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1 Preface and Summary

1.1 Introduction

The Wildlife and Pollution contract covers a long-term monitoring programme, the Predatory Bird Monitoring Scheme (PBMS), 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 subsequently assessed the success of these bans by measuring whether there has been a decline in the concentrations of organochlorine pesticides in the livers and eggs of predatory and freshwater fish-eating birds. 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, although mercury concentrations were not measured in birds collected in 2001. In recent years, investigations have been made into the effects of the newest 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 last collected in 2002.

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 and every three years these annual results are gathered together into a more substantial report in which they are integrated with previous findings. The latest report of this type covers the period up to and including 2000 (Shore *et al.*, 2005a). Results are published periodically in the scientific literature. This current report presents the results of analyses carried out on material collected in 2002 and the findings of novel work areas agreed in the 2002/2003 work programme.

The Wildlife and Pollution contract has been subject to regular scientific assessments within JNCC's rolling programme of peer review. As a result of these, some monitoring activities have been modified, so as to allow the initiation of new studies and reorientation of the PBMS so that remains focused on current chemical risks. Most notably, common kestrels *Falco tinnunculus* are no longer monitored for organochlorines and the intensity of monitoring for organochlorines in sparrowhawks has been reduced. However, kestrels have been monitored for second-generation anticoagulant rodenticides since 2001. This is because an individual study, carried out as part of the PBMS activities, demonstrated that this species may be particularly vulnerable to exposure to these compounds (Shore *et al.*, 2001). Furthermore, new studies have been initiated to investigate potential risks from other chemicals, such as polycyclic aromatic hydrocarbons (see section 8 of this report), and these studies draw on the whole range of material submitted by volunteers to the PBMS.

The core PBMS samples used for chemical monitoring are body tissues from the carcasses of sparrowhawk *Accipiter nisus*, grey heron *Ardea cinerea*, barn owl *Tyto alba*, kestrel *Falco tinnunclulus*, red kite *Milvus milvus*, and the eggs of merlin *Falco columbarius*, golden eagle *Aquila chrysaetos*, sea eagle *Haliaeetus albicilla* and gannet *Morus bassanus*. Carcasses and eggs of other predatory bird species (such as 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*) which do not form the core part of the PBMS but are sent to the Centre for Ecology & Hydrology (CEH) by volunteers, are not analysed chemically. However, post-mortem examinations are carried out the carcasses, relevant information is recorded and the cause of death is determined (and reported back to the volunteer who submitted the carcass). Samples of the egg contents and body organs for these species, and those that do form part of the core monitoring, are all archived at -20°C as part of the unique PBMS tissue and egg sample archive (see section 12 of the current report). This is an invaluable resource and is often used in specific targeted

research studies (for example, see section 11 of the current report).

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. The results from the core monitoring of organochlorine and mercury concentrations in the livers and eggs of various species and of second-generation anticoagulant rodenticides in barn owl and kestrel livers are summarised in sections 1.2-1.6 and in section 1.7, respectively. The aims and results of novel areas of work on polycyclic aromatic hydrocarbon concentrations in eggs, unknown compounds in liver tissues, and spatial variation in liver PCB concentrations in terrestrial species are summarised in sections 1.8, 1.9, 1.10, and 1.11, respectively. Updates on the role of the PBMS in monitoring decabromodiphenylether concentrations in predatory birds and on the cataloguing of the PBMS tissue and egg archive are summarised in sections 1.11 and 1.12.

1.2 Organochlorines and mercury (Hg) 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 (Hg) residues in livers. This enables surveillance of the effects of previous withdrawals of permitted uses of some of these chemicals, and to examine geographical variation in residues. For 2002, the livers from 39 Eurasian sparrowhawks and two grey herons from various localities in Scotland, England and Wales, were analysed for dichlorodiphenyldichloroethylene (DDE), hexachloro-epoxy-octahydro-dimethanonaphthalene (HEOD), PCBs and Hg. None of the sparrowhawks or herons collected during 2002 had liver concentrations of organochlorine insecticides, PCBs or mercury which were indicative of lethal exposure. Average liver concentrations of DDE in sparrowhawks and total PCBs in herons were significantly lower in birds that died in 2001. This may represent a continuation in the decline detected over the long-term course of monitoring but there has been no significant change in liver DDE in sparrowhawks or PCBs in herons over the last five-years up to 2002. Liver Hg concentrations in sparrowhawks rose significantly over the period 1997-2002. The geometric mean liver Hg in sparrowhawks that died in 2002 was similar to that detected in birds in the early 1980s. PCB toxic equivalents (TEQ) concentrations have been reported for the first time in the PBMS. Liver TEQ concentrations varied markedly between individuals. Approximately half of the birds that died in 2002 had non-detected TEQ concentrations The geometric mean concentration for the remainder was below the lowest observed effect concentration (LOEC) reported for liver concentrations in some species, but five sparrowhawks (14% of the birds that died in 2002 and were analysed) had liver TEQ concentrations that exceeded the LOEC.

1.3 Organochlorines in merlin (*Falco columbarius*) eggs

Single eggs collected in 2002 from seven merlin clutches from various parts of Scotland and England were analysed. The results confirm that the eggs of merlins in Britain are still generally contaminated with organochlorine pesticides, PCBs and Hg but concentrations (apart from the total PCB concentration in one egg) were generally low and below concentrations thought to be toxicologically significant. However, the TEQ concentrations in two of the eggs were clearly above the No Observed Effect Concentration (NOEC) range defined for embryotoxicity in birds and most eggs had TEQ concentrations within the LOEC range, although this range overlaps with that for NOECs.

1.4 Organochlorines in golden eagle (*Aquila chrysaetos*) eggs

Single eggs from 13 clutches from Scotland were analysed in 2002; five were from coastal areas (the Western Isles and the Hebrides). DDE, HEOD, total PCB and Hg concentrations in the eggs were generally low and below concentrations thought to impair reproduction, although one egg from North Uist had a total PCB concentration well above the average for golden eagles from coastal areas of western Scotland and had a relatively low shell index. Both the total PCB concentration and the associated TEQ concentration in this egg were within the ranges associated with adverse reproductive effects in birds and PCBs may have contributed to the failure of this egg to hatch. One other egg also had a very high TEQ concentration that might have been associated with embryotoxicity.

1.5 Organochlorines in gannet (*Morus bassanus*) eggs

Twenty gannet eggs were analysed in 2002, comprising 10 eggs from two colonies; Ailsa Craig and the Bass Rock. Concentrations of DDE, HEOD, total PCBs and Hg were generally low and within the range of concentrations detected in eggs previously from these colonies and which are not considered to cause reductions in overall breeding success at the colonies. Median concentrations of DDE and total PCBs, and TEQs in eggs were 1.5-3 fold higher for Bass Rock than Ailsa Craig
In contrast, Hg concentrations were significantly higher (by approximately 13%) in eggs from Ailsa Craig than in eggs from Bass Rock.

1.6 Organochlorines in sea eagle (*Haliaeetus albicilla*) eggs

One failed egg, from Mull, was collected and analysed in 2002. The PCB and TEQ concentrations in this egg were sufficiently high to have potentially caused adverse effects and may have been a contributory cause to the failure of this egg.

1.7 Second generation anticoagulant rodenticides (SGARs) in barn owls (*Tyto alba*), kestrels (*Falco tinnunculus*), and red kites (*Milvus milvus*)

A total of 46 barn owls, and 14 kestrels were received at Monks Wood in 2002 and analysed for four SGARs, difenacoum, bromadiolone, brodifacoum and flocoumafen. The year of death was not known for all birds but of those known to have died in 2002, 19 barn owls (42.2%) and 7 (53.8%) kestrels contained detectable levels of one or more SGAR. Difenacoum and bromadiolone were the compounds that were most frequently detected. The proportion of barn owls that contained residues, and (in comparison) the even higher proportion of kestrels that were contaminated, was consistent with findings in previous years. Three barn owls had liver residues that were in the potentially lethal range of > 0.1-0.2 µg/g wet weight, but none of these birds were diagnosed, on the basis of post-mortem and examination, to have died from rodenticide poisoning. One kestrel had relatively high liver SGAR residues (> 0.541 µg/g wet weight of bromadiolone). Post-mortem examination revealed signs of bruising and hemorrhage in the skull and lungs and it was considered possible that rodenticide may have been a contributory cause of death.

