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2000/01 Annual report

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1 Preface and summary

1.1 Introduction

The Wildlife and Pollution contract covers a long-term monitoring programme that examines the levels of certain pollutants in selected wildlife species in Britain. The programme was started in the early 1960s, when there were serious concerns over the effects of organochlorine insecticides and organomercury fungicides on various species of birds and mammals. This early work demonstrated the effects of the organochlorines, and eventually contributed to the ban on their use in the UK and abroad. The programme has measured levels of these compounds in predatory and fish-eating birds since then. Investigations have also been made into the levels of industrial polychlorinated biphenyls (PCBs), following their identification as pollutants in 1966. Mercury levels, derived from both agricultural and industrial sources, have also been tracked. In recent years, investigations have been made into the effects of the latest generation of rodenticides on barn owls *Tyto alba*. Northern gannet *Morus bassanus* eggs are also collected approximately biennially from two colonies and, when available, from other sites; eggs were collected from one site in 2000.

This programme is now the longest-running of its kind anywhere in the world and the findings stimulate considerable interest internationally, as well as in Britain. Annual reports give an interim summary of results. This current report presents the results of analyses carried out on material collected in 2000. Every three years these annual results are gathered together into a more substantial report in which they are integrated with previous findings. The last report of this type covered the period up to and including 1997 (Newton *et al*. 1998) and is updated here. The present report summarises the long-term trends in all contaminants (except rodenticides, analysed in Shore *et al*. 2005) that occurred during the monitoring period up to and including the year 2000. Results are published periodically in the scientific literature, and recent key papers are listed in the references to the present report.

The Wildlife and Pollution contract was the subject of scientific assessment within JNCC's rolling programme of peer review in autumn 1993 and was further assessed in 1996. As a result of the last two assessments, some monitoring was curtailed. Specifically, common kestrels *Falco tinnunculus* were no longer monitored for organochlorines, although from 2001 this species will be monitored annually for second-generation anticoagulant rodenticides. The Centre for Ecology & Hydrology (CEH) still collects specimens for studying other contaminants as part of its core research programme. Similarly, other species (peregrine falcon *Falco peregrinus*, common buzzard *Buteo buteo*, long-eared owl *Asio otus*, little owl *Athene noctua*, common kingfisher *Alcedo atthis*, great crested grebe *Podiceps cristatus*, and great bittern *Botaurus stellaris*) that were received in small numbers in occasional years were also not analysed routinely, although some were analysed in specific one-off studies and tissues from *all* birds received at CEH in 2000 are archived in deep-freeze for future potential analyses.

Each section within the Wildlife and Pollution contract is summarised below. Each is dependent on the provision of material from amateur naturalists and other interested parties, and it is not always possible to obtain desired material for analysis, especially from remote areas.

1.2 Organochlorines and mercury in the livers of predatory birds

The main objective of this work is to analyse the bodies of certain predatory and fish-eating bird species, supplied by members of the public, in order to continue the monitoring of organochlorine and mercury residues in livers. This enables us to keep a watch on the effects of withdrawals of permitted uses of some of these chemicals, and to examine geographical variation in residues.

For 2000, the livers of 84 Eurasian sparrowhawks *Accipiter nisus* and seven grey herons *Ardea cinerea* from various localities in Scotland, England and Wales, were analysed for dichlorodiphenyldichloroethylene (DDE), hexachloro-epoxy-octahydro-dimethanonaphthalene (HEOD), PCBs and mercury (Hg). DDE and HEOD are the major metabolites that are detected in the tissues of animals exposed to dichlorodiphenyltrichloroethane (DDT) and dieldrin/aldrin/endrin, respectively. Average concentrations of organochlorine pesticides were generally low. DDE residues in sparrowhawks were similar to those detected in birds in 1999 and liver residues of HEOD were slightly but significantly lower. Liver concentrations of PCBs and Hg in sparrowhawks that died in 2000 were both significantly higher than in birds that died in 1999. There were no significant differences in liver contaminant concentrations in herons between birds that died in 1999 and 2000.

Over the whole monitoring period (1963–2000), there were long-term declines in residues of DDE, HEOD and mercury in the livers of sparrowhawks, herons and kestrels. These declines appear to have now largely levelled off and concentrations are mostly stable, although average liver mercury residues in sparrowhawks have increased progressively, albeit moderately, since 1996. The long-term declines in the metabolites (DDE, HEOD) of organochlorine pesticides confirm the effectiveness of progressive restrictions that were placed on the use and release of the parent compounds. There is little evidence of major long-term declines in PCB residues in sparrowhawks and kestrels, despite the restrictions in the use of these compounds. In contrast to terrestrial-feeding predatory birds, liver PCB residues in the piscivorous heron have declined significantly since the late 1970s, although there has been little significant change since approximately the mid 1980s.

1.3 Organochlorines and mercury in peregrine falcon *Falco peregrinus* eggs

Over the whole monitoring period, concentrations of DDE and HEOD have declined in peregrine eggs in Britain and the average eggshell index has now fully recovered to the level before the widespread use of DDT. PCB concentrations also declined in peregrine eggs, particularly during the 1980s, although this decline has not been observed in all regions of Britain and there has been little decline in levels overall during the 1990s. Mercury concentrations have not changed significantly in peregrine eggs since monitoring began in the mid 1980s.

1.4 Organochlorines and mercury in merlin *Falco columbarius* eggs

Single eggs collected in 2000 from eight merlin clutches from various parts of Scotland and England were analysed. DDE, HEOD and PCBs were detected in almost all eggs, but at low levels except for one egg that contained a particularly high total PCB residue. Mercury was also detected in all the eggs and concentrations in some were relatively high. The mean shell index value was approximately 95% of the pre-DDT value.

Over the whole of the monitoring period, contaminant concentrations have generally been higher in merlin eggs than in the eggs of other species. Most merlin eggs are still contaminated with organochlorine pesticides but concentrations have declined significantly since bans on the use of these compounds have been implemented. This decline has been accompanied by an increase in shell indices and a widespread increase in breeding merlins in Britain. Neither PCB nor mercury residues have obviously declined in merlin eggs across Britain as a whole (apart from perhaps an initial decline in PCB concentrations during the 1970s); high mercury residues have consistently been detected in birds from the Shetlands and Orkneys. The PCB and mercury concentrations in the most-contaminated eggs are of a magnitude associated with embryotoxic effects in other species.

1.5 Organochlorines and mercury in golden eagle *Aquila chrysaetos* eggs

Single eggs from eight clutches from Scotland and England were analysed in 2000. These confirm the low levels of contamination in eggs found in recent years. Over the whole monitoring period, concentrations of organochlorine pesticides have declined significantly during the monitoring period whereas there has been no overall pattern of change in PCB and mercury concentrations or shell indices for eagles from throughout Scotland. Contaminant concentrations were higher in eggs from western than eastern areas, and generally greater in the eggs of coastal than inland birds in the west. The current levels of contamination in eagle eggs are generally unlikely to be directly embryotoxic, although PCB concentrations in some eggs may result in adverse effects.

1.6 Organochlorines and mercury in northern gannet *Morus bassanus* eggs

The northern gannet is the only British seabird in which contaminant levels have been monitored continuously since 1970, and so has become a key indicator species in marine pollution. Eggs from Ailsa Craig were analysed in 2000 and contaminant concentrations were generally low. Total PCB and mercury concentrations were slightly but significantly higher than in 1998, the last time eggs from Ailsa Craig were analysed; there was no significant difference between the two years in DDE and HEOD concentrations or eggshell indices. Over the whole monitoring period (1970–2000), DDE and HEOD concentrations in gannet eggs have generally declined at Ailsa Craig and at three other colonies (Bass Rock, St Kilda, Hermaness) that have been monitored over at least an eight-year time span. There have been no general long-term trends in PCBs and mercury residues in gannet eggs, concentrations decreasing at some colonies but remaining the same or increasing at others.

1.7 Organochlorines and mercury in white-tailed eagle *Haliaeetus albicilla* eggs

Two white-tailed eagle eggs were received for analysis in 2000. They contained low concentrations of DDE, HEOD and mercury but relatively high total PCB concentrations. The eggs had the lowest shell index values recorded to date for intact white-tailed eagle eggs received at CEH.

Eight white-tailed eagle eggs in total have been analysed between 1986 and 2000. Four had DDE concentrations above the lowest observed effect level for eggshell thickness in white-tailed eagles and two of them also exceeded the lowest observed effect level for productivity in this species (Helander *et al.* 2002). Total PCB concentrations in all but one of the eggs were in the range associated with adverse effects in various avian species, and were ten times the mean residue level detected in golden eagle eggs collected from the west coast of Scotland over the same time period.

1.8 Rodenticide residues in barn owls *Tyto alba*

Second-generation rodenticides have largely replaced warfarin and other first-generation rodenticides. They are both more toxic to birds and more persistent in the tissues of prey than the first-generation rodenticides. This enhances their potential to cause secondary poisoning in avian predators, and they have been considered a potential threat to barn owls. Barn owls have been collected as part of the Wildlife & Pollution study since 1983 and analysed for second-generation anticoagulant rodenticides. Fifty-three barn owls received at CEH in 2000 were analysed for difenacoum, bromadiolone, brodifacoum and flocoumafen. The residues of one or more of these compounds were found in the livers of 21 (40%) birds, and seven (13%) had levels in the potentially lethal range (>0.1-0.2 μg/g; see Section 9.3 for details). The results for the 2000 sample were consistent with the trend reported for earlier years that indicated the increase since 1983 (when monitoring began) in the proportion of birds containing residues was reaching an asymptote of approximately 40%.

2 Organochlorines and mercury in the livers of predatory birds

2.1 Introduction

The main objective of this work was to analyse the carcasses of predatory birds, supplied by members of the public, in order to continue the monitoring of organochlorine and metal residues in livers. The chemicals of interest included DDE (from the insecticide DDT), HEOD (from the insecticides aldrin and dieldrin), PCBs (polychlorinated biphenyls from industrial products) and Hg (mercury from agricultural and industrial sources). Concentrations of gamma-hexachlorocyclohexane (g-HCH) are also reported. The concentrations of organochlorines are given as µg/g in wet weight (wet wt) and of mercury as µg/g in dry weight (dry wt).

The species analysed were the Eurasian sparrowhawk *Accipiter nisus*, representing the terrestrial environment, and the fish-eating grey heron *Ardea cinerea*, which represented the aquatic environment. A small number of other species were also received during 2000. These were not analysed for organochlorine and mercury residues because of the reduction in the scope of the monitoring scheme agreed in 1998. However, post-mortem examinations were carried out on each of these birds, relevant information being recorded and the cause of death determined (and reported back to the volunteer who submitted the carcass). Body organs and tissues from *all* birds received at CEH in 2000 are archived at -20°C and can be analysed for organochlorines and other contaminants in specific future studies.

Findings from previous years are given in earlier reports in the present series and in Newton *et al.* (1993).

2.2 Results for birds received at CEH in 2000

During the past year, the livers from 84 sparrowhawks and seven herons were analysed. These included some birds that had died in earlier years but which were only sent to CEH in 2000. The results from all these birds are listed in Table 2.1 and the geometric means for each chemical (data for birds found dead in 2000 only) are given in Table 2.2. Data from the birds collected in earlier years are included in the three-yearly combined analysis that is given as part of the present report.

None of the sparrowhawks collected during 2000 had liver concentrations of g-HCH, DDE or HEOD that were of the magnitude associated with lethality. One of the seven herons analysed (13208) had a liver HEOD concentration that was within the range associated with sub-lethal effects in adult birds of some species (Peakall 1996).

Liver PCB residues were relatively high (20-69 µg/g) in ten sparrowhawks and one heron. Almost all of these birds had no visible fat depots in their bodies and residues of up to 100 µg/g are not exceptional in starved individuals. One bird (number 13348, which was found in Cambridgeshire) had a PCB residue of 140 µg/g, much higher than liver concentrations in all the other birds received at CEH in 2000, and it did contain small amounts of fat in the body. The PCB residue level in this bird was exceptionally high and

may have been a contributory cause of death. However, the biological significance of liver total PCB residues in birds is uncertain because there is considerable overlap in levels between birds dying from PCB poisoning and those surviving. The toxicological significance of residues is better-defined for individual PCB congeners.

Mercury concentrations were low in sparrowhawks and, as found in previous years, generally higher in herons. Liver total mercury concentrations in all birds were well below the concentration (30 µg/g wet wt, equivalent to approximately 105 µg/g dry wt) associated with toxic effects in birds of prey (Thompson 1996).

DDE residues in sparrowhawks collected in 2000 were similar to those in birds received at CEH in 1999. In contrast, liver HEOD residues in sparrowhawks in 2000 were slightly but significantly lower than in the previous year (0.166 µg/g vs. 0.100 µg/g; Table 2.3), although liver concentrations in the 1999 birds had actually been higher than in birds that had died in 1998 (Shore *et al*. 2002). Liver concentrations of PCBs and Hg in sparrowhawks that died in 2000 were both significantly higher than in birds that died in 1999. There were no significant differences in liver contaminant concentrations in herons between birds that died in 1999 and 2000 (Table 2.3).

2.3 Long-term trends

The nationwide trends in DDE, HEOD, PCB and Hg residues for sparrowhawks and herons are given in Figures 2.1 and 2.2 respectively. Analyses began in 1963-64 for DDE and HEOD, 1967-68 for PCBs and 1967-70 for mercury, depending on the species. Long-term trends were analysed by linear regression analyses of log-transformed individual residues on year for the whole analytical period for each contaminant (Table 2.4), as has been done in previous long-term reviews (Newton *et al*. 1998). This gives an overall robust assessment of the significance of any long-term trends, although other regression models may give a marginally better fits to the data in a few instances. Separate analyses were carried out for each compound for the past six years (1995–2000) so that the most recent trends could be examined independently of earlier results (Table 2.4). Long-term trends for common kestrel *Falco tinnunculus* are also shown (Figure 2.3) as this species was monitored for contaminants for birds received at CEH up until 1998. The significance of both long-term trends in residue magnitude and trends over the last six years in kestrels (1993-98) are given in Table 2.5.

Of the terrestrial-feeding birds, sparrowhawks, which feed on birds, generally had higher liver DDE, PCB and Hg concentrations than kestrels, which prey mainly on small mammals. Herons, which largely take fish, had broadly similar levels of DDE to those in sparrowhawks but higher HEOD, PCB and Hg residues compared with sparrowhawks or kestrels, which may reflect greater bioconcentration of these compounds through aquatic compared with terrestrial food-chains.

Over the whole monitoring period, liver residues of DDE, HEOD and Hg have declined significantly in all three species. Average liver concentrations appeared to stabilise at low levels by the early 1990s in sparrowhawks and kestrels and perhaps a little earlier in herons. Individuals of all three species with high residues are still occasionally found, however. The long-term pattern of liver PCB residues in some of the species contrasts markedly with the decline observed for the other contaminants. There is evidence of a significant downward trend in liver total PCBs throughout the monitoring period in sparrowhawks (Table 2.4) but this decline is extremely slight. In fact, residues in recent years were only slightly lower than

the peak average concentration recorded in the late 1970s, and were higher than when monitoring first began, although it is possible that some of this difference may be a result of relatively poor analytical detection when analytical methods were first developed in the 1960s. In kestrels, there is no evidence of a significant decline in liver total PCBs over the whole of the monitoring period (Table 2.5 and Figure 2.3). In both species, there is some suggestion from the three-year running means of cyclicity in the magnitude of liver PCB residues over time. More detailed statistical analysis of the data is required to determine whether this is random variation or a real pattern associated with variation in climatic or other variables. Unlike in the terrestrial-feeding predatory birds, liver PCB residues have decreased significantly in herons over the whole of the monitoring period. Liver residues declined approximately two-fold during the 1970s and 1980s but have failed to decline significantly since then.

