo *et al.*, in press).

The results confirm the suggestion that otter populations are able to do well despite high rates of PCB contamination, if the necessary resources are available. We reject our first hypothesis (i.e. that thriving populations have pollution levels below the concentrations which are suggested to cause mortality or reproductive failure).

The distribution of PCB levels in Scotland showed high mean levels in the far north (Shetland), with moderately low mean levels throughout mainland Scotland but again significantly higher concentrations in the south-west, near the English border. The few samples from south of the border ($n=3$) were also very contaminated, but the number is too small to draw conclusions. There were no strikingly high levels of contamination in intensive agricultural regions such as north-east Grampian.

The presence of high levels of PCBs in Shetland otters, which feed in the surrounding seas, is consistent with the observations of Simmonds *et al.* (1994). These authors found very high levels of PCBs in pilot whales around Faroe and Shetland, dangerous for human consumption. The waters of the north-east Atlantic Ocean are known as the greatest global environmental reservoir of PCBs (Larsson, 1985).

The distribution of PCBs in the otters from the rest of Scotland does not show any obvious point sources. However, the species is scarce near the main industrial areas (immediately north and south of the 'central belt'), and we had no samples from there: the otters' absence could have been pollution-related, involving point-sources.

Our data did not suggest that the compounds investigated here accumulated in otters with age, even when we analysed separately by area and sex. There was a possibility that in some areas a negative correlation occurred between age and PCB concentration in females, but none of these correlations was significant. This contrasted strongly with our findings for mercury (Kruuk & Conroy, 1991). We therefore reject our second hypothesis, that PCBs accumulate in otters with age: if such an accumulation takes place it is obscured by other variables. Otters are probably able to either metabolize or excrete various organochlorines, including at least some PCB congeners, and important concentrations are found in the faeces ('spraints': Mason *et al.*, 1992). In feeding experiments with captive otters reported by Smit *et al.* (1994) it was found that only 8.4% of ingested PCBs was excreted in spraints; it is possible, therefore, that a large proportion of PCBs is metabolized by otters, rather than stored or excreted.

Our third hypothesis suggested that animals with high levels of contaminants such as PCBs have a lower body condition. This was strongly supported by our data, with a highly significant negative correlation between PCB level and condition index K, throughout Scotland. The mechanism of this correlation is unclear, however; low body condition could be either cause or consequence of high PCB concentrations in the liver. The problem has been discussed by Smit *et al.* (1994). Mason and O'Sullivan (1992) found no correlation between body condition and PCB concentration in 33 otters from Ireland; the reason for the difference is not clear.

In conclusion, our data suggest that pollutants such as PCBs are present in otters in widely dispersed areas within Scotland, in concentrations which may be relatively high. The concentration of other pollutants (dieldrin, and DDE) appeared to be low and unimportant. Otter populations in some areas thrive despite high PCB contamination, and there is evidence that they are food limited (Kruuk & Moorhouse, 1990; Kruuk *et al.*, 1993; Kruuk, 1995), and able to survive despite high mortality (Kruuk & Conroy, 1991). Thus, when assessing the risks to populations created by pollution, other factors apart from the concentration of pollutants have to be taken into account, including the availability of resources.

ACKNOWLEDGEMENTS

We are grateful to staff of Scottish Natural Heritage, Jim and Rosemary Green and various others who helped with collection and storage of carcasses. Dr M. C. French carried out the analyses at the ITE Monks Wood laboratory. Drs C. F. Mason, P. Leonards, M. Smit and M. Olsson critically and very helpfully commented on the manuscript.

REFERENCES

Andrews, E. & Crawford, A. K. (1986). *Otter Survey of Wales 1984–85*. Vincent Wildlife Trust, London, UK.

Andrews, E., Howell, P. & Johnson, K. (1992). *Otter Survey of Wales 1991*. Vincent Wildlfe Trust, London, UK.

Aulerich, R. J. & Ringer, R. K. (1977). Current status of PCB toxicity to mink and effect on their reproduction. *Arch. Environ. Contamination Toxic.*, **6**, 279–92.

Aulerich, R. J., Bursian, S. G., Breslin, W. J., Olson, B. A. & Ringer, R. K. (1985). Toxicological manifestations of 2,4,5,2′,4′,5′-2,3,6,2′,3′,6′- and 3,4,5,3′,4′,5′-hexachlorobiphenyl and Aroclor 1254 in mink. *J. Toxic. Environ. Hlth*, **15**, 63–79.

Bleavins, M. R., Aulerich, R. J. & Ringer, R. K. (1980). Polychlorinated biphenyls (Aroclors 1016 and 1242): Effects on survival and reproduction in mink and ferrets. *Arch. Environ. Contamination Toxic.*, **9**, 627–35.

Broekhuizen, S. (1989). Belasting van otters met zware metalen en PCBs. *De Levende Natuur*, **90**, 43–7.

Conroy, J. W. H. & Kruuk, H. (1995). Changes in otter numbers on Shetland between 1988 and 1993. *Oryx*, **29**, 197–204.

Foster-Turley, P., Macdonald, S. M. & Mason, C. F. (1990). *Otters: An Action Plan for their Conservation.* I.U.C.N. Species Survival Commission, Gland.