1.8 Polycyclic Aromatic Hydrocarbons (PAHs) in eggs

Polycyclic aromatic hydrocarbons (PAHs) are primarily released to the environment from either point or diffuse natural and anthropogenic sources. They can have toxic (particularly embryotoxic) effects in birds and eggs are also potentially good biomonitors for environmental contamination. Measurement of PAH concentrations in eggs was identified as one potential area of new monitoring for the PBMS (Shore *et al.*, 2005c). A pilot study was carried out in which PAHs were measured in the gannet, golden eagle, merlin and sea eagle eggs received in 2002. In total, analysis for 52 individual PAHs was carried out on each egg.

Detectable concentrations of each PAH that was quantified were found in one or more sample apart from naphthalene, although various methylated naphthalenes were identified and were among the most frequently occurring and highest concentrations of any of the PAHs. The overall pattern of PAH contamination was broadly similar in the gannet, golden eagle and merlin eggs. Exploratory analysis using Principle Components failed to distinguish different bird species from their patterns of PAH accumulation.

Median sum PAH concentrations in the gannet, golden eagle and merlin eggs ranged between 92.6 and 159 ng/g lipid wt. The sum PAH concentration in the single sea eagle egg analysed was 44.3 ng/g lipid. There was no significant difference between sum PAH concentrations in the eggs of different species or in the eggs of gannets from different colonies. Comparison of the summed concentrations for sixteen PAHs reported by Shore *et al*, (1999) with the equivalent concentrations in eggs from the present study suggests that total PAH accumulation may be similar across a range of species in Britain. The broad similarity between species suggests that the egg residues that have been measured may be the result of background levels of exposure to diffuse PAH sources.

As far as can be judged from the limited available toxicity data, the concentrations of individual compounds detected in eggs in this study are unlikely to have embryotoxic effects.

1.9 Analysis of unknown compounds in the livers of predatory birds

Analysis of the livers of birds of prey submitted to the PBMS has revealed the presence of so-called “unknown compounds”. These compounds are detected as unidentified peaks on the analytical chromatographic trace. It is possible that they are other PCB congeners that are not present in analytical suite that is determined but could also be other organic xenobiotics that may or may not pose a toxic hazard to wildlife. The aim of this particular investigation was to carry out a preliminary investigation on a number of bird livers using gas chromatography coupled with mass spectrometry (GC-MS) to estimate: (i) what proportion of the “unknown” peaks that are considered to be PCBs are indeed likely to be PCB congeners; (ii) where possible to identify what non-PCB compounds may be present in the birds.

Typically, most (> 60%) of the unidentified peaks were found to be PCBs not present in the Aroclor 1254 analytical standard. Several other compounds were also detected and were tentatively identified as metabolites of organochlorine insecticides. However, in livers of the small numbers of birds with particularly high “total PCB” concentrations, more than half of the unknown peaks on the analytical trace can be due to the presence of other compounds. Although their identity and potential toxicity is unknown, it is likely that these compounds are likely to be relatively non-polar and insoluble in water.

1.10 Analysis of spatial variation of liver PCBs in terrestrial predatory birds

Liver total PCB concentrations as determined by the PBMS are typically characterised by large scale variation between individuals. The causes of this variation are multiple but a likely key factor is geographical variation in dietary PCB concentrations. Although PCBs are transported globally in the atmosphere, local sources are also likely to be important; such sources include waste disposal and landfill sites and open sources of PCBs such as plastics, paints and adhesives especially in urban areas where the density of such sources may be high. In the current study, the aim was to build on initial analyses conducted previously to determine whether geographical hotspots of PCB contamination (that might be due to local sources) could be identified, and to determine whether liver PCB concentrations varied with larger-scale geographical factors, such as latitude and longitude, rainfall and land cover.

Spatial analysis at the micro-scale indicated the presence of a statistically significant cluster of total liver PCB concentrations in sparrowhawks at a search radius of 15-20 km. This was on Merseyside. No other clusters were detected in sparrowhawks and there were no statistically significant clusters at all for liver total PCB concentrations in kestrels.

The wider scale spatial analysis found a positive association between rainfall and liver PCB concentrations in both sparrowhawks and kestrels (although only statistically significant for sparrowhawks), a significant negative association between latitude and liver PCB concentrations in sparrowhawks, and a significant positive association between degree of urban land cover and liver PCB concentrations in kestrels. In all cases, the strength of these relationships was relatively weak and explained far less of the variation in liver PCB concentrations than intrinsic factors such as the nutritional status of the bird, age and sex.

1.11 Summary report of the role of the PBMS in monitoring decabromodiphenylether (DBDE) concentrations in predatory birds

Decabromodiphenylether (DBDE) is one of the most commercially important (in terms of production and use) of the brominated diphenyl ethers. It has been used as an additive flame retardant in many plastics. The bioaccumulation potential of this compound has been assessed as low but it has been detected in the eggs of peregrine falcons in Sweden (Lindberg *et al.*, 2004). This triggered a subsequent study, initiated the UK Environment Agency and the Bromine Science and Environmental Forum (BSEF) who were representing the producers of DBDE. The aim of the study was to determine the presence and time trends of DBDE residues in predatory birds and so provide environmental information needed for the European environmental risk assessment of DBDE. The project did not formally form part of the activities covered under the annual Wildlife and Pollution project, but many of the samples analysed were from the PBMS frozen tissue and egg archive. The use of the PBMS archive means that the outcome of the work are likely to be of interest to all PBMS stakeholders and the results of the study are briefly summarised in this report.

1.12 Cataloguing of the PBMS archive

As part of the PBMS core activities, liver, kidney, brain, muscle, fat, and gizzard contents have been collected (where available) from the carcasses sent in to CEH. These samples are stored at -20°C to form a tissue archive. We have recently started to archive bone and feather samples as well. Egg contents from the eggs submitted to the PBMS are also stored in the deep freeze archive. Archiving of material from the PBMS has been carried out since the scheme began but the majority of samples date

from the 1970s onwards. The cost of accumulating this archive can be estimated from the current costs of running the PBMS and the duration of the scheme and to date is approximately 4 million pounds.

The primary uses for the archive are to develop new monitoring, for retrospective applied and fundamental studies, and to and to provide material for other studies run by stakeholders.

As the archive has expanded and been used over the course of the PBMS, it has been relocated in different freezers and samples have been used partly or completely for one-off studies. Historically, records have not been systematically kept as to how much of each sample has been used, or remains, nor is the exact location within the freezer known. Thus, management of the archive resource and easy accessing of the samples can be problematic and time consuming.

To overcome these difficulties, an initiative to catalogue the PBMS archive has begun. The aim is to produce a readily interrogated catalogue of what samples are currently held, where they are within the freezer, and an estimate of the amount of sample that is present. The procedures used to catalogue the eggs contents and tissue samples in the archive and the progress made to date is described in this report.

2 Organochlorines in the livers of predatory birds

2.1 Introduction

The main objective of this work is to analyse the livers of predatory birds in order to continue the monitoring of contamination by organochlorines and toxic metals. The livers were from carcasses of birds found dead by members of the public. The chemicals of interest included DDE (from the insecticide dichlorodiphenyltrichloroethane (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. Liver organochlorine concentrations are reported in this section as µg/g wet weight (wet wt), and mercury concentrations are expressed as µg/g dry weight (dry wt). PCB TEQ values were expressed as pg/kg wet 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 number of other species that do not form part of the core monitoring programme were also sent in to CEH during 2002. 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 Monks Wood in 2002 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 this series and by Newton *et al.* (1993).

2.2 Results

A total of 88 sparrowhawks were received at CEH in 2002. A post-mortem examination was conducted on all the birds and selected tissues from each were retained in the PBMS -20ºC tissue archive. The livers of a stratified (by month of death) random sample of about half of the carcasses were analysed chemically, as suggested by (Shore *et al.*, 2005c). In all, livers from 39 sparrowhawks and two herons were analysed. The sample of sparrowhawks included five birds that had died in previous years and one bird that the year in which the carcass was collected was not known. 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 2002 only) are given in Table 2.2.