Over the shorter period (1995–2000 for sparrowhawk and heron, 1993-98 for kestrel), there have been less-marked changes in liver residues. DDE liver concentrations have not changed significantly in any of the species. Liver concentrations of HEOD have increased marginally in all three species (Figures 2.1-2.3), and the changes were statistically significant for sparrowhawks. However, the magnitude of the increases was small and current liver residues in sparrowhawks are an order of magnitude below the critical concentration for the population, as defined by Newton (1988). Liver PCB residues have increased in sparrowhawks and herons in the past six years, although the rise was only statistically significant for sparrowhawks. Interpretation of the toxicological significance of total PCB liver residues is difficult but inspection of the long-term data suggests that the recent rise in residues in sparrowhawks is within the variation in liver PCBs observed since 1970. During this period, PCBs did not prevent the recovery of the sparrowhawk population following the crash caused by exposure to organochlorine insecticides. In contrast to sparrowhawks, liver PCB residues in kestrels declined significantly in the last six years they were monitored (1993-98) but, again, this decline was within the range of variation observed historically. Liver mercury residues have not changed significantly in recent years in herons and kestrels but have increased significantly in sparrowhawks, although the scale of this increase is again moderate. Geometric mean mercury residues in sparrowhawks remain approximately two-fold lower than they were when at a peak in the mid-1970s, and are between one and two orders of magnitude below residues associated with toxic effects (Thompson 1996).

2.4 Summary

There have been general long-term declines in liver residues of organochlorine pesticides and mercury during the monitoring period. These declines appear now to have largely levelled off. There is some uncertainty over recent short-term trends in mercury accumulation by sparrowhawks, average liver residues having increased progressively, albeit moderately, since 1996. The long-term declines in organochlorine pesticides confirm the effectiveness of progressive restrictions that have been placed on the use and release of the parent compounds. There is little evidence of major long-term declines in PCB residues in sparrowhawks and kestrels, even though the use of these compounds in open systems has been prohibited in many countries since 1972 and their production in most industrial countries was terminated by the late 1970s (Hoffman *et al*. 2001). In contrast to terrestrial-feeding predatory birds, PCB residues in the piscivorous heron have declined significantly since the late 1970s, although there is little evidence of any significant change in liver residues since approximately the mid 1980s.

Table 2.1: Levels of organochlorines (µg/g wet wt) and mercury (µg/g dry wt) in the livers of juvenile (in first year) and adult (older than first year) predatory birds received during 2000.
* indicates missing data that were either not provided by the sender of the carcass or that could not be obtained from the sample received. ND indicates not detected

Bird No.	Year found	Vice-County	Age	Sex	pp'-DDE	HEOD	PCB	Hg	g-HCH
Eurasian sparrowhawk *Accipter nisus*									
13137	2000	Mid-West Yorkshire	A	F	0.513	0.285	3.129	1.904	ND
13140	1998	Pembrokeshire	J	F	ND	0.011	2.916	1.460	ND
13141	2000	Kincardineshire	A	M	5.998	0.121	22.790	12.545	ND
13145	2000	Hertfordshire	J	F	0.276	0.053	5.080	1.885	ND
13146	2000	Mid-West Yorkshire	A	F	0.343	0.041	2.130	0.924	0.006
13147	2000	Shropshire (Salop)	A	M	0.326	0.040	2.291	4.913	ND
13149	1998	Dorset	A	M	0.023	0.011	0.441	1.151	ND
13150	1996	Devon	A	F	1.189	0.168	8.253	6.499	ND
13151	1997	Dorset	J	F	0.519	0.200	1.766	2.705	ND
13154	2000	Radnorshire	A	F	10.497	0.846	27.704	5.411	ND
13160	2000	South Devon	A	F	5.156	0.203	17.515	4.201	ND
13174	1993	Carmarthenshire	A	M	0.327	0.062	3.548	1.877	ND
13178	1999	Pembrokeshire	A	F	9.580	3.636	18.874	6.434	ND
13182	2000	West Norfolk	J	M	21.054	0.887	19.979	18.243	ND
13187	2000	Cambridgeshire	J	M	14.958	0.482	14.904	9.512	ND
13188	2000	North Wiltshire	J	M	1.165	0.133	13.208	12.149	ND
13189	2000	Berkshire	J	M	0.805	0.085	5.332	8.086	ND
13193	2000	Westmorland with N. Lancs	J	M	17.272	0.436	29.759	19.324	ND
13194	2000	Bedfordshire	J	M	5.965	0.653	4.571	3.595	0.021
13195	2000	North Lincolnshire	A	M	16.636	2.007	68.793	14.024	ND
13198	2000	Warwickshire	A	F	0.307	0.057	15.894	1.613	ND
13201	2000	West Suffolk	J	F	1.584	0.079	1.703	4.298	ND
13202	2000	East Sussex	A	M	4.877	0.595	4.994	4.101	ND
13203	2000	South Essex	J	M	1.671	0.126	3.899	5.258	ND
13209	2000	East Sussex	J	M	ND	1.091	28.208	5.632	ND
13210	2000	West Norfolk	J	M	8.359	1.257	39.335	5.936	ND
13212	2000	Cheshire	A	F	0.166	0.097	1.962	2.977	ND
13214	2000	Kirkudbrightshire	J	F	1.967	0.136	2.021	13.497	ND
13217	2000	West Gloucestershire	A	M	2.337	0.097	5.446	2.424	0.010
13220	2000	Warwickshire	J	F	9.054	0.881	19.358	12.353	ND
13230	2000	Cambridgeshire	J	M	2.485	0.111	2.734	8.712	0.006
13232	2000	West Gloucestershire	J	F	0.139	0.031	0.569	0.575	ND
13243	2000	Cheshire	J	M	3.400	2.126	31.858	10.053	ND
13247	2000	Midlothian	J	M	0.608	0.218	5.009	6.300	ND
13251	2000	South East Yorkshire	J	M	3.535	0.286	4.425	3.664	ND
13264	2000	Berkshire	J	M	0.092	0.015	0.360	0.752	ND
13268	2000	East Suffolk	J	M	1.640	0.093	0.574	1.192	ND
13272	1998	South Aberdeenshire	J	F	0.152	0.021	0.479	2.052	ND
13273	1998	Kincardineshire	J	M	11.574	0.231	45.723	12.892	ND
13274	2000	South Aberdeenshire	J	M	0.388	0.023	1.342	13.470	ND
13276	2000	West Perthshire (with Clackmannan)	J	M	0.083	0.007	0.371	4.275	ND
13278	2000	Renfrewshire	J	F	0.032	0.015	0.230	0.916	ND
13282	2000	East Sussex	J	F	0.254	0.017	0.549	1.436	ND
13285	2000	East Kent	J	F	3.620	0.090	0.473	1.645	ND

Bird No.	Year found	Vice-County	Age	Sex	pp'-DDE	HEOD	PCB	Hg	g-HCH
13287	2000	East Kent	J	M	2.203	0.046	1.992	1.955	ND
13289	2000	South West Yorkshire	J	F	0.017	0.035	0.335	0.409	ND
13291	2000	Derbyshire	J	M	0.939	0.128	4.367	1.856	ND
13292	2000	Staffordshire	J	F	0.359	0.115	4.918	1.137	ND
13294	2000	South Lincolnshire	J	F	0.493	0.099	1.345	1.829	ND
13295	2000	Ayrshire	J	F	0.061	0.012	0.610	2.803	ND
13298	2000	North West Yorkshire	J	F	0.037	0.016	0.513	0.865	ND
13299	2000	Hertfordshire	J	F	6.576	0.687	3.426	1.059	ND
13300	2000	Huntingdonshire	J	F	0.429	0.024	0.838	0.632	ND
13303	2000	*	J	F	1.142	0.070	3.797	3.363	ND
13315	*	East Cornwall	J	F	0.025	0.020	0.064	0.573	ND
13318	*	East Cornwall	J	F	1.412	0.076	1.804	3.161	0.026
13322	*	East Cornwall	J	F	0.240	0.037	0.333	0.837	ND
13323	*	East Cornwall	A	M	3.390	0.038	10.461	4.341	0.029
13324	*	East Cornwall	J	M	0.559	0.121	2.063	2.478	ND
13330	2000	Cambridgeshire	J	F	2.565	0.074	0.667	1.282	0.028
13338	2000	Hertfordshire	J	M	2.479	0.145	0.805	0.747	0.025
13346	2000	Cambridgeshire	*	F	0.077	0.005	0.261	0.423	ND
13348	2000	Cambridgeshire	A	F	6.820	1.195	139.588	10.275	0.036
13351	2000	Shropshire	A	F	23.686	3.078	23.208	5.518	ND
13362	2000	South Wiltshire	J	M	0.100	0.015	1.121	0.751	ND
13380	2000	Berkshire	J	M	16.168	0.349	27.889	2.049	0.021
13382	2000	Leicestershire	J	F	1.442	0.001	0.320	1.520	0.027
13383	2000	Buckinghamshire	J	M	4.744	0.268	24.258	6.674	0.038
13385	2000	Hertfordshire	J	M	0.312	0.026	3.228	1.254	ND
13387	2000	Berkshire	J	F	0.171	0.022	0.973	0.175	ND
13389	2000	Shropshire	J	F	5.403	0.509	16.993	1.866	ND
13391	2000	South Lancashire	J	M	0.272	0.028	2.153	1.799	ND
13399	1998	North Ebudes	J	M	0.147	0.069	2.575	4.626	ND
13400	1998	North Lincolnshire	J	M	0.467	0.160	1.069	1.037	0.009
13401	1998	Argyll Main	J	F	1.837	0.203	5.784	5.380	ND
13402	1998	*	J	F	0.402	0.064	2.421	2.120	ND
13404	2000	North Edudes	J	F	0.808	0.072	7.040	8.384	ND
13405	1999	South Lincolnshire	J	M	1.054	0.169	2.503	2.209	ND
13408	2000	East Sussex	A	M	33.451	0.831	46.790	4.060	ND
13416	2000	Kincardineshire	J	M	1.540	0.712	1.840	9.328	ND
13417	2000	South West Yorkshire	J	M	0.104	0.030	1.058	1.086	ND
13421	2000	Oxfordshire	J	M	0.982	0.001	2.024	0.804	0.026
13422	2000	East Inverness-shire	J	M	0.088	0.001	0.498	8.508	ND
13451	2001	South West Yorkshire	A	F	0.824	0.179	1.160	1.197	ND
Grey heron *Ardea cinerea*									
13164	1999	Pembrokeshire	J	F	0.100	0.131	0.896	6.422	ND
13165	2000	Pembrokeshire	J	F	0.172	0.107	0.988	19.09	ND
13166	1992	Pembrokeshire	J	M	0.017	0.042	0.184	3.162	ND
13167	1998	Pembrokeshire	J	F	0.726	0.130	7.076	2.136	ND
13208	2000	South Devon	A	F	1.378	6.281	23.313	43.56	ND
13211	1999	East Sussex	J	M	3.374	0.443	5.089	18.67	ND
13225	2000	East Sussex	A	M	2.724	0.057	0.296	8.055	ND

Table 2.2: Geometric mean levels of pollutants in the sparrowhawk and heron in Table 2.1 (data are only for birds found dead in 2000). GSE=geometric standard error. Organochlorine insecticide and PCB concentrations are in µg/g wet wt, mercury in µg/g dry wt.

	pp'-DDE	HEOD	PCB	Hg
Sparrowhawk				
Geometric mean	0.977	0.100	3.442	2.857
N	67	67	67	67
Range within 1 GSE	0.764 – 1.249	0.080 – 0.124	2.838 – 4.175	2.505 – 3.259
Heron				
Geometric mean	0.864	0.337	1.896	18.851
N	3	3	3	3
Range within 1 GSE	0.377 – 1.984	0.077 – 1.472	0.516 – 6.971	11.580 – 30.688

Table 2.3: Comparison of geometric mean residue levels (log values) from birds collected in 1999 and 2000; values for the two years and the statistical t-values are shown. Minus values indicate a decrease and plus values indicate an increase from 1999. Organochlorine insecticide and PCB concentrations are in µg/g wet wt, mercury in µg/g dry wt.

		pp'-DDE	HEOD	PCB	Hg
Sparrowhawk					
	1999	1.003	0.166	2.009	1.891
	2000	0.977	0.100	3.442	2.857
		$t_{117} = -0.29$	$t_{117} = -2.01$*	$t_{117} = +1.99$*	$t_{117} = +2.42$*
Heron					
	1999	0.520	0.132	4.018	11.444
	2000	0.864	0.337	1.896	18.851
		$t_8 = +0.42$	$t_8 = +0.48$	$t_8 = -0.50$	$t_8 = +0.84$

Significance of difference: *P<0.05; **P<0.01; ***P<0.001.

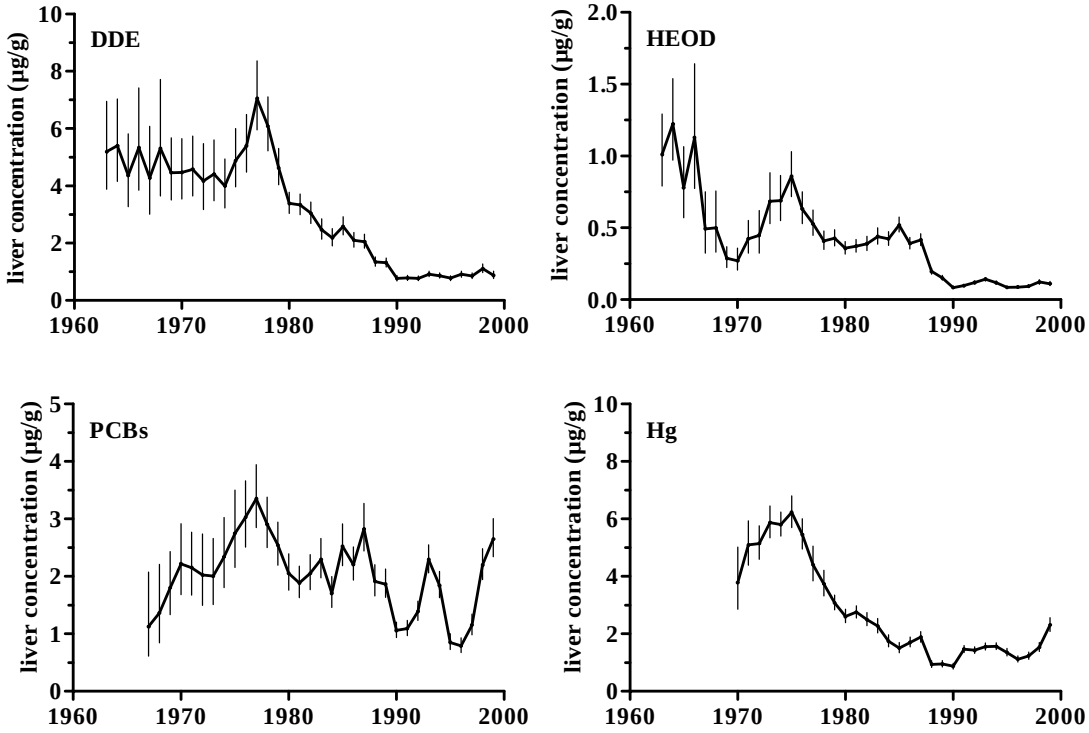

Figure 2.1: Trends in pollutant residues in livers of sparrowhawks, 1963-99. Data are three-year moving geometric means with one geometric standard error on either side and are wet wt concentrations for DDE, HEOD and PCBs and dry wt concentrations for Hg.