Green, J. & Green, R. (1980). *Otter Survey Of Scotland 1977–79.* Vincent Wildlife Trust, London, UK.

Green, J. & Green, R. (1984). *Otter Survey of Scotland 1984–85.* Vincent Wildlife Trust, London, UK.

Heggberget, T. M. (1984). Age determination in the European otter (*Lutra lutra*). *Z. Saugetierkunde*, **49**, 299–305.

Jefferies, D. J. (1989). The changing otter population of Britain 1700–1989. *Biol. J. Linnean Soc.*, **38**, 61–9.

Jefferies, D. J., French, M. C. & Stebbings, R. E. (1974). Pollution and mammals. *Monks Wood Experimental Station Report for 1972–73.* Natural Environment Research Council, Huntingdon, UK, pp. 13–15.

Jensen, S., Kihlstrom, J. E., Olsson, M., Lundberg, C. & Ordberg, J. (1977). Effects of PCB and DDT on mink (*Mustela vison*) during the reproductive season. *Ambio*, **6**, 239.

Johnstone, P. A., Stringer, R. L. & French, M. C. (1991). Pollution of U.K. estuaries: historical and current problems. *Sci. Total Environ.*, **106**, 55–70.

Kruuk, H. (1995). *Wild Otters: Predation and Populations.* Oxford University Press, Oxford, UK.

Kruuk, H. & Conroy, J. W. H. (1991). Mortality of otters (*Lutra lutra*) in Shetland. *J. Appl. Ecol.*, **28**, 95–101.

Kruuk, H. & Moorhouse, A. (1990). Seasonal and spatial differences in food selection by otters (*Lutra lutra*) in Shetland. *J. Zool. London*, **221**, 621–37.

Kruuk, H., Conroy, J. W. H. & Moorhouse, A. (1987). Seasonal reproduction, mortality and food of otters *Lutra lutra* L. in Shetland. *Symp. Zool. Soc. London*, **58**, 263–78.

Kruuk, H., Moorhouse, A., Conroy, J. W. H., Durbin, L. & Frears, S. (1989). An estimate of numbers and habitat preferences of otters *Lutra lutra* in Shetland, U.K. *Biol. Conserv.*, **49**, 241–54.

Kruuk, H., Carss, D. N., Conroy, J. W. H. & Durbin, L. (1993). Otter (*Lutra lutra*) numbers and fish productivity in two areas in North East Scotland. *Symp. Zool. Soc. London*, **65**, 171–91.

Larsson, P. (1985). Contaminated sediments of lakes and oceans act as sources of chlorinated hydrocarbons for release to water and atmosphere. *Nature*, **317**, 347–9.

Mason, C. F. (1989). Water pollution and otter distribution: a review. *Lutra*, **32**, 97–131.

Mason, C. F. & Macdonald, S. M. (1986). *Otters: Ecology and Conservation.* Cambridge University Press, Cambridge, UK.

Mason, C. F. & Ratford, J. R. (1994). PCB congeneres in tissues of European otter (*Lutra lutra*). *Bull. Environ. Contamination Toxic.*, **53**, 548–54.

Mason, C. F. & O'Sullivan, W. M. (1992). Organochlorine pesticide residues and PCBs in otters (*Lutra lutra*) from Ireland. *Bull. Environ. Contamination Toxic.*, **48**, 387–93.

Mason, C. F., Macdonald, S. M., Bland, H. C. & Ratford, J. (1992). Organochlorine pesticide and PCB content in otter (*Lutra lutra*) scats from Western Scotland. *Water Air Soil Pollut.*, **64**, 617–26.

Newton, I. (1979). *Population Ecology of Raptors.* Poyser, Berkamsted.

Newton, I. (1988). Determination of critical pollutant levels in wild populations, with examples from organochlorine insecticides in birds of prey. *Environ. Pollut.*, **55**, 29–40.

Newton, I. & Wyllie, I. (1992). Recovery of a sparrowhawk population in relation to declining pesticide contamination. *J. Appl. Ecol.* **29**, 476–84.

Newton, I., Wyllie, I. & Asher, A. (1993). Long-term trends in organochlorine and mercury residues in some predatory birds in Britain. *Environ. Pollut.*, **79**, 143–51.

Ruiz-Olmo, J., Lopez-Martin, J. M. & Delibes, M. (in press). Organochlorine residue levels in otters (*Lutra lutra*) from Spain. *Habitat.*

Simmonds, M. P., Johnston, P. A., French, M. C., Reeve R. & Hutchinson, J. D. (1994). Organochlorines and mercury in pilot whale blubber consumed by Faroe islanders. *Sci. Total Environ.*, **149**, 97–111.

Skaren, U. (1988). Chlorinated hydrocarbons, PCBs and cesium isotopes in otters (*Lutra lutra* L.) from central Finland. *Ann. Zool. Fennici*, **25**, 271–6.

Smit, M. D., Leonards, P. E. G., van Hattum, B. & de Jongh, A. W. J. J. (1994). *PCBs in European Otter (Lutra lutra) Populations.* Institute for Environmental Studies, Amsterdam, The Netherlands.

Strachan, R., Birks, J. D. S., Chanin, P. R. F. & Jefferies, D. J. (1990). *Otter Survey of England 1984–1986.* Nature Conservancy Council, Peterborough, UK.