None of the sparrowhawks collected during 2002 had liver concentrations of organochlorine insecticides which were indicative of lethal exposure. Liver pp'-DDE and HEOD residues were all below 20 µg/g wet wt and 2 µg/g wet wt respectively, concentrations that are typically found in sparrowhawks in Britain currently (Newton *et al.*, 1992, 1993). Gamma-HCH was only detected in the liver of one sparrowhawk and the concentration (0.032 µg/g wet wt) was approximately two orders of magnitude below residues associated with mortality (Wiemeyer, 1996).

Liver total PCB concentrations in sparrowhawks were typically lower than 20 µg/g wet wt. Four individuals had residues of between 20 and 50 µg/g wet wt and the highest concentration was 70.6 µg/g wet wt (bird 13759). None of these five birds had any visible body fat depots. Analysis of the PCB liver concentration data for since monitoring began for all sparrowhawks diagnosed as having died from starvation indicates that the upper quartile (75th percentile), the 95th percentile and the 99th percentile total PCB liver concentrations are approximately 20, 50 and 100 µg/g wet wt respectively. Thus, liver residues of < 100 µg/g wet wt are not exceptional in starved individuals.

The concentrations of organochlorine insecticides and PCBs in the two herons analysed were low and not considered to be toxicologically significant. Gamma-HCH was not detected in either bird.

This is the first year in which TEQ concentrations have been reported. Liver TEQ concentrations varied markedly between individuals. Approximately half of the birds that died in 2002 had non-detected TEQ concentrations associated with their PCB contamination and the geometric mean concentration for the remainder was 80.5 pg/g wet wt. This is within the ranges reported in the livers of various predatory bird species from Europe, the USA and Japan (Coady *et al.*, 2001; Kannan *et al.*, 2003; Senthilkumar *et al.*, 2002). The toxicological significance of liver TEQ concentrations is less well established for livers than for eggs (Hoffman *et al.*, 1996), but 25 ng/g on a lipid weight (lipid wt) basis has been reported as the lowest observed effect concentration (LOEC) for induction of cytochrome P450 enzymes and for a 50% reduction in plasma thyroxine levels in common tern (*Sterna hirundo*) chicks (Bosveld *et al.*, 2000). The geometric mean TEQ concentration for sparrowhawks that died in 2002 and that had detectable TEQ concentrations was 2.73 ng/g, when expressed on a lipid weight basis. This was approximately an order of magnitude lower than the LOEC reported for tern chicks. However, the TEQ liver concentrations in sparrowhawks 13699, 13737, 13839, 13755 and 13759 were 38.1, 57.7, 109, 4920 and 6700 ng/g lipid respectively, above (and for two birds more than 100 fold more than) the LOEC value for tern chicks. These five birds comprised 14% of the sample of sparrowhawks that died in 2002 and it is possible that these individuals may have suffered adverse effects associated with their liver TEQ concentrations.

Mercury concentrations in sparrowhawks and herons were well below the concentration (30 µg/g wet wt, equivalent to approximately 105 µg/g dw) associated with toxic effects in birds of prey (Thompson, 1996).

Statistically significant differences between liver residues in birds that died in 2002 and those that died in the previous year for the compounds analysed were decreases in liver DDE concentrations in sparrowhawks and liver PCB concentrations in herons (Table 2.3). These falls may represent a continuation of the decline in exposure that has occurred over the course of monitoring (Table 2.4). However, these declines have been extremely gradual since 1990 (Shore *et al.*, 2005a) and it is notable that there has been no significant decline over the last five years for either DDE in sparrowhawks or PCBs in herons (Table 2.4). Thus, it is not possible to say whether these differences reflect a real change in exposure between the two years for these compounds.

Analysis of the trend for liver Hg concentrations in sparrowhawks for the last five years indicates that there has been a significant rise in liver concentrations between 1997 and 2002. The geometric mean concentration reported for birds that died in 2002 was similar to that typically detected in birds in the early 1980s. The cause for this increase in contamination is not known.

Table 2.1: **Concentrations of organochlorines insecticides, total PCBs (µg/g wet wt), TEQs (pg/g wet wt) and mercury (µg/g dw) in the livers of juvenile (in first year) and adult (older than first year) sparrowhawks and herons received during 2002.** Lipid wt concentrations for organochlorines and PCBs can be calculated by multiplying the wet wt concentrations by the conversion factor (CF). * indicates missing data that were either not provided by the sender of the carcass or that could not be obtained from the sample received. Congener specific data for PCBs are given in Tables 14.1 and 14.2 in the appendix

Bird No/	Year Found	Vice-County	Age	Sex	CF	pp' DDE	HEOD	Total PCB	PCB (TEQ)	Hg
Eurasian sparrowhawk *Accipiter nisus*										
13668	2002	*	J	M	30.1	0.579	0.030	4.04	ND	3.30
13699	2002	Mid-West Yorkshire	A	F	34.1	4.35	0.310	23.4	1689	2.63
13701	2002	East Inverness-shire	A	F	26.8	5.59	0.419	5.34	169	8.63
13706	2001	Moray	A	M	26.4	0.218	ND	0.701	1.43	9.19
13707	2000	East Ross	J	F	37.9	0.058	0.023	0.511	ND	2.29
13712	2002	*	J	M	31.3	0.385	0.081	5.36	2.30	4.73
13717	2002	East Gloucestershire	A	M	33.1	0.769	0.029	2.47	ND	3.87
13725	2002	South Essex	A	F	32.2	1.129	0.124	8.97	18.4	3.46
13737	2002	NE Yorkshire	J	F	40.1	2.10	0.299	44.3	2707	4.94
13747	2002	NE Yorkshire	A	F	23.5	0.246	0.236	1.11	ND	2.50
13748	2002	NE Yorkshire	A	F	38.6	0.638	0.257	3.68	21.7	4.81
13751	2002	Dorset	A	F	23.5	0.827	0.101	9.68	119	2.62
13755	2002	Oxfordshire	J	F	33.1	4.47	0.351	41.0	148668	10.5
13759	2002	SW Yorkshire	A	F	42.4	1.78	0.729	70.6	158070	5.12
13765	2002	Orkney	J	M	25.9	0.153	0.267	0.148	ND	14.7
13788	2002	South Devon	J	M	28.2	2.06	0.069	6.29	82.7	4.72
13792	2002	West Cornwall	A	M	21.1	0.118	0.116	0.515	ND	3.80
13808	1997	Cambridgeshire	J	M	48.9	4.17	1.52	9.82	29.5	1.95
13812	1999	Cambridgeshire	J	F	34.3	3.33	0.143	1.43	0.300	1.17
13816	1999	West Suffolk	A	M	27.0	0.402	0.056	3.83	56.9	4.68
13822	*	*	J	F	26.8	0.624	0.114	2.60	25.9	1.07
13826	2002	Moray (Elgin)	*	M	27.3	0.533	0.038	0.696	ND	2.65
13839	2002	South Lincolnshire	A	F	42.3	19.2	0.837	33.4	901	7.44
13844	2002	South Somerset	J	M	26.9	ND	0.136	0.149	ND	1.61
13845	2002	Pembrokeshire	J	F	48.4	0.114	ND	0.588	ND	3.39
13849	2002	West Perthshire (with Clackmannan)	J	M	35.1	0.081	0.022	0.104	ND	4.32
13850	2002	South Devon	J	F	42.7	0.552	0.025	1.16	2.01	4.95
13858	2002	East Suffolk	J	F	34.0	0.977	0.111	0.707	2.50	*
13861	2002	Leicestershire (with Rutland)	J	F	33.0	0.190	0.118	0.374	ND	1.57
13867	2002	Argyll Main	J	F	36.7	0.336	0.101	10.2	76.4	8.81
13879	2002	West Sussex	J	F	31.0	0.593	0.074	1.49	16.6	0.69
13894	2002	*	J	F	30.2	0.146	0.027	0.591	0.169	1.77

Table 2.1 cont:

Bird No/	Year Found	Vice-County	Age	Sex	CF	pp' DDE	HEOD	Total PCB	PCB (TEQ)	Hg
Eurasian sparrowhawk *Accipiter nisus*										
13903	2002	Argyll Main	J	M	33.2	0.589	0.043	1.87	ND	11.3
13905	2002	West Sussex	J	F	43.0	0.697	0.090	8.29	2.70	3.27
13919	2002	SW Yorkshire	J	M	30.8	0.089	ND	0.229	ND	0.595
13925	2002	Durham	J	M	38.8	0.466	0.092	7.09	152	3.79
13947	2002	South Essex	J	F	33.9	0.078	0.054	0.920	ND	0.714
13977	2002	Huntingdonshire	A	F	29.8	1.22	0.073	1.10	ND	0.729
13993	2002	North Somerset	A	F	26.2	0.410	0.070	1.18	24.9	1.26
Grey heron *Ardea cinerea*										
13670	2002	Norfolk	A	F	16.6	0.207	0.050	0.257	0.155	2.47
13909	2002	*	J	F	31.4	ND	0.022	0.496	ND	5.83

Table 2.2: Geometric mean concentrations of pollutants in the sparrowhawk and heron in Table 1 (data are only for birds found dead in 2002). GSE=geometric standard error.

	pp'- DDE µg/g wet wt	HEOD µg/g wet wt	Total PCB µg/g wet wt	PCB (TEQ) pg/g wet wt	Hg µg/g dw
Sparrowhawk					
Geometric mean	0.493	0.080	2.38	0.668	3.24
N	33	33	33	33	32
Range of 1 GSE	0.365 - 0.664	0.062 - 0.104	1.75 - 3.23	0.223 – 2.00	2.80 - 3.75
Heron					
Geometric mean	0.014	0.033	0.357	0.01245	3.796
N	2	2	2	2	2
Range of 1 GSE	0.001 - 0.207	0.022 - 0.050	0.257 - 0.496	0.001 - 0.155	2.47 - 5.83

ND values were assigned a value of 0.001 µg/g (organochlorines, PCBs and Hg) and 0.001 pg/g to calculate the geometric mean

Table 2.3: Results from student t-test comparison (log_{10} transformed data) of residue levels from birds collected in 2002 and 2001; values for the two years and the statistical t-values are shown. Minus values indicate a decrease and plus values indicate an increase from 2001. Analysis for Hg and TEQs not shown as concentrations were not measured in 2001

	pp'- DDE	HEOD	Total PCB
Sparrowhawk			
2002	0.493	0.080	2.38
2001	1.66	0.116	2.14
	t_{80} = -3.13**	t_{80} = -1.17	t_{80} = 0.26
Heron			
2002	0.014	0.033	0.357
2001	0.223	0.056	2.44
	t_4 = -1.01	t_4 = -1.22	t_4 = -4.01*

Significance of difference: *P<0.05; **P<0.01; ***P<0.001
Non-detected values taken as 0.001 µg/g

Table 2.4: Trends in pollutant levels in livers of predatory birds during 1963-2002 and 1997-2002. 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). Data for TEQs not shown as data for previous years have not been reported.

		1963-2002		1997-2002	
		N	b	N	b
Sparrowhawk					
	pp'-DDE	2010	-0.031 ***	331	-0.036 ns
	HEOD	2011	-0.031 ***	331	-0.018 ns
	Total PCB	1966	-0.006 *	376	0.013 ns
	Hg	1712	-0.014 ***	281	0.098 ***
Heron					
	pp'-DDE	820	-0.043 ***	26	-0.165 ns
	HEOD	810	-0.048 ***	26	-0.027 ns
	Total PCB	686	-0.022 ***	26	-0.051 ns
	Hg	519	-0.020 ***	22	-0.003 ns

*P=<0.05; **P=<0.01;***P<0.001; ns=not significant
Non-detected values taken as 0.001 µg/g

3 Organochlorines in merlin (*Falco columbarius*) eggs

3.1 Introduction

The eggs of merlins have been monitored since the late 1960s for organochlorine compounds as part of the core PBMS monitoring. The findings from previous analyses of merlin eggs are reported elsewhere (Newton *et al.*, 1982, 1999a; Newton & Haas, 1988) and in previous recent reports in this series (Shore *et al.*, 2005a, b). Those from seven eggs (one per clutch) collected during 2002 are summarised in Table 3.1.

3.2 Results

The analyses of the eggs collected in 2002 confirm that the eggs of merlins in Britain are still generally contaminated with organochlorine pesticides and PCBs. PCB and DDE residues were detected in all seven eggs and HEOD residues in six. The concentrations of all three contaminants were generally low and are below concentrations that are thought to be toxicologically significant (AMAP, 1998; Blus, 1996; Hoffman *et al.*, 1996; Peakall, 1996). The one exception was the PCB concentration of 17.2 µg/g wet wt in egg number E7999. Total PCB concentrations of between 3.5 and 25 µg/g wet wt have been associated with bill deformities and decreased hatching success in a range of avian species including raptors (AMAP, 1998; Hoffman *et al.*, 1996). PCBs may therefore have contributed to the failure of the egg but PCB concentrations that are associated with impaired reproductive success in merlins have not been defined. Furthermore, it is notable that this egg had a non detected TEQ concentration and it may be that the PCB contamination was not toxicologically significant, or at least would not have had a significant toxic effect mediated through the aryl hydrocarbon (Ah) receptor.

No Observable Effect Concentrations (NOECs) for eggs of various experimental and wild bird species range between 1.5 and 200 pg TEQs/ g wet wt; LOECs range between 10 and 2200 pg/g wet wt and the LD_{50} for embryo mortality in white leghorn chickens, one of the more sensitive species, is 115-147 pg/g wet wt (AMAP, 1998). Thus, there is considerable overlap between NOEC, LOEC and LD_{50} TEQ values which in part reflects species variation in sensitivity. The geometric mean and the maximum calculated TEQ concentration associated with PCB contamination in the merlin eggs received in 2002 was 16.4 pg/g wet wt and 473 pg/g wet wt, respectively. Thus, although the total PCB concentrations in most of the merlin eggs received in 2002 were not toxicologically significant, this was not so apparent when toxicity was assessed on a TEQ basis. TEQ concentrations in two of the eggs were clearly above the NOEC range and most eggs had TEQ concentrations within the LOEC range. However, NOEC and LOEC concentrations have not been derived for merlins and it is possible that they are less sensitive to the effects of PCBs than some other species. Accumulation of further TEQ data for this and other species through the PBMS will assist in interpreting the long-term trends and toxicological significance of TEQ concentrations in merlins eggs in Britain.

Mercury was detected in all of the eggs received in 2002. The geometric mean and the maximum concentration was 2.95 µg/g dry wt and 4.64 µg/g dry wt and was typical of concentrations recorded in previous years (Shore *et al.*, 2005a). Total mercury concentrations greater than approximately 2 µg/g wet wt have been associated with impaired hatching in laboratory studies on some species, although the extent to which this effect level can be extrapolated to other species is uncertain as there appears to be considerable variation in sensitivity between species (Thompson, 1996). When expressed on a wet wt

basis, the maximum mercury concentration in the merlin eggs received in 2002 was 0.948 µg/g wet wt, approximately half the concentration associated with adverse effects on reproduction.

Table 3.1: Concentrations of organochlorines insecticides and total PCBs (all in µg/g wet wt), TEQs (pg/g wet wt), mercury (µg/g dw) and the shell indices (SI) for merlin eggs received in 2002. Lipid wt concentrations for organochlorines and PCBs can be calculated by multiplying the wet wt concentrations by the conversion factor (CF). * indicates where shell indices could not be measured because of the poor condition of the eggshell. Congener specific data for PCBs and TEQs are given in Tables 14.3 and 14.4, respectively of the Appendix.

Egg No/	Year	Vice-County	SI	CF	pp'-DDE	HEOD	Total PCB	PCB TEQ	Hg
Southern Scotland									
E8007	2002	Lothian	*	18.35	2.88	0.451	1.83	35.8	4.23
E8008	2002	Borders	1.33	18.10	1.35	0.146	2.26	26.2	2.46
North East England									
E7999	2002	North Yorkshire	1.07	16.39	2.16	0.458	17.2	ND	2.22
E8106	2002	Northumberland	1.54	15.85	0.975	0.139	2.40	50.0	3.56
E8110	2002	Northumberland	1.20	22.78	2.97	0.665	2.86	35.1	2.37
E8115	2002	Northumberland	1.33	20.99	0.800	0.166	3.09	413	2.12
E8119	2002	Northumberland	1.23	19.10	0.300	ND	3.03	472	4.64

ND is not detected

4 Organochlorines and mercury in golden eagle (*Aquila chrysaetos*) eggs

4.1 Introduction

The findings from the long-term monitoring of contaminants in golden eagle eggs carried out as part of the PBMS have been reported in previous reports by Newton & Galbraith (1991) and were recently summarised as part of the series of reports for the PBMS (Shore *et al.*, 2005a). Eggs from 13 clutches were received in 2002, and five were from coastal areas, (Western Isles and Hebrides). The results of the chemical analyses are given in Table 4.1.