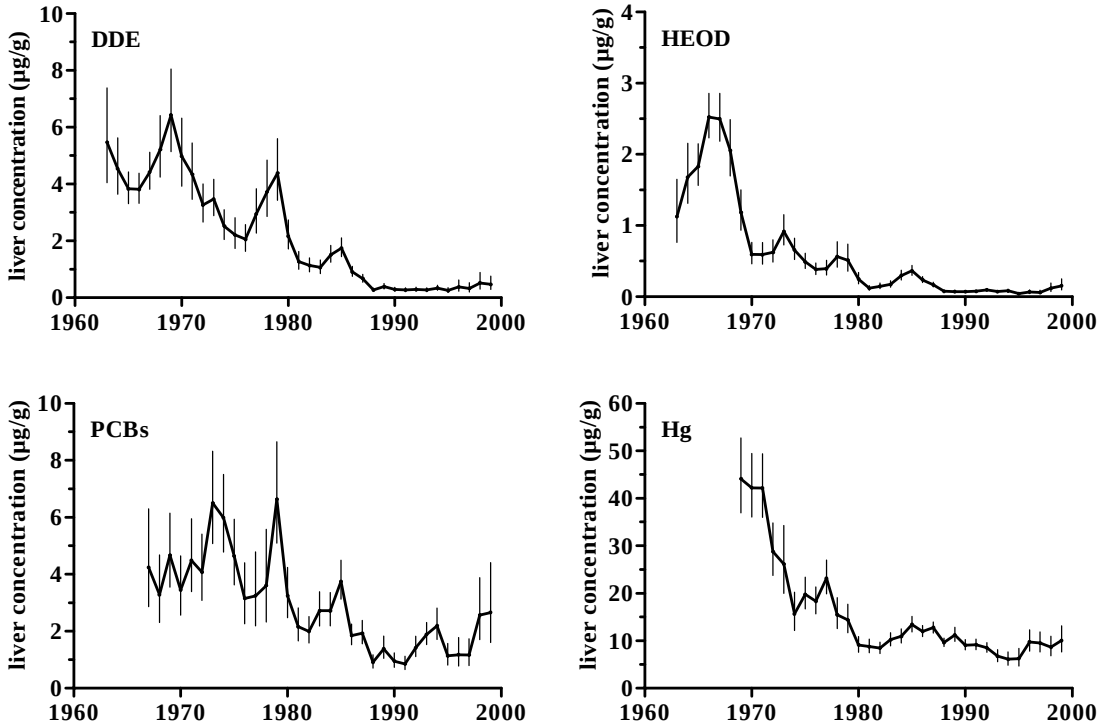

Figure 2.2: Trends in pollutant residues in livers of herons, 1963-99. Data are three-year moving geometric means with one geometric standard error on either side and are wet wt concentrations for DDE, HEOD and PCBs and dry wt concentrations for Hg.

Table 2.4: Trends in pollutant levels in livers of sparrowhawks and herons during 1963–2000 and 1995–2000. Figures show sample sizes (N) and linear regression coefficients (b) based on log values regressed against year. (Analyses for PCBs and Hg were started in 1967 and 1970 respectively in sparrowhawk and heron).

	1963–2000			1995–2000		
	N	b		N	b	
Sparrowhawk						
pp'-DDE	1925	-0.033	***	398	0.020	ns
HEOD	1926	-0.032	***	398	0.042	*
PCB	1881	-0.008	**	398	0.172	***
Hg	1675	-0.017	***	393	0.087	***
Heron						
pp'-DDE	814	-0.042	***	33	0.107	ns
HEOD	804	-0.048	***	33	0.170	ns
PCB	680	-0.022	***	33	0.135	ns
Hg	517	-0.019	***	33	0.040	ns

*P=<0.05; **P=<0.01;***P<0.001; ns=not significant

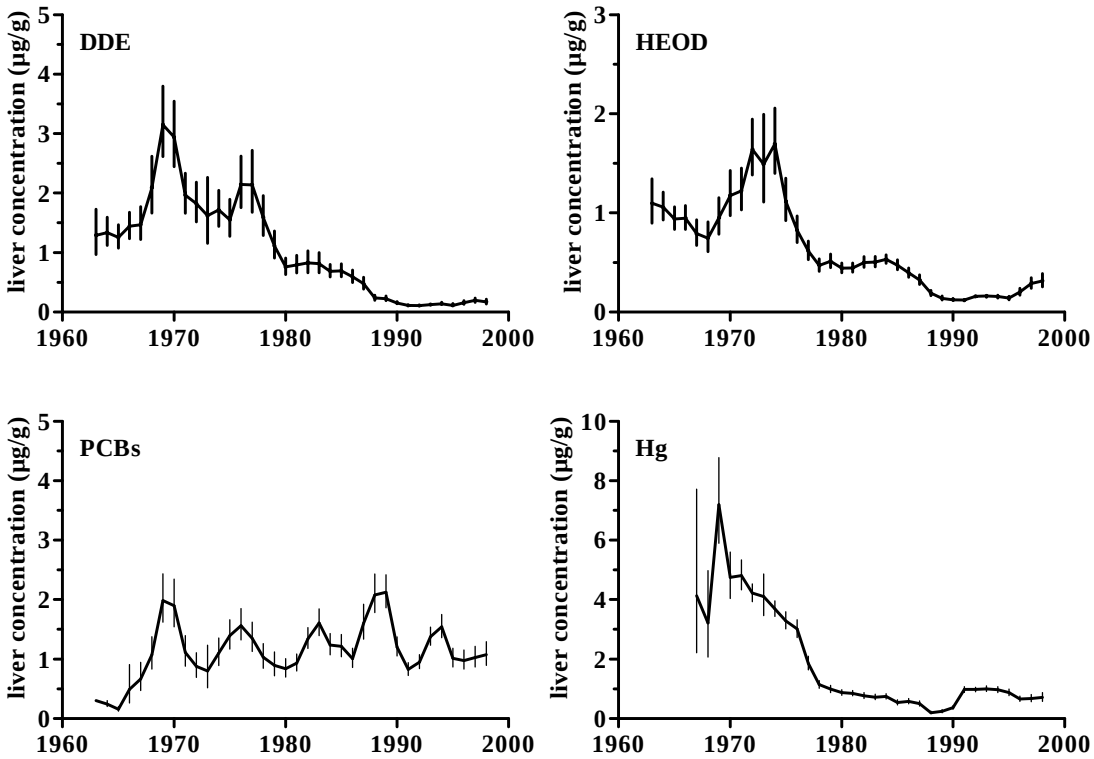

Figure 2.3: Trends in pollutant residues in livers of kestrels, 1963-97. Data are three-year moving geometric means with one geometric standard error on either side and are wet wt concentrations for DDE, HEOD and PCBs and dry wt concentrations for Hg.

Table 2.5: Trends in pollutant levels in livers in kestrels during 1963-98 and 1993-98. Figures show sample sizes (N) and linear regression coefficients (b) based on log values regressed against year.

	1963-98			**1993-98**		
	N	**b**		**N**	**b**	
pp'-DDE	1419	·0.0425	***	185	·0.0233	ns
HEOD	1390	·0.0326	***	183	0.0387	ns
PCB	1278	0.0021	ns	186	·0.0669	**
Hg	1083	·0.0322	***	182	·0.0339	ns

*P=<0.05; **P=<0.01;***P<0.001; ns=not significant

3 Organochlorines and mercury in peregrine falcon *Falco peregrinus* eggs

3.1 Introduction

The findings from all peregrine eggs analysed between 1961 and 1986 were summarised in Newton *et al*. (1989), and updated in the 1997-98 report in the present series (Newton *et al*. 1998). Peregrine eggs are not formally part of the Wildlife & Pollution monitoring scheme, although analyses of eggs received at CEH have usually been analysed as part of CEH's core science programme and reported in the present series of Wildlife and Pollution reports.

No peregrine eggs were analysed in 2000. The long-term trends in residues in eggs received at CEH have been updated to include the few eggs analysed since the last report.

3.2 Long-term trends

Long-term trends in contaminant levels in peregrine falcon eggs have been described by Newton *et al*. (1989) and last reviewed by Newton *et al*. (1998). The long-term trends for contaminant levels for the 719 unhatched peregrine eggs that have been examined between 1963 and 1999 are given in Figure 3.1 as plots of three-year moving geometric means (with geometric standard errors). Linear regression analyses for individual log-transformed residues against year are given in Table 3.1. Each of the eggs was from a separate clutch and the data are for Britain as a whole. Regional variation in contaminant levels in peregrine eggs was described in the last review in the present series of reports (Newton *et al.* 1998) and is not repeated here because relatively few eggs have been analysed since 1997, and the regional differences identified in that report have not have changed significantly.

Concentrations of organochlorine compounds declined significantly in peregrine eggs over the course of the monitoring period (Figure 3.1 and Table 3.1) and are currently at low levels that are below concentrations that are associated with embryotoxic effects (Blus 1996; Hoffman *et al*. 1996; Peakall 1996). The decline in DDE and HEOD concentrations followed the voluntary and subsequently mandatory bans on the agricultural use of DDT, dieldrin and aldrin. PCBs were restricted to closed systems in Britain in the early 1970s as a mitigation measure designed to restrict their release to the environment, and total PCB concentrations have declined overall in peregrine eggs. However, this decline was only apparent from the late 1970s onwards (Figure 3.1), suggesting that there was a lag phase before the mitigation began to have an effect, perhaps reflecting the persistence of these compounds in both the environment and biota. PCB concentrations in peregrine eggs appear to have remained fairly stable since the early 1990s. Current levels presumably reflect both the continuing release into the environment from usage of PCB-containing products and again the persistent nature of these compounds once they are released. An improvement in eggshell indices accompanied the decline in organochlorines in peregrine eggs. Current analysis of the whole dataset for eggshell indices suggests that this recovery is better-described by a second-order polynomial regression than a linear regression model (comparison of fits for the two models: $F_{(1,576)} = 7.37$, $P<0.01$), and that eggshell thickness has now fully recovered to pre-DDT values (Figure 3.1).

Peregrine eggs were analysed regularly for mercury from 1985 onwards and only occasional eggs were analysed prior to this date. This is reflected by the relatively high degree of variation in average concentrations in earlier years. Overall, there has been no significant decline in mercury concentrations in peregrine eggs and most eggs contain residues, although these are generally low, and well below concentrations associated with embryotoxic effects (Thompson 1996).

3.3 Summary

Concentrations of DDE and HEOD have declined in peregrine eggs in Britain and eggshell indices have recovered to pre-DDT levels. PCB concentrations also declined in peregrine eggs in the 1980s in particular, although this decline has not been observed in all regions of Britain (Newton *et al*. 1998) and there was little decline in levels overall during the 1990s. Mercury concentrations have not changed significantly in peregrine eggs since more intensive monitoring began in the mid 1980s.

Table 3.1: Trends in pollutant levels in peregrine falcon eggs over the period 1964-98 as revealed by regression analyses of individual (log_{10}) residue levels against year. N=number of clutches represented at one egg per clutch, b=regression coefficient (slope), significance of the regression analysis is indicated by: *P<0.05, **P<0.01, ***P<0.001.

	DDE		HEOD		PCB		Hg		SI	
	N	**b**	**N**	**b**	**N**	**b**	**N**	**b**	**N**	**b**
All areas	719	-0.044 ** *	719	-0.040 ***	686	-0.010 ***	214	-0.0027 ns	580	0.009 ***

ns=not significant

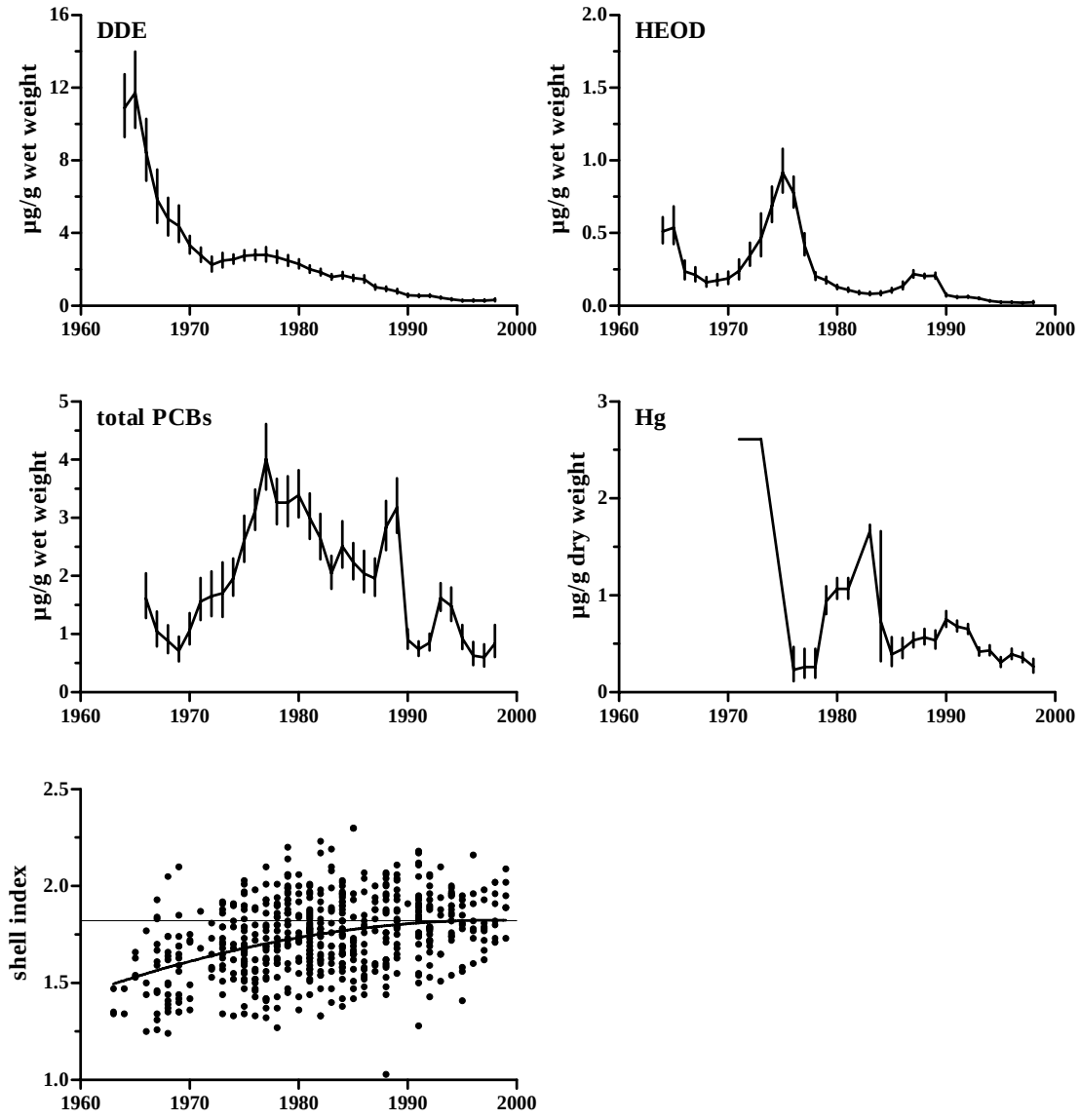

Figure 3.1: Long-term trends in pollutant residues (1967-99) and shell indices (1970–2000) in peregrine falcon eggs. Data for contaminant concentrations are three-year moving geometric means with one geometric standard error on either side. Data for shell indices are values for individual eggs. The regression line shown for the shell indices is a second-order polynomial regression for individual shell indices against year. The dotted line shows the pre-DDT average eggshell index.

4 Organochlorines and mercury in merlin *Falco columbarius* eggs

4.1 Introduction

The merlin *Falco columbarius* declined in numbers in both Europe and North America between the 1950s and 1970s following widespread introduction and use of DDT and other organochlorine pesticides. Eggshell-thinning and/or reduced breeding success was linked to organochlorine levels in eggs, impaired breeding success and subsequent effects on populations (see Newton *et al*. 1999a and references therein). The long-term trends (1967-97) in organochlorine and mercury concentrations in merlin eggs from Britain have previously been summarised by Newton & Haas (1988), Newton *et al*. (1998) and Newton *et al*. (1999a). Recent reports in the present series have documented the levels of contaminants in merlin eggs that have been received at CEH in 1998 (Newton *et al*. 2000) and in 1999 (Shore *et al*. 2002). In the current report, the contaminant concentrations in seven eggs (one per clutch) collected during 2000 are given (Table 4.1) and the long-term national trends in contaminant residues and shell index values in merlin eggs are also updated.

4.2 Results for eggs received in 2000

The analysis of eggs collected in 2000 confirm results from analyses carried out in previous years that indicated that the eggs of merlins in Britain are still generally contaminated with organochlorine pesticides, PCBs and mercury (Table 4.1). DDE and HEOD were detected in all seven and in six out of the seven eggs, respectively, although concentrations of both compounds were always low and can be considered 'background'. All of the eggs also contained PCBs and one egg, from the Scottish Borders, contained a particularly high total PCB residue (29.1 µg/g wet wt). PCBs may have contributed to the failure of the egg; total PCB concentrations of between 8 and 25 µg/g wet wt have been associated with bill deformities and decreased hatching success in a range of avian species including raptors (Hoffman *et al*. 1996), although egg PCB concentrations that are associated with impaired reproductive success in merlins have not been defined. Mercury was also detected in all of the eggs, and concentrations in the three eggs from the Scottish islands and one egg from the Borders were high compared to the three-year rolling geometric mean concentrations recorded previously for merlin eggs (see Section 4.3), although relatively high mercury concentrations have been detected previously in eggs from the Northern Isles (see previous reports in the present series). The residues in the two most contaminated eggs, both from Shetland, were equivalent to 3.61 µg/g wet wt and 2.20 µg/g wet wt. There are no data that relate mercury concentrations in eggs to hatching success for merlins but total mercury concentrations greater than approximately 2 µg/g wet wt have been associated with impaired hatching in laboratory studies on some species (Thompson 1996). However, while mercury concentrations of this magnitude have been measured in the eggs of free-living birds, they have only occasionally been associated with impaired hatching, perhaps indicating that there is considerable variation in sensitivity between species (Thompson 1996). Therefore, although the mercury residues detected in the merlin eggs collected from Shetland in 2000 were of the magnitude associated with embryotoxic effects in some species, it is unknown whether mercury contributed to the failure of these eggs to hatch.

Overall, shell-indices could be calculated for five of the eggs analysed in 2000. The mean shell index value was 1.19, approximately 95% of the pre-DDT value.

Table 4.1: Residue levels and shell indices (SI) for merlin eggs received in 2000. Organochlorine insecticide and PCB concentrations are expressed as µg/g wet wt (µg/g lipid weight in parentheses) and mercury is expressed as µg/g dry wt. * indicates where shell indices could not be measured because of the poor condition of the eggshell.

Number	Year	County	SI	pp'-DDE		HEOD		PCB		Hg
Northern Isles										
E7789	2000	Shetland	1.26	0.733	(11.286)	0.031	(0.484)	1.645	(25.316)	18.433
E7793	2000	Shetland	*	0.496	(8.932)	0.017	(0.297)	3.258	(58.708)	13.386
Western Isles										
E7824	2000	Rum	*	1.781	(33.528)	0.015	(0.282)	2.827	(53.205)	7.042
Southern Scotland										
E7763	2000	Borders	1.09	1.639	(30.998)	0.152	(2.884)	29.106	(550.48)	9.986
E7764	2000	Borders	1.18	0.106	(1.776)	ND	(ND)	0.300	(5.027)	5.767
E7766	2000	Borders	1.19	2.378	(67.215)	0.067	(1.903)	2.831	(80.031)	4.361
Northern England										
E7772	2000	Cumbria	1.21	3.13	(45.223)	0.388	(5.600)	9.182	(132.67)	3.736

ND = Not detected.

4.3 Long-term trends in merlin eggs

Over the period 1963 to 2000, contaminants have been analysed at CEH in up to 714 merlin eggs, each from a different clutch. When more than one egg was collected from a clutch, the egg that was analysed was selected at random. The number of eggs analysed per year were generally lower in the earlier years than in the later ones. The eggs came from various parts of the country but approximately one-third were from north-east England and were collected by Northumbria Ringing Group. Trends in changes in residue levels over time in eggs from north-east England generally mirrored those for most of the rest of Britain, and a detailed regional breakdown of contaminant levels in merlin eggs is given by Newton *et al*. (1999a).

Long-term changes over time in contaminant concentrations and shell indices for merlin eggs are given as a plot of three-year moving geometric means (contaminant concentrations) and individual values (shell indices) against year for eggs collected from throughout Britain (Figure 4.1). Data are also presented as geometric mean values for the two periods, 1967-86 and 1987–2000 (Table 4.2). The data were split at 1987 because this was when a complete ban on DDT, aldrin and dieldrin use in Britain was implemented. This is consistent with analysis presented when the long-term data were last reviewed (Newton *et al*. 1998).

In previous reviews of the long-term data (Newton *et al*. 1998; Newton *et al*. 1999a), residues in merlin eggs have been expressed on a lipid weight (lipid wt) basis. This is because concentrations were not always recorded on a wet wt basis in the early years when merlin eggs were first analysed. Review of the long-term data on a wet wt basis would therefore result in the exclusion of some data from early years. However, descriptions of long-term trends in residues in the eggs of other species are given on a wet wt basis in the present report

(see Sections 3, 5 and 6). To facilitate comparison of contaminant concentrations between merlin eggs and the eggs of other species, the relationship betweenlipid wt concentration and wet wt concentration is given in Figure 4.2 for DDE, HEOD and PCBs. The geometric mean wet wt concentrations of these compounds in eggs collected after 1986 are also given with Table 4.2.

Residues of HEOD were generally much lower than those of DDE, but concentrations of each declined between 1963 and 2000 (Figure 4.1). Linear regression analysis of $\log_{10}$ contaminant concentrations against year indicated that these declines were highly significant (gradient = -0.21, $F_{(1,701)}$= 83.6 for DDE; gradient = -0.25, $F_{(1,682)}$= 76.8 for HEOD, both P<0.0001). This decline was also reflected by the comparison of DDE and HEOD concentrations in eggs before and after 1987, the post-1986 geometric mean values for both compounds being just over half their pre-1987 values (Table 4.2).

PCB residues in merlin eggs fluctuated since 1970, although linear regression analysis suggested that overall there has been a weak decline (gradient = -0.008, $F_{(1,699)}$= 8.81, P<0.005). High residues that were detected in a relatively small number of eggs in the 1970s influenced the statistical significance of this decline markedly. When data for all eggs collected before 1987 were pooled, the influence of these relatively few high residues on the geometric mean for this period was not pronounced and, consequently, the mean pre-1987 concentration was similar to that for eggs collected after 1986 (Table 4.2). Overall, there appears to have been no real change in PCBs residues during the 1980s and 1990s.

Mercury concentrations showed no significant change over time when analysed by regression analysis (gradient = +0.007, $F_{(1,608)}$= 3.77, P>0.05). Concentrations did differ between pre-1987 and post-1986 eggs and, in contrast to the organochlorine pesticides, residues were significantly higher in the later period (Table 4.2). However, this is probably the result of the way that levels have varied over the years, concentrations being low in the early 1980s, and may also reflect variation between the time periods in the proportion of eggs that came from Shetland, Orkney and parts of north-west Scotland. Some eggs from these areas have had unusually high mercury residues (Newton *et al*. 1999a), as was still evident in the eggs collected in 2000, and it is doubtful whether there has been any real overall increase in mercury residues in eggs since 1987.

Shell indices for merlin eggs increased significantly over the whole time that analyses has been carried out ($F_{(1,598)}$= 100.6, P<0.0001, Figure 4.1) and values for post-1986 eggs were significantly higher than those for eggs collected earlier (Table 4.2). This is consistent with what would be expected given that DDE contamination in eggs, a main causal agent of eggshell thinning (Cooke 1973), declined during this period. However, eggshell indices have still not fully recovered to pre-DDT values in merlins.

4.4 Summary

In comparison to the eggs of other predatory species that have been examined over the same time period, merlin eggs have contained higher concentrations of the contaminants that have been monitored. For example, the post-1986 geometric mean wet wt concentrations for DDE, HEOD PCB and dry wt concentration for mercury in merlin eggs (Table 4.2) were some 6-80 times greater than the equivalent concentrations in golden eagle eggs (Table 5.3). Currently, most merlin eggs are still contaminated with organochlorine pesticides but concentrations have declined significantly since bans on the use of these compounds have been implemented.

This decline has been accompanied by an increase in shell indices and a widespread increase in breeding merlins in Britain, although there has been some regional variation in recovery (see Newton *et al*. 1999a and references therein). Declines in organochlorine pesticide residues, with concomitant population recoveries, have also been observed in other raptors in Britain (Ratcliffe 1980; Newton 1986).

Neither PCB nor mercury residues have clearly declined in merlin eggs across Britain as a whole, apart from perhaps an initial decline in PCB concentrations during the 1970s. PCB and mercury residues detected in some eggs have been high both in the past and currently. Whether this simply reflects natural variability in exposure or hotspots of contamination is not clear. However, high mercury residues have consistently been detected in birds from the Shetlands and Orkneys (Newton *et al*. 1999a), and this would suggest that there is significant regional variation in exposure to this contaminant at least. The PCB and mercury concentrations in the most-contaminated eggs are of a magnitude associated with embryotoxic effects in other species (Hoffman *et al*. 1996; Thompson 1996). Whether these contaminants cause embryotoxicity in merlins is unknown. However, if embryotoxicity does occur in some eggs, the scale of these effects has clearly not been sufficient to prevent the population recovery of this species that accompanied the decline in organochlorine pesticide contamination.

Table 4.2: Geometric mean pollutant levels and arithmetic mean shell indices for merlin eggs from across Britain for two different periods. Organochlorine levels expressed as µg/glipid wt and mercury levels as µg/g dry wt. The number of clutches represented at one egg per clutch is indicated by N. Statistical significance of the difference between pre 1987 and post 1986 concentrations (student t-tests on log data) is indicated as *P<0.05, **P<0.01, ***P<0.001

	Pre-1987			Post-1986			
Compound	**N**	**geometric mean**	**range of one geometric SE**	**n**	**geometric mean[2]**	**range of one geometric SE**	**% change**
DDE	261	92.64	86.53 - 99.19	453	53.9	51.8 - 56.1	-41.9 ***
HEOD	261	6.09	5.71 - 6.51	453	3.38	3.20 - 3.57	-44.5 ***
PCB	259	58.99	55.01 - 63.21	453	56.1	53.6 - 58.7	-5.0 ns
Hg	176	1.75	1.57 - 1.95	456	2.70	2.62 - 2.78	54.3 ***
Shell index[1]	283	1.09	1.08 ± 1.09	642	1.14	1.13 ± 1.14	4.69 ***

ns=not significant

[1] arithmetic mean and standard error

[2] post-1986 geometric means expressed on a wet wt basis for DDE, HEOD and PCBs were 3.11, 0.161 and 3.24 µg/g wet wt, respectively

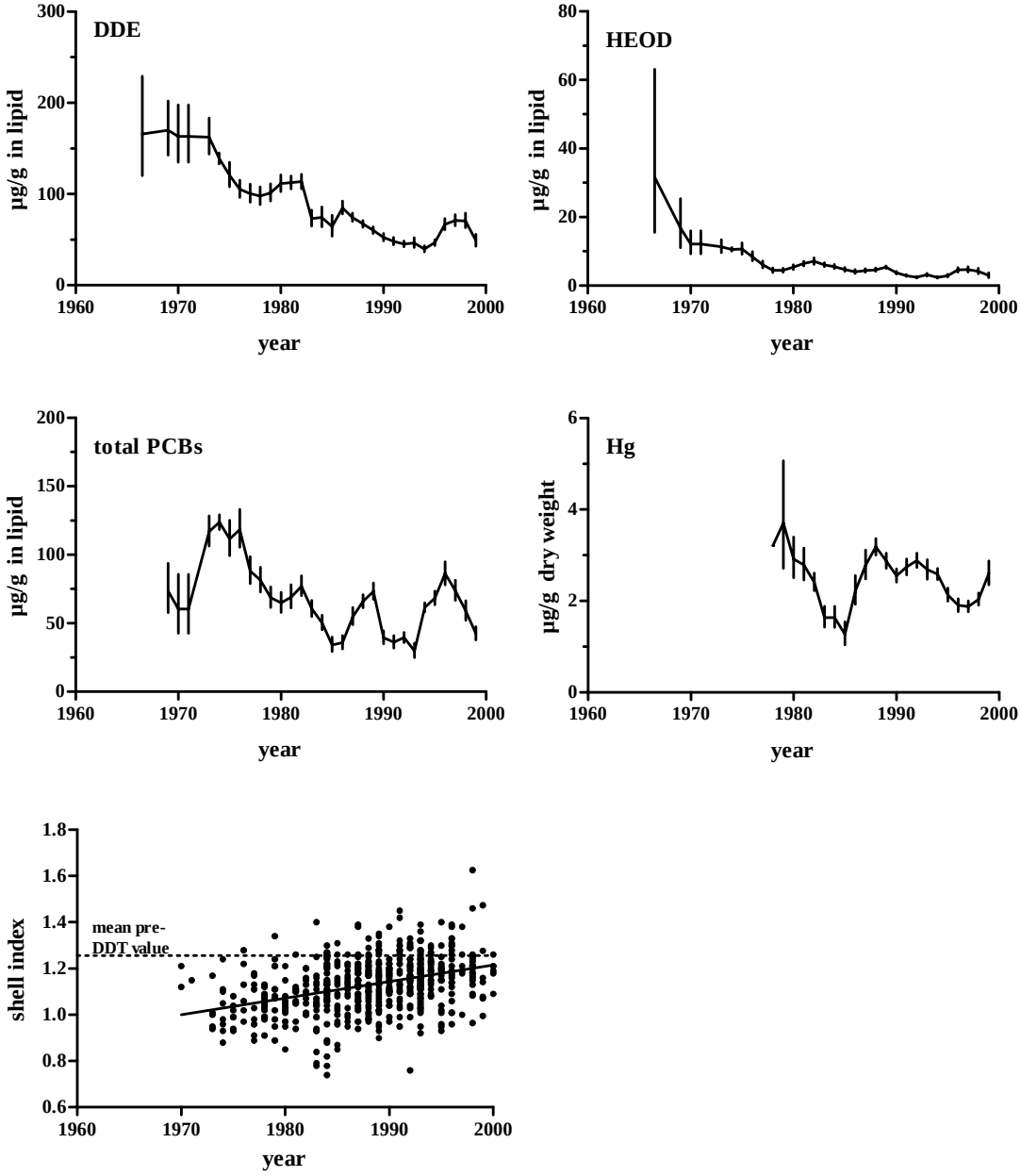

Figure 4.1: Long-term trends in pollutant residues (1967-99) and shell indices (1970–2000) in merlin eggs. Data for contaminant concentrations are three-year moving geometric means with one geometric standard error on either side. Data for shell indices are values for individual eggs and the linear regression is the value for individual shell indices against year.

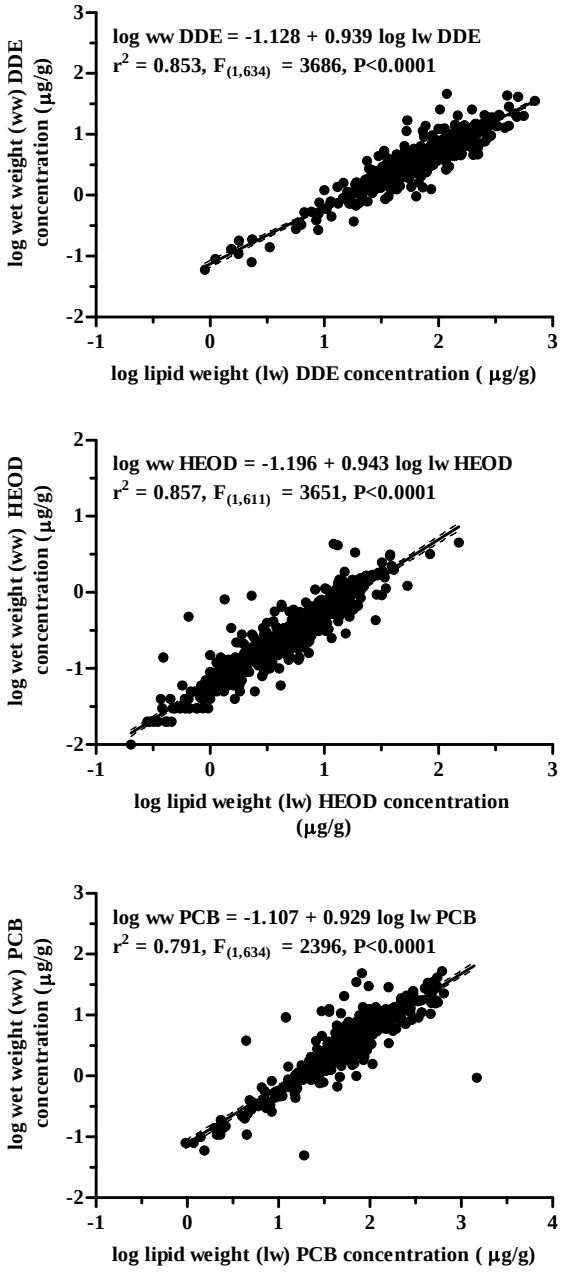

Figure 4.2: Relationship between residues expressed on alipid wt and a wet wt basis for DDE, HEOD and PCBs in merlin eggs collected between 1967 and 2000.