4.2 Results

The DDE, HEOD, total PCB and Hg concentrations in the eggs received in 2002 were generally low and below concentrations thought to impair reproduction (AMAP, 1998; Blus, 1996; Hoffman *et al.*, 1996; Peakall, 1996; Thompson, 1996). However one egg (E8055) from North Uist had a total PCB concentration (14.1 ug/g wet wt) that was well above the average for golden eagles from coastal areas of western Scotland (Shore *et al.*, 2005a), and had a relatively low shell index. Both the total PCB concentration and the associated TEQ concentration (1532 pg/g wet wt) were within the ranges associated with adverse reproductive effects in birds (AMAP, 1998; Hoffman *et al.*, 1996) and PCBs may have contributed to the failure of this egg to hatch.

This is the first year in which TEQ concentrations have been reported for golden eagle eggs. Eight of the 13 eggs had non-detected TEQ concentrations. The only egg other than E8055 (see above) that had a high TEQ value was egg E8010 (Table 4.1) from a nest in Mull. The total PCB concentration for this egg was not high but congener 81, which has a high TEF value, was detected and accounts for the high calculated TEQ concentration (Tables 14.5 and 14.6). It is possible that contamination with congener 81 was a contributory factor in the failure of this egg.

Table 4.1: Concentrations of organochlorines insecticides and total PCBs (all in µg/g wet wt), TEQs (pg/g wet wt), mercury (µg/g dw) and the shell indices (SI) for golden eagle eggs received in 2002. Lipid wt concentrations for organochlorines and PCBs can be calculated by multiplying the wet wt concentrations by the conversion factor (CF). * indicates where shell indices could not be measured because of the poor condition of the eggshell. Congener specific data for PCBs and TEQs are given in Tables 14.5 and 14.6 respectively in the Appendix.

Egg Number	Year	Vice-County	SI	CF	pp'-DDE	HEOD	Total PCB	PCB TEQ	Hg
Eastern Highlands									
E8005	2002	Highland	3.54	25.52	0.076	ND	0.650	ND	ND
Western Highlands									
E8010	2002	Mull	3.11	16.54	0.057	0.018	0.230	39652	0.083
E8011	2002	Mull	*	4.22	0.230	0.098	0.601	ND	0.054
Hebrides									
E8055	2002	North Uist	2.39	31.16	0.803	0.044	14.1	1532	1.26
E8056	2002	North Uist	4.19	20.93	0.093	0.048	0.819	1.68	0.312
E8057	2002	Skye	5.80	21.36	0.058	0.020	0.217	ND	0.327
South West Scotland									
E7990	2002	Argyll	2.84	23.48	0.063	0.094	0.420	ND	0.154
E7991	2002	Argyll	2.55	23.43	0.045	0.020	0.248	1.70	0.121
E7993	2002	Argyll	3.56	17.81	0.082	0.094	0.143	ND	0.237
E7995	2002	Argyll	3.03	15.89	0.054	0.045	0.458	5.16	0.091
E7996	2002	Argyll	2.87	27.11	0.075	0.070	0.052	ND	0.323
E7998	2002	Argyll	2.42	21.31	0.036	0.035	0.133	ND	0.127
E8000	2002	Argyll	2.79	19.97	0.007	0.055	0.211	ND	0.423

ND is not detected

5 Organochlorines and mercury in gannet (*Morus bassanus*) eggs

5.1 Introduction

The findings from all gannet eggs examined up to 1988 were published by Newton *et al.* (1990a) and long-term trends in contaminant levels were summarised as part of the series of reports for the PBMS by Shore *et al.* (2005a). Gannet eggs are typically collected during visits to colonies made during laying or the early incubation period, and about ten eggs are taken from each colony. Collections are made approximately every two years. Twenty gannet eggs were received for analysis in 2002.

5.2 Results

Ten gannet eggs from Ailsa Craig and ten from Bass Rock were analysed. The contaminant concentrations and eggshell indices are given in Table 5.1.

Concentrations of DDE, HEOD, total PCBs and Hg were generally low and within the range of concentrations detected in eggs previously from these colonies (Shore *et al.*, 2005a). These concentrations are too low to cause reductions in overall breeding success at Ailsa Craig and Bass Rock (Newton *et al.*, 1990a) but the long-term analysis of the gannet eggs gives an indication of 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.

Differences in contaminant concentration between colonies were compared by non-parametric statistics because the data were not distributed normally. There was no significant difference between the median HEOD concentrations in eggs from Bass Rock and Ailsa Craig (0.048 µg/g wet wt vs 0.035 µg/g wet wt, respectively; Mann-Whitney U test, $U = 29$, $P > 0.05$). However, the median concentrations of DDE and total PCBs were 1.5-3 fold higher in the eggs from Bass Rock compared with those from Ailsa Craig (Mann-Whitney U test, $U \leq 10$, $P < 0.005$ in both comparisons). This was not attributable to any possible differences in moisture content in the eggs—the eggs Ailsa Craig were cracked when they were received by CEH and may have desiccated to some extent— as the same (in terms of magnitude and statistical significance) inter-colony differences were evident when concentrations were expressed on a lipid wt basis.

The estimated TEQ concentrations were also significantly higher in eggs from Bass Rock (Mann-Whitney U test, $U = 19$, $P < 0.05$) than in those from Ailsa Craig. The difference between the two colonies was four fold, somewhat higher than the inter-colony difference detected for total PCBs. The TEQ concentrations in the majority of the eggs that were analysed from each colony were below the LOEC concentration reported for a range of species (AMAP, 1998).

In contrast to the organic contaminants, mercury concentrations were significantly higher (by approximately 13%) in eggs from Ailsa Craig than in eggs from Bass Rock. This difference, although slight, was statistically significant (Mann-Whitney U test, $U = 23$, $P < 0.05$).

Table 5.1: Concentrations of organochlorines insecticides and total PCBs (all in µg/g wet wt), TEQs (pg/g wet wt), mercury (µg/g dw) and the shell indices (SI) for gannet eggs received in 2002 from Ailsa Craig and Bass Rock. Lipid wt concentrations for organochlorines and PCBs can be calculated by multiplying the wet wt concentrations by the conversion factor (CF). * indicates where shell indices could not be measured because of the poor condition of the eggshell. Congener specific data for PCBs and TEQs are given in Tables 14.7-14.10. of the Appendix.

Egg Number	Year	SI[1]	CF	pp'-DDE	HEOD	Total PCB	PCB TEQ	Hg
Ailsa Craig								
G1073	2002	*	28.51	ND	ND	0.414	3.27	3.20
G1074	2002	*	23.00	0.048	0.030	0.437	0.230	1.90
G1075	2002	*	25.70	0.052	0.025	0.691	0.250	2.03
G1076	2002	*	20.87	0.060	0.078	1.23	4.40	2.63
G1077	2002	*	26.56	0.062	0.046	1.43	4.42	1.80
G1078	2002	*	27.11	0.038	0.046	0.655	16.9	1.78
G1079	2002	*	29.58	0.046	0.055	1.44	21.4	3.15
G1080	2002	*	20.81	ND	ND	1.35	4.46	1.94
G1081	2002	*	30.30	0.049	0.036	0.278	0.300	2.19
G1082	2002	*	29.74	0.052	0.033	1.13	4.38	2.27
Bass Rock								
G1083	2002	2.93	22.89	0.078	0.051	2.63	9.28	2.21
G1084	2002	3.26	38.09	0.054	0.034	0.909	1.95	1.85
G1085	2002	2.69	25.67	0.117	ND	3.80	9.69	1.46
G1086	2002	3.05	19.64	0.101	0.050	2.35	6.55	1.88
G1087	2002	2.90	21.77	0.084	0.047	4.64	37.4	2.16
G1088	2002	2.66	24.05	0.082	0.045	3.02	29.7	1.72
G1089	2002	3.00	19.87	0.073	0.042	1.92	26.0	1.91
G1090	2002	3.05	25.39	0.058	0.075	0.900	3.83	1.44
G1091	2002	2.87	26.06	0.103	0.092	4.08	46.1	1.91
G1092	2002	2.62	23.53	0.097	0.086	2.64	31.6	1.85

[1]eggs from Ailsa Craig were broken on arrival and calculation of shell index was not possible. ND is not detected

6 Organochlorines and mercury in sea eagle (*Haliaeetus albicilla*) eggs

6.1 Introduction

Sea eagles were reintroduced to western Scotland between 1976 and 1985. They 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 may be exposure to contaminants which the birds could acquire particularly from the marine component (various fish and seabirds) of their diet.