5 Organochlorines and mercury in golden eagle *Aquila chrysaetos* eggs

5.1 Introduction

The findings from analyses of golden eagle eggs obtained during 1963-86 were reviewed by Newton & Galbraith (1991), and long-term trends in residues analysed between 1963 and 1997 were summarised by Newton *et al*. (1998). Failed eggs were collected from seven clutches in 1998 and four clutches in 1999, and a single egg from each clutch was analysed (Newton *et al*. 2000, Shore *et al*. 2002). Single failed eggs from eight clutches were analysed in 2000 for organochlorines and mercury, and are reported in Table 5.1. The long-term trends reported previously by Newton *et al*. (1998) are updated (Tables 5.2-5.4 and Figures 5.1-5.3) to include the more recent analyses of eggs.

5.2 Results for eggs received in 2000

The concentrations of organochlorine pesticides, PCBs and mercury in all eight eggs were relatively low and within the range of low-level contamination values recorded in golden eagle eggs in recent years. None of the residues detected were of a magnitude that might be expected to be associated with embryotoxic effects.

Table 5.1: Residue levels (organochlorine µg/g wet wt (lipid weight), mercury µg/g dry wt) and shell indices (SI) for golden eagle eggs received in 2000

Number	Year	County	SI	pp'-DDE		HEOD		PCB		Hg
Western Isles										
E7739	2000	North Uist	3.31	0.041	(0.713)	0.008	(0.135)	0.892	(15.425)	0.806
Central & Eastern Highlands										
E7705	1999	Grampian	2.82	0.019	(2.495)	ND	(ND)	0.630	(83.143)	ND
E7828	2000	Grampian	3.37	0.013	(0.180)	0.005	(0.069)	2.383	(75.538)	ND
E7741	2000	Highland	3.78	0.012	(0.209)	ND	(ND)	1.848	(31.040)	ND
E7762	2000	Highland	3.54	0.115	(2.045)	0.010	(0.177)	2.648	(49.935)	0.319
E7744	2000	Tayside	3.29	0.035	(0.766)	ND	(ND)	0.430	(9.510)	ND
E7851	2000	Tayside	3.15	ND	(ND)	ND	(ND)	0.079	(1.482)	ND
Northern England										
E7729	2000	Cumbria	3.00	0.031	(0.836)	0.011	(0.298)	0.182	(9.510)	ND

ND = Not detected.

5.3 Long-term trends in golden eagle eggs

To date, 358 golden eagle eggs have been examined for contaminants. These came from regions A and B as described by Dennis *et al*. (1984) in eastern Scotland, and regions C-H, which covers western Scotland but also includes the English Lake District. Trends in residues in eggs from western and eastern areas were considered separately because regional variation

in exposure of golden eagles to organochlorine pesticides has been noted previously (Lockie *et al*. 1969). This was thought to be due to sheep comprising a greater proportion of the diet of golden eagles in hilly western Scotland where sheep are abundant; the use of DDT and dieldrin in sheep-dips up to the mid-1960s was probably one of the most important routes of exposure of eagles to these compounds (Newton & Galbraith 1991 and references therein). Trends in residues in eggs from coastal and inland territories have also been considered separately where possible (birds from western Scotland), as seabirds can accumulate high levels of mercury (Thompson 1996) and are likely to be a more important component of the diet in coastal than inland-nesting eagles. For the purposes of this analysis, coastal territories have been defined as those known to border the sea, or where such information was lacking, as sites within 3 km of the coast.

The data on long-term trends in residues in golden eagle eggs are shown as plots of the three-year moving geometric mean concentrations and shell indices are shown as individual values for eggs plotted against year. The data are presented separately for eggs from eastern inland, western inland and western coastal regions (Figures 5.1-5.3). The regression analyses for the individual log-transformed (contaminant residues) or arithmetic (shell indices) data against year are given in Table 5.2 and the mean contaminant concentrations and shell indices for eggs from the different regions in different time periods are given in Tables 5.3 and 5.4. The data are split at two time-points; the first (1967) corresponds to the start of the voluntary ban on the use of dieldrin in sheep-dip, the second (1987) to the mandatory ban on DDT, aldrin and dieldrin in Britain. All eggs analysed were from separate clutches.

Concentrations of DDE and HEOD in golden eagle eggs were usually below 1.00 µg/g and 0.5 µg/g wet wt (Figures 5.1-5.3). The top 5% of DDE and HEOD residues in eggs (all regions pooled) ranged between 2.2 µg/g and 7.8 µg/g wet wt (DDE) and between 1.0 µg/g and 6.9 µg/g wet wt (HEOD). Almost all these were in eggs collected before 1983, and residues of both compounds have declined significantly in all three regions during the period in which monitoring has been carried out (Tables 5.2, 5.3 and Figures 5.1-5.3). This decline is associated with the ban on the agricultural use (including use in sheep-dip) of these chemicals. Current concentrations in golden eagle eggs are low, and unlikely to be of toxicological significance.

DDE is a major causative agent of eggshell-thinning, and the decline in DDE levels in eastern Scotland coincided with a significant improvement in eggshell indices in that region (Table 5.4 and Figure 5.3). However, it is doubtful DDE was a significant factor in this instance. This is because DDE levels in eagle eggs were generally too low to have caused marked eggshell-thinning and breakage. Furthermore, DDE levels similarly declined in golden eagle eggs in the other Scottish regions (Tables 5.2 and 5.3) but this was not associated with any significant increase in eggshell index. The underlying cause(s) for the increase in shell indices in birds from eastern Scotland is uncertain. Currently, golden eagle eggshell indices do not differ significantly between the three regions.

Total PCBs have been monitored in golden eagle eggs only since 1970, and changes in concentrations over time have not shown the clear pattern of decline seen for the organochlorine pesticides. The only significant long-term decline has been in eggs for birds from inland western Scotland (Table 5.2). PCB concentrations have generally been higher in the eggs of coastal than inland birds (Figures 5.1-5.3 and Table 5.3). Total PCB concentrations of between 8 and 25 µg/g wet wt have been associated with bill deformities and decreased hatching success in a range of avian species, including some raptors (Hoffman

et al. 1996). Approximately 5% (n=13) of all the golden eagle eggs analysed had concentrations within or exceeding this range. Almost all of these eggs were from coastal-nesting birds. Whether golden eagles are as sensitive to PCBs as species that have been tested is unknown, and so it is uncertain whether the relatively high PCBs residues detected in some eggs may have contributed to their failure.

As with the organochlorines, mercury concentrations in eggs were generally higher in eggs from the west coast than elsewhere and levels in eggs from inland birds were higher in the west than the east (Table 5.3). There were no significant long-term changes in mercury residues in western coastal and eastern inland eggs. Mercury concentrations over the whole of the monitoring period have actually risen significantly in eggs from western inland areas (Table 5.2), although average residue levels have fallen progressively in recent years (Figure 5.2). The magnitude of residues in eggs from any region was relatively low, and below that associated with embryotoxic effects.

The overall greater level of contamination in birds from western compared with eastern Scotland and in coastal compared with inland birds is most likely to be explained by regional variations in diet (see Newton & Galbraith 1991). The diet of golden eagles in eastern areas is generally uncontaminated whereas birds from western areas take a wider range of prey, including seabirds that often contain high concentrations of organochlorines and mercury. Although breeding success in golden eagles has been poorer in western than eastern Scotland, this is more likely to have been due to the availability and quality of food supply rather than any direct effect of contaminants (Newton & Galbraith 1991). Persecution has also been identified as a major limiting factor in areas where productivity is low (Scottish Raptor Study Group 1998).

5.4 Summary

A total of 358 unhatched golden eagle eggs, each from a different clutch, were analysed between 1963 and 2000. Concentrations of organochlorine pesticides, PCBs and mercury varied regionally and were higher in western than eastern areas, and generally greater in coastal than inland birds in the west. These regional differences most probably reflect variation in the degree of contamination in prey. Concentrations of organochlorine pesticides have declined significantly during the monitoring period, whereas there has been no overall pattern of change in PCB and mercury concentrations or shell indices for eagles from throughout Scotland. The current levels of contamination in eagle eggs are generally unlikely to be directly embryotoxic although PCB concentrations in some eggs may result in adverse effects.

Table 5.2: Trends in pollutant levels in golden eagle eggs as revealed by regression analyses of individual (log_{10}) residue levels against year. Data are broken down into three geographical regions. N=number of clutches represented at one egg per clutch, b=regression coefficient slope and the significance of the linear regression analysis is indicated as: *P<0.05, **P<0.01, ***P<0.001.

	DDE			HEOD			PCB			Hg			Shell Index		
	N	**b**		**N**	**b**		**N**	**b**		**N**	**b**		**N**	**b**	
W. Scotland Coastal	118	-0.022	***	118	-0.041	***	88	0.003	ns	44	0.023	ns	62	0.000	ns
W. Scotland Inland	182	-0.023	***	182	-0.031	***	131	-0.011	*	81	0.042	***	105	0.002	ns
E. Scotland	58	-0.032	***	58	0.016	*	47	-0.007	ns	34	-0.021	ns	32	0.014	**
All areas	358	-0.028	***	358	-0.033	***	266	-0.012	*	159	0.013	ns	199	0.004	ns

ns=not significant.

Table 5.3: Geometric mean (GM) pollutant levels for golden eagle eggs from various regions of Britain in three different periods. DDE, HEOD and PCB concentrations are expressed in µg/g wet wt, mercury in µg/g dry wt. The number of clutches represented at one egg per clutch is indicated by N. The significance of differences in contaminant levels between regions within time periods is indicated by the F statistic and its associated probability value. The significance of differences between residue levels in eggs collected after 1986 and those in eggs collected in either 1963-66 or 1967-86 (as tested by Tukey pairwise comparison post-hoc tests following ANOVA) is indicated by: *P<0.05, **P<0.01, ***P<0.001. Statistical analyses were carried out on log-transformed data. ns=not significant.

	1963-66				1967-86				Post-1986			
	N	**GM**	**Range for one geometric SE**		**N**	**GM**	**Range for one geometric SE**		**N**	**GM**	**Range for one geometric SE**	
DDE												
W. Scotland coastal	19	0.792	0.626 – 1.003	***	77	0.302	0.256 – 0.355	**	22	0.102	0.073 – 0.142	
W. Scotland inland	32	0.245	0.188 – 0.319	***	92	0.123	0.110 – 0.138	***	58	0.045	0.038 – 0.054	
E. Scotland	3	0.093	0.043 – 0.199	ns	32	0.054	0.043 – 0.067	***	23	0.010	0.008 – 0.013	
ANOVA	$F_{2,51}$=6.15; P=0.004				$F_{2,198}$=16.27; P<0.001				$F_{2,100}$=16.25; P<0.001			
HEOD												
W. Scotland coastal	19	0.695	0.570 – 0.847	***	77	0.081	0.070 – 0.095	**	22	0.033	0.028 – 0.038	
W. Scotland inland	32	0.416	0.331 – 0.524	***	92	0.060	0.053 – 0.066	**	58	0.028	0.024 – 0.034	
E. Scotland	3	0.058	0.034 – 0.099	ns	32	0.023	0.019 – 0.026	ns	23	0.010	0.008 – 0.013	
ANOVA	$F_{2,51}$=6.14; P=0.004				$F_{2,198}$=10.28; P<0.001				$F_{2,100}$=1.36; P=0.260			

	1963-66				1967-86				Post-1986		
	N	**GM**	**Range for one geometric SE**		**N**	**GM**	**Range for one geometric SE**		**N**	**GM**	**Range for one geometric SE**
PCBs											
W. Scotland coastal	-	-	-		66	1.365	1.112 – 1.678	ns	22	1.117	0.784 – 1.592
W. Scotland inland	-	-	-		73	0.961	0.841 – 1.098	**	58	0.448	0.365 – 0.551
E. Scotland	-	-	-		24	0.258	0.174 – 0.383	ns	23	0.134	0.089 – 0.200
ANOVA						$F_{2,160}$=10.91; P<0.001				$F_{2,100}$=9.17; P<0.001	
mercury											
W. Scotland coastal	1	0.010	-		21	0.083	0.055 – 0.125	ns	22	0.103	0.054- 0.195
W. Scotland inland	-	-	-		23	0.014	0.011 – 0.017	*	58	0.045	0.032 – 0.063
E. Scotland	-	-	-		11	0.011	0.010 – 0.013	ns	23	0.008	0.005 – 0.013
ANOVA						$F_{2,52}$=12.12; P<0.001				$F_{2,100}$=5.92; P=0.004	
Sum regions											
				**							
DDE	54	0.351	0.288 – 0.428	*	201	0.152	0.138 – 0.168	***	103	0.038	0.033 – 0.045
HEOD	54	0.447	0.376 – 0.530	**	201	0.058	0.053 – 0.063	***	103	0.026	0.023 – 0.030
PCB	-	-	-		163	0.913	0.806 – 1.034	***	103	0.416	0.348 – 0.497
Hg	1	0.010	-		55	0.026	0.021 – 0.033	ns	103	0.037	0.028 – 0.048

Table 5.4: Arithmetic mean shell indices for golden eagle eggs from various regions of Britain in two different periods. The number of clutches represented at one egg per clutch is indicated by N. The significance of differences in eggshell indices between regions within time periods is indicated by the F statistic and its associated probability value. The significance of differences between eggshell indices of eggs collected after 1986 and those of eggs collected in either 1963-66 or 1967-86 (as tested by Tukey pairwise comparison post-hoc tests following ANOVA) is indicated by: *P<0.05, **P<0.01, ***P<0.001. ns=not significant.

	1963-66			1967-86				Post-1986		
	n	**mean**	**Range for one SE**	**n**	**mean**	**Range for one SE**		**n**	**mean**	**Range for one SE**
W. Scotland coastal	-	-	-	43	3.052	2.995 – 3.109	ns	19	3.052	2.966 –3.139
W. Scotland inland				52	3.070	3.034 – 3.105	ns	53	3.108	3.068 –3.148
E. Scotland				16	2.969	2.908 – 3.031	**	16	3.221	3.156 –3.287
ANOVA					$F_{2,108}$=0.67; P=0.513				$F_{2,85}$=1.36; P=0.263	
All regions	-	-	-	111	3.048	3.019 – 3.077	ns	88	3.117	3.084 –3.150

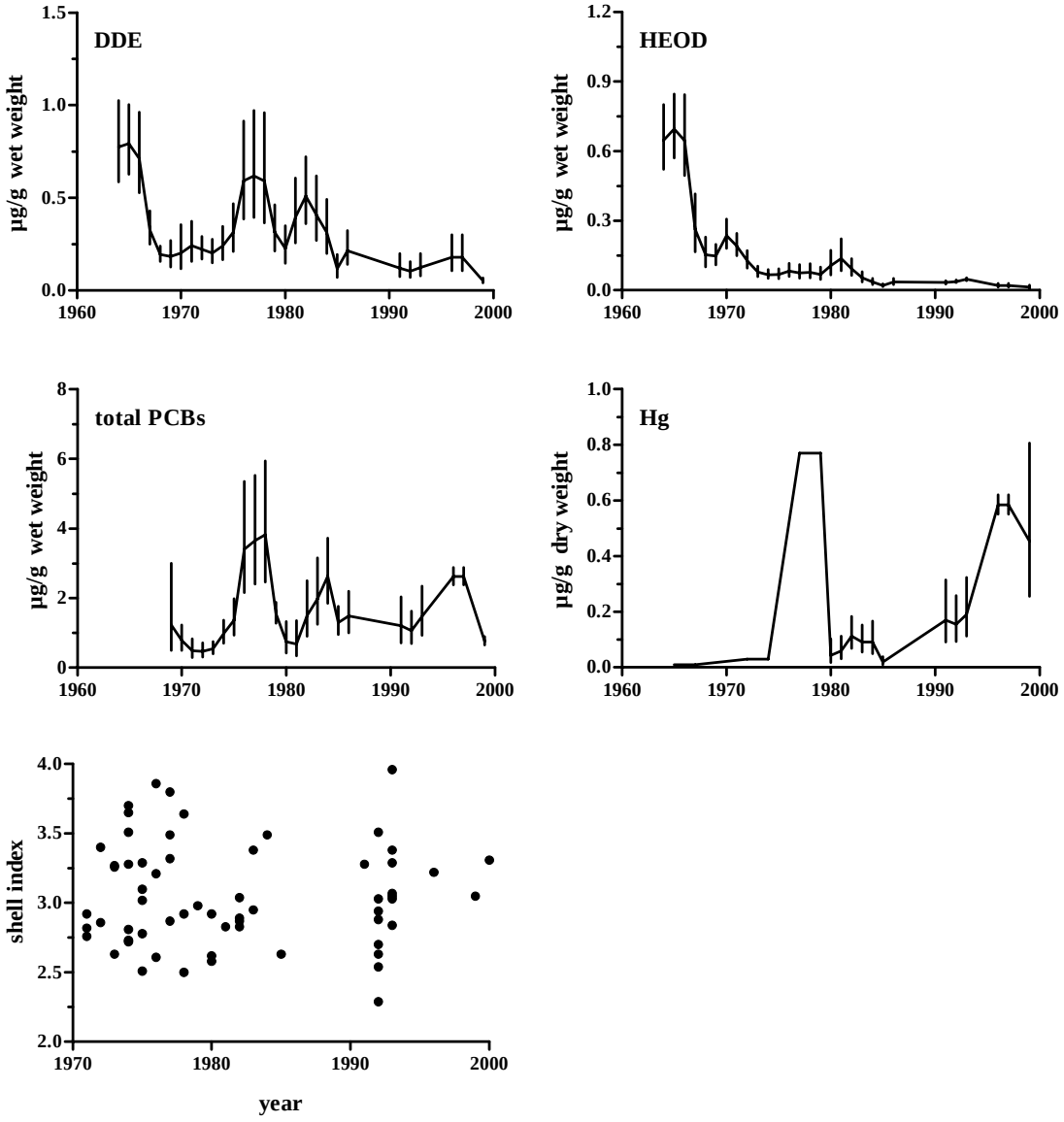

Figure 5.1: Trends in pollutant residues and shell indices in golden eagle eggs from coastal districts of western Scotland, 1963-97. Data are three-year moving geometric mean with one geometric standard error on either side for contaminant concentrations and shell indices for individual eggs.

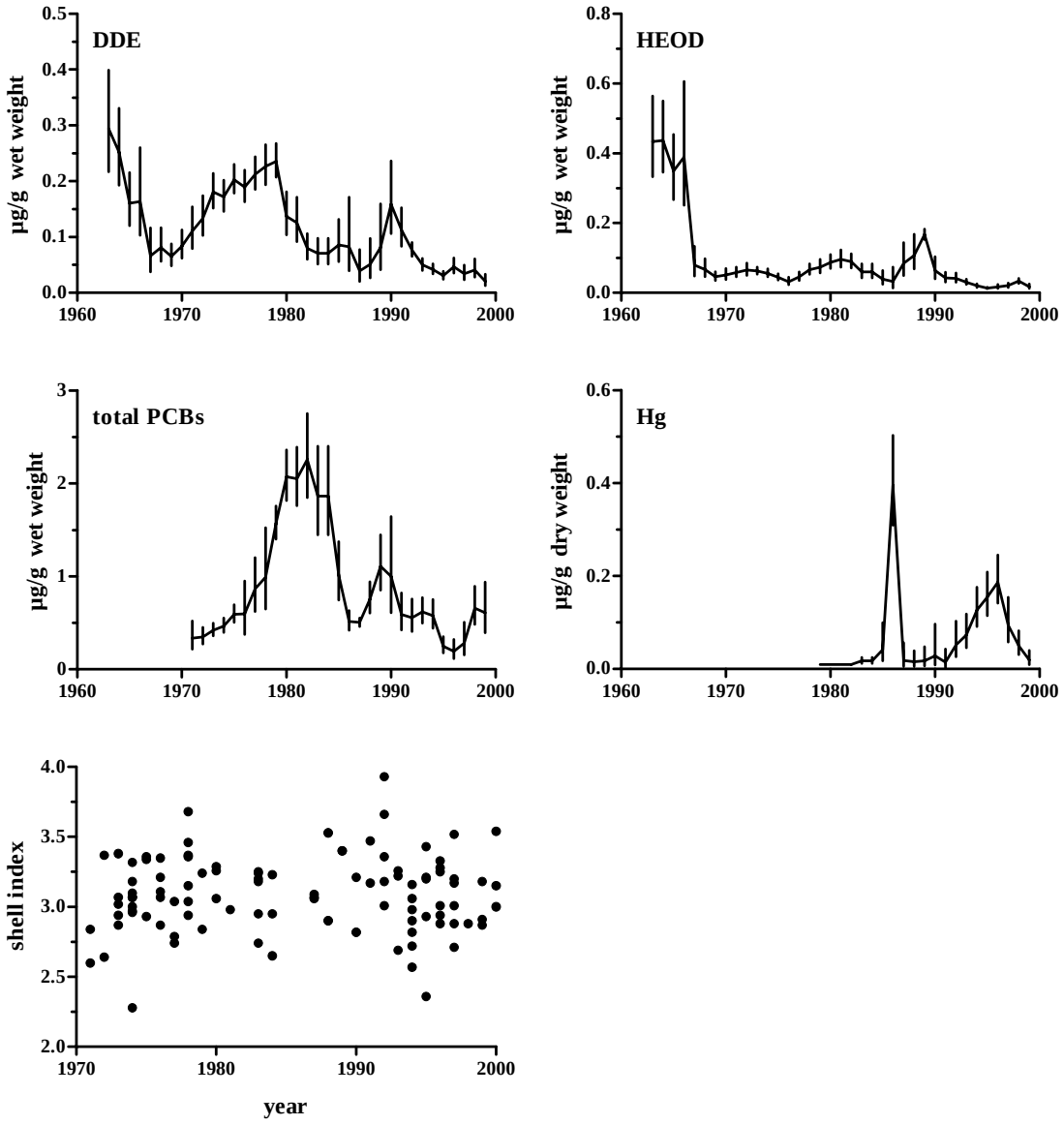

Figure 5.2: Trends in pollutant residues and shell indices in golden eagle eggs from inland districts of western Scotland, 1963-97. Data are three-year moving geometric mean with one geometric standard error on either side for contaminant concentrations and shell indices for individual eggs.

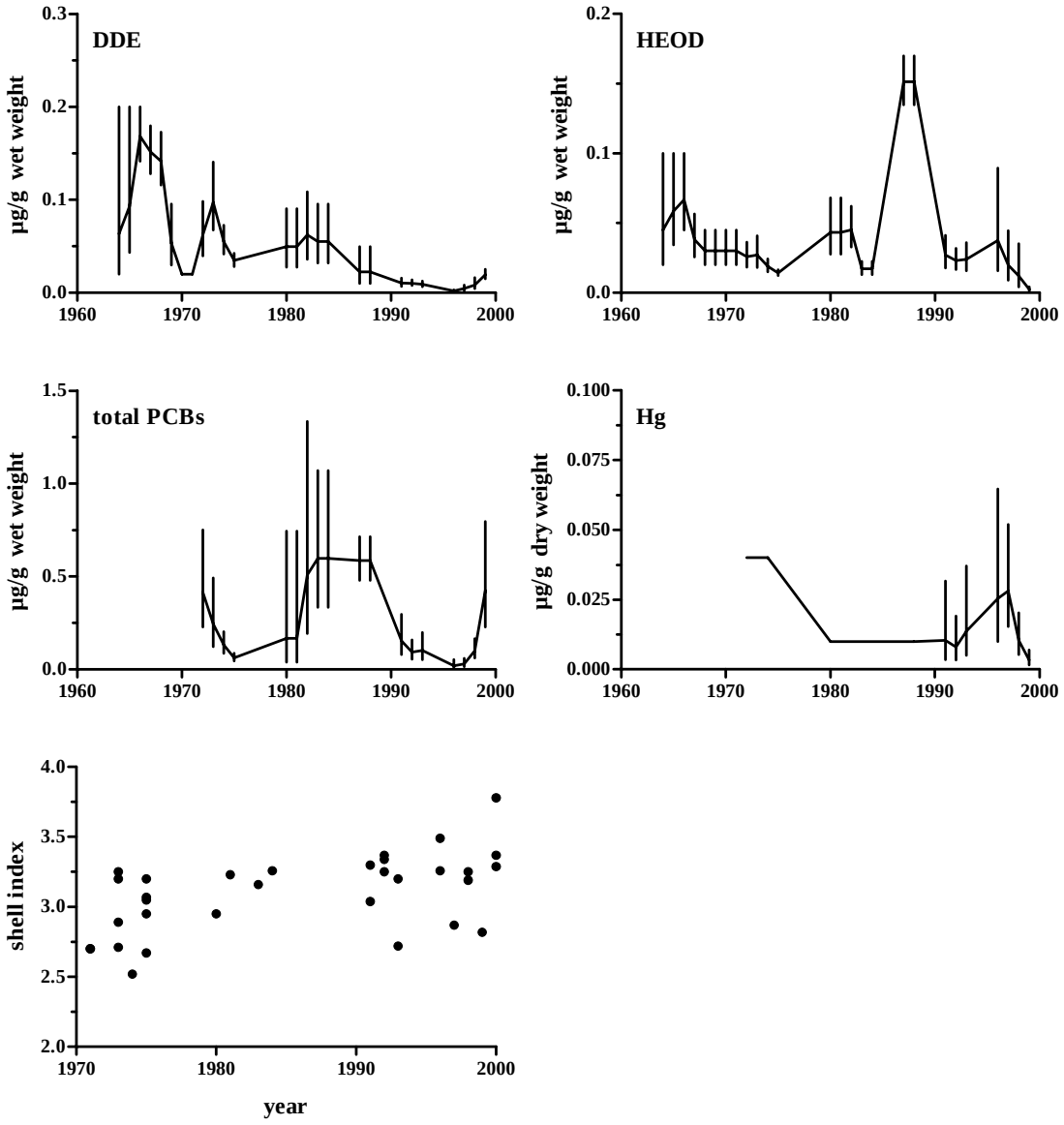

Figure 5.3: Trends in pollutant residues and shell indices in golden eagle eggs from eastern Scotland, 1963-97. Data are three-year moving geometric mean with one geometric standard error on either side for contaminant concentrations and shell indices for individual eggs.

6 Organochlorines and mercury in northern gannet *Morus bassanus* eggs

6.1 Introduction

Contaminant levels in gannet eggs have been monitored by collecting eggs mainly from two colonies (Ailsa Craig in the Firth of Clyde and Bass Rock in the Firth of Forth) every 1-2 years. Eggs have also been collected from other colonies when the opportunity arose. Eggs are normally collected during visits to colonies made during laying or the early incubation period, and about ten eggs are taken from each colony.

Long-term trends in contaminant levels in the gannet eggs examined up to 1988 have been reported by Newton *et al.* (1990a) and trends up to 1998 have been described in the 1998/99 report in the present series (Newton *et al*. 2000). Newton *et al*. (1990a) discussed the relationships between DDE levels and eggshell features in British and Irish gannets and also concluded that the concentrations of organochlorines and mercury were too low to cause reductions in overall breeding success in colonies. This conclusion still stands and is not discussed further here. The aim here is to report the current levels of contamination in gannet eggs and update the data on long-term trends by incorporating the data for eggs analysed since the last review. In general, the analysis of gannet eggs gives an indication of long-term trends in the levels of contamination in gannet prey, and can be used as an index of changes in contamination of the wider marine environment.

6.2 Analysis of northern gannet eggs received in 2000 at CEH

Ten gannet eggs were collected from Ailsa Craig in 2000 but not from any other colonies. The contaminant concentrations and eggshell indices are given in Table 6.1. Residue levels were generally low and within the range of concentrations detected in eggs previously from this colony. However, both total PCB and mercury concentrations were slightly but significantly (student t-test, $P<0.05$ in both cases) higher than in 1998, the last time eggs were collected from this colony. There was no significant difference between the two years in DDE and HEOD concentrations in the eggs or in the eggshell index.

6.3 Long-term trends

The significance of long-term trends in contaminant residues is given for the seven colonies for which time series data are available (Table 6.2).

DDE levels declined significantly in all but one colony. The exception was at Little Skellig from where eggs were collected in two years only (1973 and 1988). DDE concentrations in eggs from here did, in fact, decline between the two collection periods (geometric mean concentrations in eggs in 1973 and 1988 were 1.247 and 0.568 µg/g wet wt, respectively), but the difference was not statistically significant. There was also a general downward trend in HEOD concentrations at most colonies but it was only statistically significant at Ailsa Craig and Bass Rock. In contrast to the other colonies, HEOD concentrations significantly increased at Grassholm, but eggs were only sampled in two years and are not necessarily indicative of long-term change. The long-term declines in organochlorine pesticide concentrations in

gannet eggs may reflect a reduction of inputs of these compounds into sea water following restrictions and bans on their use. Declines in concentrations of DDE and HEOD in gannet eggs have also been observed elsewhere (Fimreite *et al*. 1982; Chapdelaine *et al*. 1987; Elliott *et al*. 1988).

Long-term trends in PCB and mercury concentrations were much more variable than the organochlorine pesticides, and showed no overall consistent pattern. Concentrations of both contaminants increased at some colonies but decreased at others; these trends were only sometimes statistically significant (Table 6.2). The direction of change in concentrations of both contaminants was usually the same at any one particular colony.

The long-term trends that are probably the most meaningful are the ones from Ailsa Craig (Figure 6.1) and Bass Rock (Figure 6.2) because these colonies that have been sampled for the longest time period and the most frequently. The long-term decline in DDE and HEOD but more variable nature of changes in PCBs and Hg is apparent at these two sites. Eggshell indices have not changed significantly at Bass Rock over the whole monitoring period but have increased significantly at Ailsa Craig ($F_{(1,156)}$=12.44, P<0.001). This was associated with the decline in contaminant levels and eggshell-thinning in gannets has been largely attributed to DDE (Newton *et al.* 1990a).

6.4 Comparisons of colonies

Examination of the trends for individual compounds in eggs from each of the four gannet colonies indicated that linear regression models gave significant fits to the log-transformed data and most frequently provided the best fit compared with any other single regression model. Thus, linear regression analysis of log-transformed data was used to make inter-colony comparisons of long-term trends in contaminant residues at Ailsa Craig, Bass Rock, St Kilda and Hermaness, the four colonies that have been sampled most frequently (Figure 6.3).

DDE and HEOD concentrations in eggs declined over time at all four colonies but the rate of decline varied significantly between colonies ($F_{(3,479)}$=8.56 for DDE, $F_{(3,454)}$=6.23 for HEOD, P<0.001 in both cases), concentrations in the 1970s of both compounds tending to be higher and the subsequent rate of decline faster at Ailsa Craig than elsewhere. The rates of change in PCB and mercury residues have also varied significantly between colonies ($F_{(3,480)}$=8.17 for PCBs, $F_{(3,488)}$=49.7 for mercury, P<0.001 in both cases). Concentrations of both contaminants in eggs during the 1970s and early 1980s were higher at Ailsa Craig than elsewhere but have declined since 1970, whereas concentrations in eggs from the other colonies have all increased, although these increases have not always been significant at each colony (Table 6.2 and Figure 6.3).