Some of the Scottish white-tailed eagles nest on inaccessible sea cliffs. This makes collection of samples difficult. One failed egg was collected in 2002 and a total of ten eggs has been obtained and analysed during the course of this monitoring scheme.

The lipid wt equivalent concentrations of the DDE and total PCB wet wt concentrations in the egg were 33.9 µg/g and 440 µg/g, respectively. Lipid DDE concentrations in sea eagle eggs of 30-50 µg/g and 100-120 µg/g have been suggested as the LOECs for eggshell thickness and productivity and complete reproductive failure has been associated with a DDE concentration of 900 µg/g lipid wt (Helander *et al.*, 2002). Adverse effects on productivity due to PCBs appear to occur at lipid concentrations of about 300 µg/g in sea eagle eggs, although, because there is a strong association between PCB and DDE residues, there is some uncertainty about whether such effects are actually due to PCBs or DDE (Helander *et al.*, 2002). The total PCB concentration in the sea eagle egg collected from Mull in 2002 was above the LOEC associated with adverse effects in sea eagles and within the range of 3.25 - 25 µg/g wet weight associated with decreased hatching success in various avian species (AMAP, 1998; Hoffman *et al.*, 1996). The PCB TEQ concentration was also above the range of NOECs and within the range of LOECs for reproduction that have been reported for various avian species (see section 3.2 and AMAP (1998)). Thus, although it is not certain that PCBs were a contributory cause of reproductive failure of this egg, the residues were of a magnitude that might have been expected to have adverse effects.

Table 6.1: Concentrations of organochlorines insecticides and total PCBs (all in µg/g wet wt), TEQs (pg/g wet wt), mercury (µg/g dw) and the shell indices (SI) for the white-tailed sea eagle egg received in 2002. Lipid wt concentrations for organochlorines and PCBs can be calculated by multiplying the wet wt concentrations by the conversion factor (CF). Congener specific data for PCBs and TEQs are given in Tables 14.11 in the Appendix

Egg Number	Year	County	SI	CF	pp'-DDE	HEOD	Total PCB	PCB TEQ	Hg
Western Highlands									
E7988	2002	Mull	2.83	22.03	1.537	0.116	20.0	260	1.64

7 Second generation anticoagulant rodenticides (SGARs) in barn owls (*Tyto alba*), kestrels (*Falco tinnunculus*)

7.1 Introduction

The aim of this work is to monitor the exposure of certain predatory bird species to second-generation anticoagulant rodenticides (SGARs). The compounds of interest are difenacoum, bromadiolone, brodifacoum and flocoumafen and the species monitored are the barn owl and kestrel. The carcasses were supplied by members of the public and included birds that had died from various causes, mainly accidents. The PBMS has monitored SGAR residues in barn owls since 1983 and the findings from barn owls analysed in previous years have been reported by Newton *et al.*, 1990b; 1999b) and long-term trends were last reviewed in this report series by (Shore *et al.*, 2005b). This is the second year in which the PBMS has routinely monitored kestrels for SGARs. Kestrels have been incorporated into the scheme because a study of birds that died between 1997 and 2000 indicated that a very high proportion (24/36 individuals) of the sample) had detectable concentrations of one or more SGAR in the liver (Shore *et al.*, 2001).

The results of the analysis of the livers of 46 barn owls and 14 kestrels that were sent in to CEH in 2002 are reported in Table 9 and 10, respectively.

7.2 Methods

Analysis of rodenticides in liver tissue was carried out using the general technique outlined by Hunter, (1985) and described in previous reports and 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. A detailed description of the methods is given in Shore *et al.* (2003). Livers from barn owls, kestrels and kites were analysed together and in random order.

7.3 Results of analyses of birds received in 2002

Of the 46 barn owls received in 2002, one died in 2000 and there was no information on year of death provided with five others but these were presumed to have been found in 2002. Of the birds known or thought to have died in 2002, 19 (42.2%) contained detectable levels of one or more SGAR. This proportion was very similar to that that (22/53 = 41.5%) for owls that were received and at Monks wood in 2001. Overall, the data for the 2002 birds was consistent with the trend reported for earlier years that suggested the increase since 1983 (when monitoring began) in the proportion of birds exposed was leveling off at about 40% (Newton *et al.*, 1999b).

Difenacoum, bromadiolone, brodifacoum and flocoumafen occurred in 13 (28.3% of the sample), 14 (30.4%), 3 (6.5 %) and 1 (2.2%) barn owls, respectively. The predominance of difenacoum and bromadiolone and low levels of brodifacoum and flocoumafen (both indoor use only) is consistent with findings in barn owls in previous years.

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 wet wt (Newton *et al.*, 1998) and > 0.2 µg/g wet wt (Newton *et al.*, 1999b) and is so classed on the basis of two sets of observations. These are that owls diagnosed at post-mortem of having died from rodenticide poisoning (because they had characteristic signs of haemorrhaging from such organs as the heart, lungs, liver, brain and/or subcutaneous areas) almost all had liver residues >0.1 µg/g wet wt, and, secondly, that owls that had been experimentally poisoned had residues of the range 0.2-1.72 µg/g wet wt (Newton *et al.*, 1999b). Of the barn owls that died or were assumed to have died in 2002, 3 (6.7% of the sample) had liver residues (summed values for all four SGARS that were monitored) greater than 0.1 µg/g wet wt but none had a liver residue >0.2 µg/g wet wt. The proportion of owls that died in 2002 that had "high" residues was somewhat lower than in recent previous years but it is possible that this may simply due to random inter-year variation. Post-mortem examination did not reveal signs of haemorrhaging consistent with rodenticide poisoning in any of the three owls birds with SGAR residues of > 0.1 µg/g wet wt and the attributed causes of death were road traffic accidents (two birds) and collision.

Of the 14 kestrels received in 2002, it was known that 13 of them had been found that year. There was no information on the year of death of the other bird. In total, eight of the 14 kestrels (57.1 % of the sample) contained detectable concentrations of one or more SGAR; this was similar to the proportion (60.9%) in the sample of 23 kestrels that were received in 2001 (Shore *et al.*, 2005b). The difference between the proportions of barn owls and kestrels received in 2002 that contained detectable liver residues of one or more SGAR (41.5% vs 57.1 %) was not statistically significant (Fisher's Exact test, $P > 0.05$) but the trend of higher frequency of detection in kestrels than barn owls was also found in birds that died in 2001 and in birds that died in earlier years (Shore *et al.*, 2001). Difenacoum and bromadiolone were detected most frequently in kestrels and occurred in 3 (21.4% of the sample) and 6 (42.9%) birds, respectively; brodifacoum and flocoumafen were detected in 1 (7.1 %) and no birds. Two kestrels contained detectable concentrations of difenacoum and bromadiolone and one bird contained residues of bromadiolone and brodifacoum.

Only one kestrel had a liver SGAR residue >0.2 µg/g wet wt; this bird (13771) was found on farmland and had a liver bromadiolone residue of 0.541 µg/g. Post-mortem examination did not reveal evidence of hemorrhaging that was clearly characteristic of rodenticide poisoning, but there were signs of bruising and hemorrhage in the skull and lungs. These may have been the result of an earlier collision that eventually proved fatal, but there was no evidence of bone fractures and it is possible that the bruising and hemorrhaging was associated with the bird's exposure to bromadiolone. Rodenticide may have been a contributory cause of death.