Variation between colonies in the magnitude and long-term trends in contaminant concentrations may reflect differences in the proximity of colonies and feeding areas to sources of contamination and differences between colonies in prey taken.

6.5 Summary

Since monitoring was started, eggs have been obtained in 1-18 years from eight different gannet colonies around Britain and Ireland. Four of these colonies have been sampled in at

least eight different years, and the colonies at Ailsa Craig and Bass Rock have been studied most intensively. Over the whole monitoring period, DDE and HEOD concentrations in eggs have generally declined whereas long-term trends in PCBs and mercury have been more variable, concentrations decreasing at some colonies but remaining the same or increasing at others.

Table 6.1: Residue levels (organochlorine µg/g wet wt (lipid weight), mercury µg/g dry wt) and shell indices (SI) for gannet eggs received in 2000

Number	Year	SI	pp'-DDE		HEOD		PCB		Hg
Ailsa Craig									
G1054	2000	3.11	1.115	(23.886)	0.036	(0.778)	8.852	(189.669)	4.054
G1055	2000	3.06	0.024	(0.501)	0.022	(0.452)	1.369	(28.376)	2.518
G1056	2000	2.53	0.106	(1.842)	0.102	(1.760)	9.120	(157.895)	1.652
G1057	2000	3.15	0.081	(1.455)	0.029	(0.523)	3.624	(64.708)	3.739
G1058	2000	3.03	0.026	(0.433)	0.022	(0.369)	1.805	(30.072)	2.591
G1059	2000	3.42	0.076	(1.566)	0.040	(0.819)	4.079	(84.430)	3.535
G1060	2000	*	0.035	(0.643)	0.028	(0.511)	2.662	(49.148)	2.302
G1061	2000	*	0.027	(0.720)	0.027	(0.717)	10.233	(269.967)	1.837
G1062	2000	*	0.140	(1.938)	0.085	(1.181)	5.289	(73.370)	2.506
G1063	2000	*	0.072	(1.682)	0.046	(1.084)	2.922	(68.458)	3.022

Table 6.2: Trends or annual differences in residues in eggs from different gannet colonies around Britain and Ireland. Trends examined by regression of individual residue levels against year, or, where only two years of data were available, by a comparison of the geometric mean values for each year, using a t-test. D = decrease, I = increase, ns = no significant trend or difference. *P<0.05, **P<0.01, ***P<0.001.

Colony study	Period of years	No. years	No. eggs	DDE	HEOD	PCB	Hg
Ailsa Craig	1971–2000	18	160	D***	D***	D***	D***
Bass Rock	1973-98	16	191	D***	D**	ns	I***
St Kilda	1979-98	8	66	D***	ns	ns	I***
Hermaness	1980-96	8	74	D***	ns	I***	I***
Grassholm	1980-84	2	20	D*	I*	I***	I***
Little Skellig	1973-88	2	13	ns	ns	I**	ns
Great Saltee	1988	1	31	-	-	-	-
Scar Rocks	1971-83	5	42	D***	ns	D***	D***

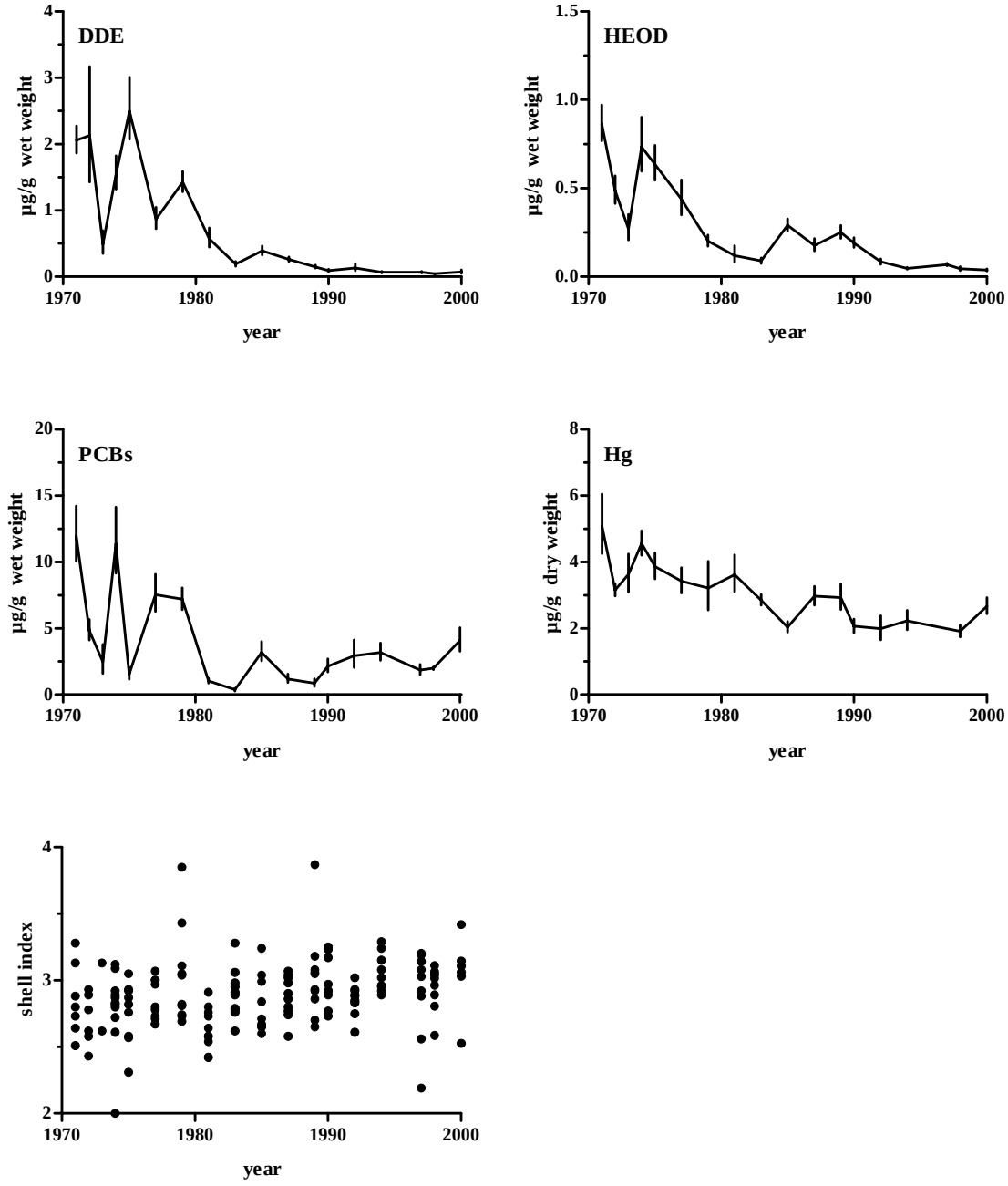

Figure 6.1: Trends in pollutant residues in gannet eggs from Ailsa Craig, 1971–2000. Data are geometric means with one geometric standard error either side for contaminant residues and shell indices for individual eggs.

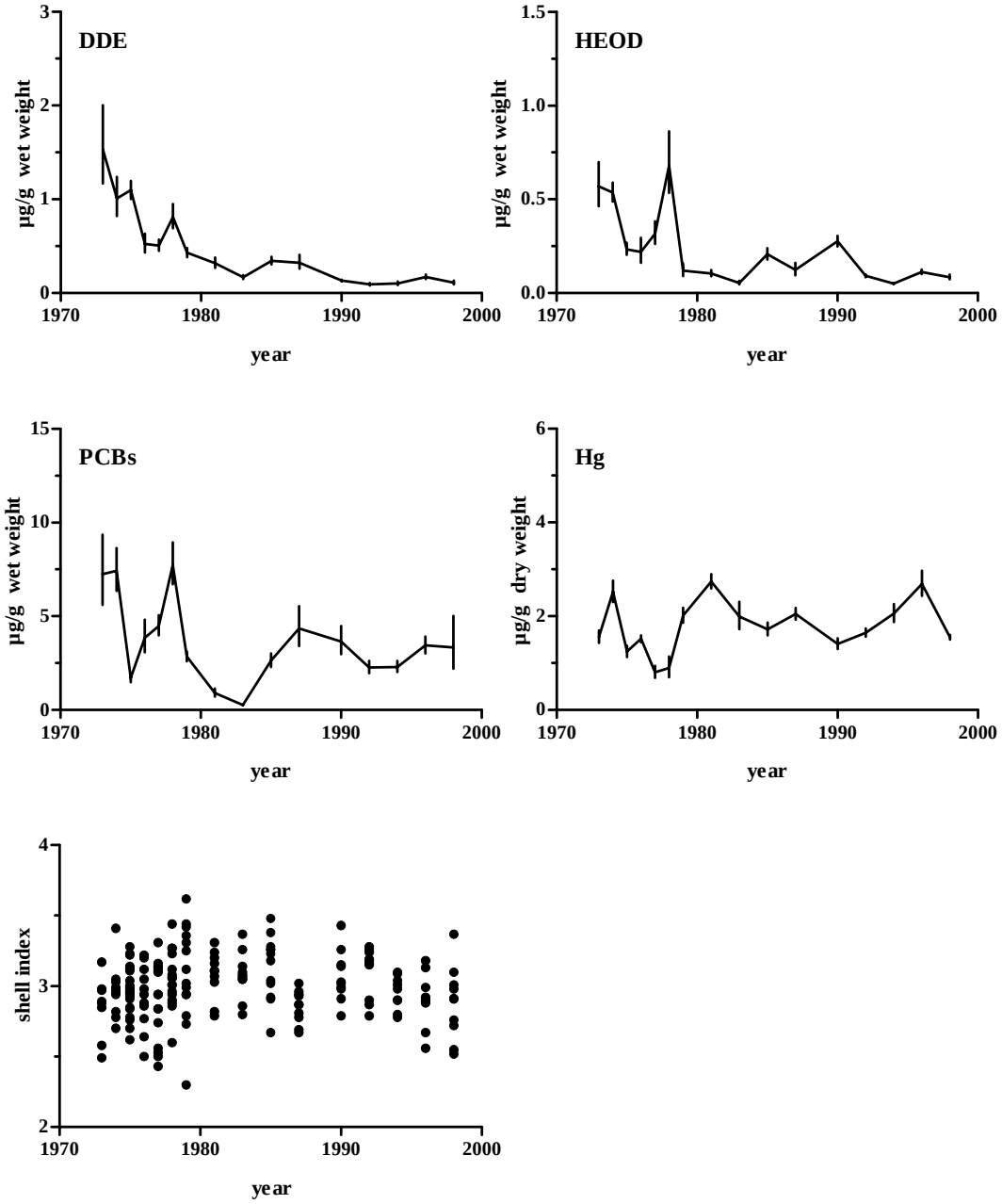

Figure 6.2: Trends in pollutant residues in gannet eggs from Bass Rock, 1971–2000. Data are geometric means with one geometric standard error either side for contaminant residues and shell indices for individual eggs.

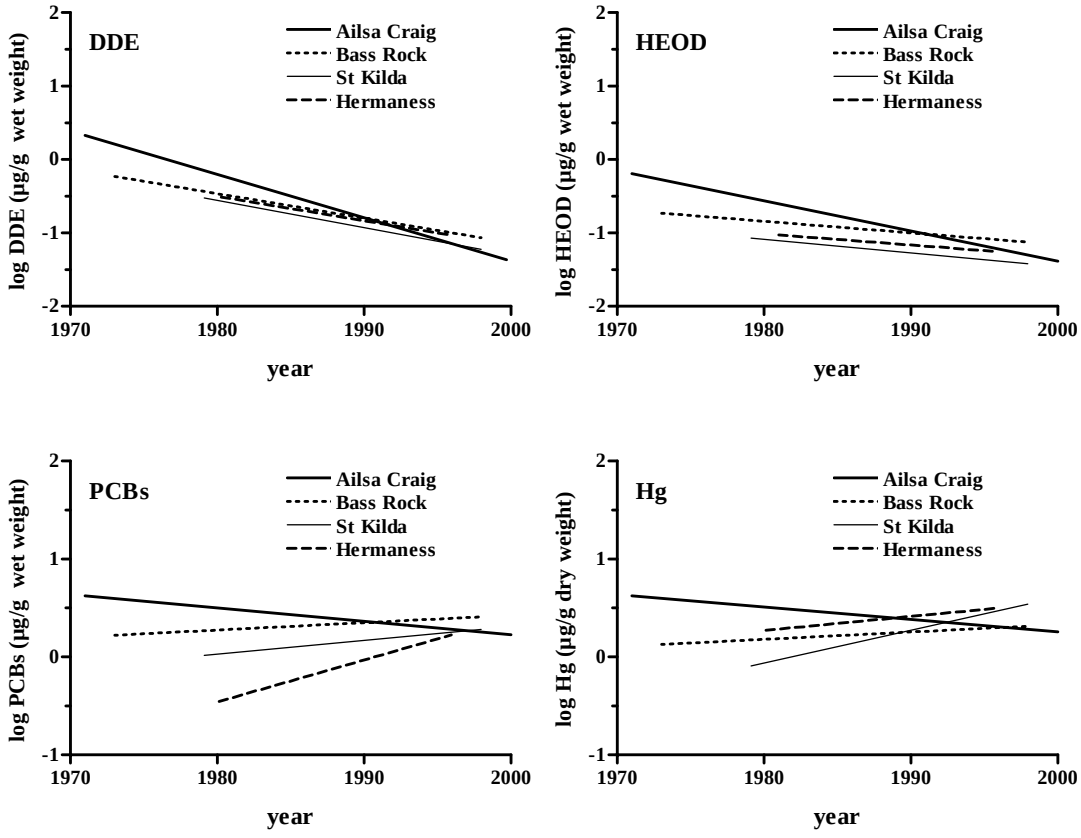

Figure 6.3: Regression analyses of long-term trends in pollutant residues in gannet eggs from Ailsa Craig, Bass Rock, St Kilda and Hermaness. Linear regressions are based on the values for individual eggs against year.

7 Organochlorines and mercury in white-tailed eagle *Haliaeetus albicilla* eggs

7.1 Introduction

Following their reintroduction to western Scotland in 1976-85 white-tailed eagles have had lower breeding success than individuals in some populations in continental Europe, although productivity has been similar to that of birds in Iceland. The relatively poor breeding success of the Scottish population is due to the number of total nest failures, and a few pairs persistently fail to rear young. One potential cause of breeding failure might be exposure to organochlorines and mercury, which the birds could acquire particularly from the marine component of their diet, various fish and seabirds.

Some of the Scottish white-tailed eagles nest on inaccessible sea cliffs, making collection of samples difficult. Two failed eggs were received at CEH in 2000, one of which had been collected from Skye in 1998, the other from Mull in 1999 (Table 7.1). So far, a total of eight eggs has been obtained and analysed during the course of the present monitoring scheme, and the data for all the eggs are given in Table 7.1.

7.2 Results for eggs analysed in 2000 and analysis of long-term data

The two eggs received at CEH in 2000 contained low concentrations of HEOD and mercury (Table 7.1), as found in all the white-tailed eagle eggs analysed in previous years, and low levels of DDE. Both contained relatively high total PCB concentrations, particularly the egg from Skye, although higher wet wt PCB concentrations have been recorded in white-tailed eagle eggs in previous years (Table 7.1). Both the eggs had the lowest shell index values recorded to date for intact white-tailed eagle eggs received at CEH. However, the wet wt PCB concentrations in the eggs were well below that (mean of 105 μg/g wet wt) associated with eggshell-thinning in mallards *Anas platyrynchos* experimentally fed PCBs (Hoffman *et al*. 1996).

In a recent review of DDE and PCB mediated effects on the reproduction of white tailed eagles in Sweden, Helander *et al.* (2002) suggested that lipid DDE concentrations in eggs of 30-50 μg/g and 100-120 μg/g were the lowest observed effect levels (LOELs) for eggshell thickness and productivity, respectively; complete reproductive failure was associated with a DDE concentration of 900 μg/g. However, there was considerable variation in individual productivity over a wide range of residue burdens. The first effects on productivity from PCBs appeared to occur at lipid concentrations of about 300 μg/g, but this may have been a result of co-variation with DDE residues. Because of the strong association between PCB and DDE, firm conclusions about effects levels for PCBs could not be drawn. Of the eight eggs from Scottish white-tailed eagles that have been analysed to date, four had lipid DDE concentrations above the LOEL for eggshell thickness, and two of them exceeded the LOEL for productivity (Table 7.1). Five eggs also had total PCB lipid concentrations greater than 300 μg/g, and all but one had PCB residues of between 8 and 25 μg/g wet wt, the range associated with decreased hatching success in various avian species (Hoffman *et al*. 1996). Although it is not certain that DDE or PCBs were a contributory cause of reproductive failure in white-tailed eagles from Scotland, the residues in some eggs were of a magnitude that might have been expected to have adverse effects.

Furthermore, initial inspection of the data suggests that both DDE and PCB concentrations in the eggs of white-tailed eagles from Scotland are negatively associated with shell index values, but the data are scant. It is clearly important to analyse further eggs that become available.

Table 7.1: Residue levels (organochlorine µg/g wet wt (lipid weight), mercury µg/g dry wt) and shell indices (SI) for all white-tailed white-tailed eagle eggs received at CEH, including two eggs received in 2000.

Year	Location	SI	pp'-DDE		HEOD		PCB		Hg
1986	Mull		29.27	(313.01)	8.07	(86.27)	32.19	(344.21)	0.56
1990	Mull		2.32	(73.44)	1.77	(56.20)	14.73	(467.02)	ND
1991	Mull	4.15	0.31	(13.00)	0.02	(0.67)	0.15	(6.14)	0.46
1994	-	3.50	0.79	(25.47)	0.02	(0.73)	10.90	(349.69)	0.34
1997	Western Isles		20.61	(204.99)	0.70	(6.85)	132.96	(1302.94)	0.36
1998	Mull	3.33	0.89	(17.5)	0.03	(0.53)	12.65	(247.93)	0.11
1998	Skye	2.55	2.64	(78.08)	0.02	(0.66)	28.95	(856.97)	0.64
1999	Mull	2.90	0.18	(4.40)	0.02	(0.48)	11.33	(279.56)	0.55

8 Rodenticide residues in barn owls *Tyto alba*

8.1 Introduction

The aim of this work was to screen barn owl carcasses for residues of 'second-generation' rodenticides. The carcasses, supplied by members of the public, included birds that had died from various causes, mainly accidents. The compounds of interest included difenacoum, bromadiolone, brodifacoum and flocoumafen. The findings from all barn owls analysed in previous years were given in Newton *et al.* (1997, 1999b), and in previous reports in the present series, while those from 53 birds sent to CEH in 2000 are given in Table 8.1. The latest full review of long-term trends in rodenticide residues in barn owls is given in Shore *et al*. (2005).

8.2 Methods

Analysis of rodenticides in liver tissue was carried out by the same methods as in previous reports and described by Newton *et al.* (1990b), but using new HPLC and detection equipment (Hewlett-Packard LC-MS Series 1100) first employed to analyse birds collected in 1998 (Newton *et al*. 2000). Quantification was carried out on the basis of peak height calibrated against heights for analytical standards.

The analytical standards used in the monitoring scheme have been made up from technical grade material originally supplied to CEH by rodenticide manufacturers. These CEH standards were compared with those that have, in more recent years, become commercially available (Greyhound Chromatography and Allied Chemicals, Birkenhead, Merseyside, UK). This comparison indicated that there was no difference between the two standards in the response obtained for either bromadiolone or brodifacoum, but that the CEH standard gave a significantly higher response for difenacoum but lower response for flocoumafen (Figure 8.1). The consequence of this is that the liver concentrations in owls reported in previous years were lower for difenacoum and higher for flocoumafen than they would have been if they had been quantified using the Greyhound standard (results would have been the same for bromadiolone and brodifacoum). The effect of using different standards had no significant effect on actual detection limits for difenacoum and flocoumafen however, because the effect was trivial relative to the otherwise relatively small variation that occurs between individual samples (because of slight variations in sample weight) and analytical runs.

For the data in the present report, all analyses were carried out using the commercially-available Greyhound standard. Thus, any comparisons of the magnitude of detected residues of difenacoum and flocoumafen with those in previous years need to take into account variation that is attributable to changes in the standards used to quantify these compounds. Previously-reported detectable difenacoum and flocoumafen liver concentrations in owls can be standardized to values that would have been reported if commercially available Greyhound standards had been used, by multiplying the reported residue value by 1.236 (difenacoum) and 0.641 (flocoumafen).

8.3 Results

Of the 53 barn owls received in 2000, 21 (40%) contained detectable levels of rodenticides (Table 8.1). This was similar to that for birds received at CEH in 1999 (20/54 = 37%) (Shore *et al*. 2002). Of the owls received in 2000, seven were sent in with no information on year of death and it was assumed that they had died in 2000. Eight others had been found dead in 1993 (1), 1996 (1), 1997 (2), 1998 (1) or 1999 (3). Thus, of the owls that died or were assumed to have died in 2000, 42% (19/45) contained rodenticides, although this value may subsequently change slightly, as some birds received at CEH in 2001 and future years may include individuals that died in 2000.

Overall therefore, the proportion of owls received in 2000 that contained detectable levels of rodenticide (40%) was similar to that for 1999 (37%). These data are also consistent with the trend reported for earlier years that suggested the increase since 1983 (when monitoring began) in the proportion of birds exposed was reaching an asymptote of about 40% (Newton *et al*. 2000).

Difenacoum, bromadiolone, brodifacoum and flocoumafen occurred in 14 (26% of the sample), 10 (19%), 2 (3.8%) and 0 (0%) barn owls, respectively. The predominance of difenacoum and lack of flocoumafen is consistent with findings in previous years (Newton *et al.* 1999b, 2000) and in other predators (Shore *et al*. 2003). The proportion of birds that contained bromadiolone and brodifacoum was within the range observed in previous years during the 1990s (Newton *et al*. 2000).

A number of the barn owls had residue levels considered to be in the potentially lethal range. This range has variously been described as >0.1 µg/g (Newton *et al*. 2000) and >0.2 µg/g (Newton *et al.* 1999b) and is so classed on the basis of two criteria of observations. These are that almost all owls diagnosed at post-mortem of having died from rodenticide poisoning (because they typically had multiple haemorrhaging from such organs as the heart, lungs, liver, brain and subcutaneous areas) had liver residues >0.1 µg/g, and, secondly, that owls that had been experimentally poisoned had residues of the range 0.2-1.72 µg/g (see Newton *et al*. 1999b for review). Re-examination of the residue data for poisoned owls so that it is corrected for calibration against commercially-available standards for difenacoum and flocoumafen indicates that >0.1-0.2 µg/g is still the appropriate cut-off to describe the potentially lethal range. Of the barn owls in the 2000 sample, five (9.4% of the sample) had liver residues between 0.1 and 0.2 µg/g and two (3.8%) had a liver residue >0.2 µg/g; this is broadly consistent with findings for previous years. Only one of these birds (13414) showed signs of haemorrhage consistent with clinical sign of poisoning by anticoagulants, had a residue in the potentially lethal range, and is therefore considered likely to have been poisoned by rodenticides. Three other birds (numbers 13224, 13309, 13368) also showed signs of haemorrhage but the circumstances in which the birds was found and post-mortem observations were not characteristic of rodenticide poisoning and the birds did not contain significant residues of second-generation rodenticides. Road traffic accident (13309), collision (13224) and unidentified trauma (13368) were considered to be the causes of death.

Table 8.1: Levels of rodenticides (µg/g wet wt) in the livers of adult (A) and juvenile (J) male (M) and female (F) barn owls received in 2000. Juveniles are birds in first year, adults are birds older than first year; brod=brodifacoum, difen=difenacoum, brom=bromadialone, floc=flocoumafen ND=none detected.

						Rodenticide residue			
Specimen No.	**Date**		**County**	**Age**	**Sex**	**brod**	**difen**	**brom**	**floc**
13129	September	1999	Caernarvonshire	J	F	ND	ND	ND	ND
13130	January	2000	East Norfolk	J	M	ND	ND	ND	ND
13131	November	1999	Cardiganshire	J	M	ND	ND	ND	ND
13133	January	2000	Middlesex	A	F	ND	ND	ND	ND
13139	February	2000	West Norfolk	J	M	ND	ND	ND	ND
13142	February	2000	West Norfolk	J	M	ND	0.055	0.050	ND
13153	January	1996	Dorset	J	M	ND	ND	ND	ND
13155	February	2000	Cambridgeshire	J	M	ND	ND	0.029	ND
13158	February	2000	North Wiltshire	J	M	ND	0.054	ND	ND
13159	February	2000	Roxburghshire	J	M	ND	ND	0.141	ND
13168	April	1993	Pembrokeshire	A	M	ND	ND	ND	ND
13181	February	2000	Herefordshire	A	F	ND	0.050	0.072	ND
13185	February	2000	South Essex	A	M	ND	ND	ND	ND
13192	February	2000	West Norfolk	J	M	ND	0.039	ND	ND
13218	April	2000	Surrey	J	M	ND	0.017	ND	ND
13219	April	2000	Fife	J	M	ND	ND	ND	ND
13223	March	2000	East Sussex	J	F	0.085	ND	ND	ND
13224	April	2000	East Kent	J	M	ND	0.050	ND	ND
13249	June	2000	Cambridgeshire	J	M	ND	ND	0.343	ND
13255	June	2000	Oxfordshire	J	F	ND	ND	ND	ND
13263	July	2000	South Hampshire	J	*	ND	ND	ND	ND
13267	July	2000	East Kent	J	*	ND	ND	ND	ND
13293	August	2000	East Norfolk	J	*	ND	ND	ND	ND
13301	September	2000	Cheshire	J	*	ND	0.174	ND	ND
13304	September	2000	Caernarvonshire	J	M	ND	ND	ND	ND
13307	September	2000	Northamptonshire	J	*	ND	ND	ND	ND
13309	September	2000	West Norfolk	J	M	ND	ND	ND	ND
13314	July	2000	East Cornwall	J	M	ND	ND	ND	ND
13321	August	2000	East Cornwall	J	M	ND	ND	ND	ND
13331	December	1997	West Suffolk	*	F	ND	ND	ND	ND
13332	*		Huntingdonshire	*	F	ND	0.098	ND	ND
13335	March	1998	Huntingdonshire	A	F	ND	ND	0.015	ND
13339	May	2000	West Suffolk	*	F	ND	ND	ND	ND
13340	January	1999	Huntingdonshire	*	M	ND	ND	0.028	ND
13344	*		*	J	F	ND	0.054	ND	ND
13350	*		Herefordshire	A	F	ND	ND	ND	ND
13357	October	2000	Warwickshire	J	M	ND	ND	ND	ND
13363	*		North Essex	*	F	ND	0.089	0.130	ND
13364	October	2000	Kirkudbrightshire	J	F	ND	ND	ND	ND
13367	*		East Cornwall	J	F	ND	0.079	ND	ND
13368	October	2000	South Lincolnshire	J	F	ND	ND	ND	ND
13369	*		*	J	M	0.087	0.011	ND	ND
13376	November	2000	North Aberdeenshire	J	M	ND	ND	ND	ND
13377	November	2000	South-East Yorkshire	J	M	ND	ND	ND	ND
13378	*		West Norfolk	J	F	ND	ND	ND	ND
13386	November	2000	Staffordshire	A	F	ND	0.016	ND	ND

Specimen No.	Date		County	Age	Sex	Rodenticide residue			
						brod	difen	brom	floc
13390	November	2000	West Gloucestershire	J	F	ND	ND	ND	ND
13392	November	2000	North Lincolnshire	A	F	ND	ND	ND	ND
13398	September	1997	Anglesey	J	M	ND	ND	ND	ND
13410	February	2000	East Ross	A	F	ND	ND	ND	ND
13414	November	2000	East Kent	J	M	ND	ND	0.165	ND
13415	November	2000	Shropshire	J	F	ND	ND	ND	ND
13418	December	2000	Huntingdonshire	A	F	ND	0.071	0.036	ND

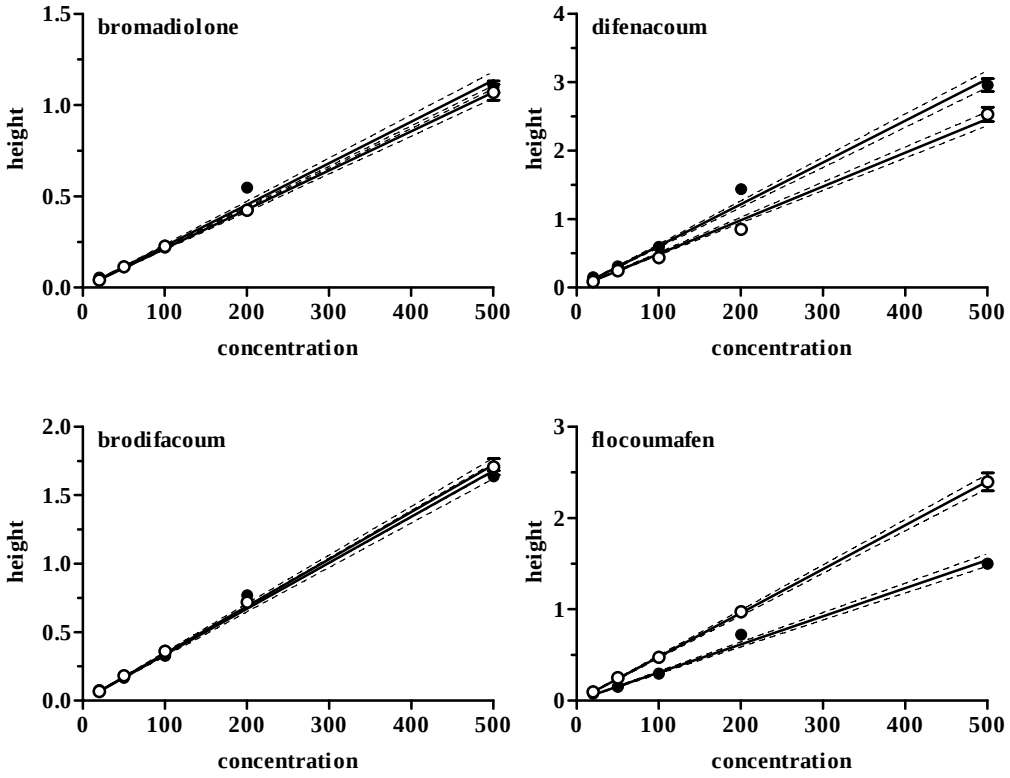

Figure 8.1: Standard curves (concentration of the compound plotted against peak height, hashed lines indicate 95% confidence intervals) for bromadiolone, difenacoum, brodifacoum and flocoumafen using CEH standards (●) and commercially-available standards (○).

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10 Appendix

Abbreviations used in the text:

b	linear regression coefficient
DDE	dichlorodiphenyldichloroethylene
DDT	dichlorodiphenyltrichloroethane
CEH	Centre for Ecology & Hydrology
dry wt	dry weight
g-HCH	gamma-hexachlorocyclohexane
GM	geometric mean
HEOD	hexachloro-epoxy-octahydro-dimethanonaphthalene
Hg	mercury
LOEL	lowest observed effect level
lipid wt	lipid weight
JNCC	Joint Nature Conservation Committee
N	number of samples analysed
ND	not detected
ns	not significant
PBMS	Predatory Bird Monitoring Scheme
PCB	polychlorinated biphenyl
SI	shell index
wet wt	wet weight