Table 7.1: Difenacoum (difen), bromadiolone (brom), flocoumafen (floc) and brodifacoum (brodif) concentrations (µg/g wet wt) in the livers of 46 male (M) and female (F) barn owls received in 2002. Juveniles are birds in first year, adults are birds older than first year.

bird no/	date		Vice-county	age	sex	difen	brom	floc	brodif
13782	Jan	2002	*	*	*	ND	ND	ND	ND
13785	Feb	2002	*	*	*	0.038	0.055	ND	ND
13789	Mar	2002	*	*	*	ND	ND	ND	ND
13692	Jan	2002	North Wiltshire	A	F	0.012	0.064	ND	0.018
13710	Jan	2002	East Ross	J	M	ND	ND	ND	ND
13714	Feb	2002	South Devon	J	F	ND	ND	ND	ND
13715	Feb	2002	Cambridgeshire	A	F	ND	ND	ND	ND
13720	*	*	Dorset	A	F	0.076	0.060	ND	ND
13730	Mar	2002	Cardiganshire	J		ND	ND	ND	ND
13739	Mar	2002	North Wiltshire	A	M	0.008	0.028	ND	ND
13742	Mar	2002	South Lancashire	J	M	ND	0.086	ND	ND
13749	Apr	2002	West Norfolk	J	M	0.141	0.041	ND	0.011
13752	Apr	2002	Oxfordshire	A	F	ND	0.175	ND	ND
13775	Apr	2002	West Perthshire (with Clackmannan)	J	M	ND	0.033	ND	ND
13793	Jan	2002	*	J	F	ND	ND	ND	ND
13820	*	*	West Suffolk	A	M	ND	ND	ND	ND
13824	Jun	2002	South Lincolnshire	J	M	0.014	0.057	ND	ND
13836	Jul	2002	South-East Yorkshire	J	M	ND	ND	ND	ND
13846	Aug	2002	Northamptonshire	J	F	ND	ND	ND	ND
13847	Jul	2002	Cambridgeshire	J	M	ND	ND	ND	ND
13851	Aug	2002	South Hampshire	J	M	ND	0.050	ND	ND
13866	Aug	2002	West Cornwall (with Scilly)	J	M	0.052	ND	ND	ND
13868	*	*	East Norfolk	J	M	ND	ND	ND	ND
13872	Sept	2002	North Lincolnshire	J	F	ND	ND	ND	ND
13874	*	*	Pembrokeshire	J	M	ND	ND	ND	ND
13878	Sept	2002	West Perthshire		M	ND	ND	ND	ND
13883	Jun	2002	West Cornwall	A	M	ND	ND	ND	ND
13886	Oct	2002	East Cornwall	J	M	ND	ND	ND	ND
13887	Jul	2002	East Cornwall	J	F	ND	ND	ND	ND
13888	Jul	2002	East Cornwall	J	M	0.010	0.089	ND	ND
13907	Sept	2002	South-East Yorkshire	J	M	ND	0.088	ND	ND
13910	*	*	Berkshire	J	M	ND	0.037	ND	0.007
13921	Oct	2002	*	A	F	0.145	ND	ND	ND
13927	Oct	2002	Lincolnshire	J	F	ND	ND	ND	ND
13957	Dec	2000	East Cornwall	A	F	ND	ND	ND	ND
13961	Mar	2002	East Cornwall	A	M	ND	ND	ND	ND
13966	Jul	2002	West Cornwall	J		0.032	ND	ND	ND
13970	Aug	2002	West Cornwall	A	M	ND	ND	ND	ND
13974	Oct	2002	East Cornwall	A	F	0.014	ND	0.010	ND
13978	Nov	2002	West Inverness-shire	J	M	ND	ND	ND	ND

Table 7.1 continued

bird no/	date	Vice-county	age	sex	difen	brom	floc	brodif
13981	Mar 2002	North Wiltshire	A	F	0.008	ND	ND	ND
13982	Nov 2002	Kirkcudbrightshire	A	F	ND	ND	ND	ND
13983	Nov 2002	East Norfolk	J	F	0.020	0.044	ND	ND
13984	Dec 2002	North Essex	J	M	ND	ND	ND	ND
13986	Oct 2002	East Ross	J	F	ND	ND	ND	ND
13992	Dec 2002	Glamorgan	A	F	ND	ND	ND	ND

ND is not detected

Table 7.2: Difenacoum (difen), bromadiolone (brom), flocoumafen (floc) and brodifacoum (brodif) concentrations (µg/g wet wt) in the livers of 14 male (M) and female (F) kestrels received in 2002. Juveniles are bird in first year, adults are birds older than first year.

bird no/	date		location	age	sex	difen	brom	floc	brodif
13724	Feb	2002	East Sussex	J	F	ND	ND	ND	ND
13727	Feb	2002	Westmorland with North Lancashire	J	F	ND	ND	ND	ND
13728	Feb	2002	Westmorland with North Lancashire	J	M	ND	0.120	ND	ND
13766	Dec	1992	Orkney	J	F	0.026	0.037	ND	ND
13770	Apr	2002	South Hampshire	A	F	ND	ND	ND	ND
13771	Jan	2002	South-east Yorkshire	A	F	0.02	0.541	ND	ND
13825	Jan	2002	South Somerset	J	F	ND	0.069	ND	ND
13827	Jun	2002	West Sussex	A	M	ND	0.053	ND	0.011
13829	Jul	2002	Cambridgeshire	J		0.078	ND	ND	ND
13835	Jul	2002	South Somerset	J	M	ND	ND	ND	ND
13857	*	*	East Kent	J	F	0.065	ND	ND	ND
13895	Aug	2002	West Cornwall	J	M	ND	ND	ND	ND
13980	Nov	2002	Northamptonshire	J	F	ND	0.083	ND	ND
13987	Dec	2002	North Somerset	J	M	ND	ND	ND	ND

ND is not detected

8 Polycyclic aromatic hydrocarbons (PAHs) in eggs

8.1 Introduction

Polycyclic aromatic hydrocarbons (PAHs) are primarily released to the environment as a result of incomplete combustion of organic materials. Releases are predominantly to the atmosphere and may be from either point or diffuse sources. These sources can be natural or anthropogenic and include forest fires, volcanoes, coal coking, aluminium production, power plants, incinerators and traffic (Environment Canada, 1994; WHO, 1998). Another source of PAHs, particularly for marine birds, is through contact with crude oil that has been discharged deliberately or accidentally into the environment (Shore *et al.*, 1999).

There appear to be no little or data on uptake rates for PAHs by free-living terrestrial vertebrates. The presence of detectable residues and metabolites in the body tissues and eggs of some biota (Albers & Loughlin, 2003) indicates that exposure does occur. This may be inhalation, topical contact (for example, oiled incubating birds can transfer oil from the feathers to their eggs, causing *in ovo* exposure) or oral routes (contaminated diet; preening/grooming of oiled feathers and fur). Exposure may lead to subsequent toxic effects. Some PAHs are carcinogens and they can also cause lesions in fish (Hallett & Brecher, 1984). Of particular interest with respect to the PBMS is that various individual PAHs have been shown to have toxic effects on bird eggs (Brunstrom *et al.*, 1991; Hoffman, 1979). Furthermore, because PAHs are lipophilic, eggs are potentially good biomonitors of PAH contamination in the environment. PAH concentrations in eggs can be some two or three orders of magnitude higher than in livers (Hallett & Brecher, 1984).

Despite the high potential for PAHs to cause biological impacts, particularly in birds, there is generally little information on present concentrations in the eggs of birds in Britain. Most of the data that are available on PAH concentrations in eggs relate to coastal nesting species and in seabirds (Albers & Loughlin, 2003; Shore *et al.*, 1999), reflecting the concerns arising from oiling of birds from ship wrecks and illegal discharges of oil residues to the marine environment. In the recent review of potential modifications to the PBMS, measurement of PAH concentrations in eggs was identified as one potential area of new monitoring. The value of this in policy terms would be that it would establish whether there was evidence of significant accumulation of embryotoxic compounds in bird eggs in terrestrial and marine environments and whether the extent of exposure is changing over time (Shore *et al.*, 2005c). It was concluded that a sensible first step would be a pilot study in which the eggs of merlins, gannets, and possibly other species, would be analysed for their PAH concentrations. The aim was to determine whether detectable concentrations of PAHs are found in the eggs of these species and to provide a preliminary assessment as to whether they could be useful for measuring temporal trends in exposure. The results of this pilot study are reported here.

8.2 Methods

PAHs were measured in the gannet, golden eagle, merlin and sea eagle eggs received in 2002 and that were analysed for organochlorine insecticides and PCBs (sections 2-6). A total of 52 individual PAHs (Table 8.1) were determined and the analytical suite used was in part determined on the basis of PAHs known to be associated with oil pollution (Aronson *et al.*, 2001; Goldberg *et al.*, 1978). This suite included the sixteen PAHs that were analysed in herring gull *Larus argentatus*, cormorant *Phalacrocorax carbo*, shag *Phalacrocorax aristotelis* and chough *Pyrrhocorax pyrrhocorax* eggs in the mid 1990s (Shore *et al.*, 1999), the only other published data on PAH concentrations in bird in Britain.

Solvent extraction of each egg involved first homogenising the contents and drying an accurately weighed sub-sample (about 4 g) with anhydrous sodium sulphate. The mixture was left to stand, with occasional stirring, in 20 ml glass-distilled hexane for an hour. Using sequential 10 ml hexane washings, the extract was then transferred via a drying funnel containing 5 g anhydrous sodium sulphate to a pre-weighed measuring cylinder. The extract was made up to 50 ml volume, mixed by shaking and inverting, and a 40 ml aliquot was reduced to 2 ml in a Kurdena-Danish flask in an 80°C water bath. Fine adjustment of volume was achieved either by addition of hexane or through evaporation under an ECD grade nitrogen stream.

A 200 µl aliquot of each concentrated extract was cleaned-up using an alumina column consisting of 1.0 g deactivated alumina in a plugged Pasteur pipette. The sample and subsequent 1 ml hexane washings were passed through the column until 5 ml of eluant were collected. This was reduced under a nitrogen stream to 0.5 ml and a 5 µl aliquot was analysed by Gas Chromatography - Mass Spectrometry (GC-MS: Agilent 6890 gas chromatograph, 5973 Mass Selective Detector with electron impact ionisation). PAHs were separated on a 30m x 0.25mm id x 0.25 micron HP5-MS column (Agilent) fitted with a 5m x 0.25mm id methyl deactivated retention gap (SGE UK Ltd). The helium carrier gas was maintained at a constant flow of 2 ml min^{-1} by electronic pressure control. Samples (60µl) were injected onto the column via a Programmable Temperature Vaporising inlet in solvent vent mode. The column oven temperature was held at 50°C for 5 minutes, then ramped at 15°C min^{-1} to 200°C, held for 5 minutes, then ramped at 5°C min^{-1} to 250°C, then ramped at 10°C min^{-1} to 300°C and held for 10 minutes. The MSD was operated in select ion mode at the highest resolution available; where possible three ions were monitored for each compound.

Peaks were identified by their relative retention time to the closest eluting deuterated PAH and by their qualifier ion ratios (where more than one ion was collected). The upper and lower limits for the qualifier ion ratios and relative retention time windows were 20% and 0.2% respectively. The concentration of each compound was calculated using the isotope dilution method. The peak area of the target ion was used for calculations. The Limits of Detection (LoDs) were calculated from all the analytical blanks run with each batch of samples. The LoD for each compound was defined as three times the standard deviation of the total amount of that compound in the sample blanks. The LoDs was applied after subtraction of the average of the total amounts found in the blanks. Lod values are given in Table 8.1

Samples were analysed in batches of 14 samples. In addition to the samples, each batch included two analytical blanks and two control (chicken egg) samples. Every sample, blank and control was spiked with known amounts of deuterated PAHs prior to extraction. No suitable certified reference material was obtainable. Recovery data are also given in Table 8.1.

PAH concentrations are reported on a lipid wt basis for ease of comparison with previously published data on PAHs in eggs in Britain (Shore *et al.*, 1999). Residue data had skewed distributions and the assumptions of parametric statistical tests were not often met. Therefore, medians were used to indicate average concentrations, and comparisons of PAH residues in eggs between different bird species were made using the non-parametric Kruskal-Wallis test.

Table 8.1: Limits of Detection (LoD) expressed as ng/g lipid in the total sample, and mean % recovery (based on recoveries from deuterated PAHs) for the 52 polycyclic aromatic hydrocarbons (PAHs) determined in this study

PAH	LoD (ng)	mean recovery (%)	PAH	LoD (ng)	mean recovery (%)
Naphthalene	10.9	38.6	1-Methylfluoranthene	0.100	66.3
2-Methylnaphthalene	5.18	38.6	Benzo[a]fluorene	0.145	66.3
1-Methylnaphthalene	3.13	38.6	Benzo[b]fluorene	0.099	66.3
2-Ethylnaphthalene	0.491	38.6	1-Methylpyrene	0.042	66.3
1-Ethylnaphthalene	0.503	38.6	Benzo[ghi]fluoranthene	0.022	66.3
2,6 & 2,7-Dimethylnaphthalene	0.354	38.6	Benzo[c]phenanthrene	0.012	66.3
1,3 & 1,7-Dimethylnaphthalene	0.433	38.6	Cyclopenta[cd]pyrene	0.013	66.3
1,6-Dimethylnaphthalene	0.196	38.6	Benz[a]anthracene	0.056	66.3
2,3 & 1,4-Dimethylnaphthalene	0.275	38.6	Triphenylene & Chrysene	0.079	66.3
1,5-Dimethylnaphthalene	0.435	38.6	3-Methylchrysene	0.066	89.7
Acenaphthylene	0.824	38.6	2-Methylchrysene	0.021	89.7
1,2-Dimethylnaphthalene	0.162	38.6	5-Methylchrysene	0.001	89.7
1,8-Dimethylnaphthalene	0.033	38.6	4 & 6-Methylchrysene	0.004	89.7
Acenaphthene	0.757	38.6	1-Methylchrysene	0.014	89.7
2,3,5-Trimethylnaphthalene	0.658	38.6	Benzo[b & j & k]fluoranthene	0.102	89.7
Fluorene	1.11	38.6	Benzo[e]pyrene	0.058	89.7
Dibenzothiophene	3.09	38.6	Benzo[a]pyrene	0.080	66.8
Phenanthrene	2.51	67.2	Perylene	0.040	89.7
Anthracene	0.140	67.2	Ideno[1,2,3-cd]pyrene	0.226	89.7
2-Methylphenanthrene	0.571	67.2	Dibenz[ah]anthracene	0.015	89.7
1-Methylphenanthrene	0.396	67.2	Benzo[ghi]perylene	0.053	89.7
3,6-Dimethylphenanthrene	0.100	67.2	Anthanthrene	0.018	89.7
Fluoranthene	0.468	62.9	Dibenzo[a,i]pyrene	0.018	89.7
9,10-Dimethylphenanthrene	0.046	62.9	Coronene	0.026	89.7
Pyrene	0.606	66.3	Dibenzo[a,e]pyrene	0.010	89.7
2-Methylfluoranthene	0.029	66.3	Dibenzo[a,h]pyrene	0.010	89.7

Mean ± SE lipid wt in samples analysed was 0.198 ± 0.02 g (n = 41).

8.3 Results

Detectable concentrations of each PAH that was quantified were found in one or more sample apart from naphthalene which was not detected at all. This apparently contrasts with reported concentrations of naphthalene in coastal nesting birds (Shore *et al.*, 1999), but the difference between the studies may have been due to methodological differences in the chemical analysis. The recovery for naphthalene and the associated detection limit were poorer in the present study than in the study by Shore *et al.* (1999). This is attributable to differences in extraction and clean-up methods as the analysis used in the present study was optimized for the detection of a wider range of PAHs. In the present study, although naphthalene itself was not detected, various methylated naphthalenes were identified and were among the most frequently occurring and highest concentrations of any of the PAHs (Table 8.2). Methylated naphthalenes were not quantified in the study by Shore *et al.* (1999).

Overall, the pattern of PAH contamination was broadly similar in the gannet, golden eagle and merlin eggs. A total of 23 individual PAHs had median concentrations above the detection limit in the eggs of one or more species (Table 8.2). Fifteen of these compounds occurred in more than 50% of all the samples analysed and eleven had median concentrations above the detection limit in all three species (Table 8.2). Exploratory analysis using Principle Components failed to distinguish different bird species from their patterns of PAH accumulation.

The sum PAH concentrations for the different birds are given in Table 8.3. Median sum PAH concentrations in the different species ranged between 92.6 and 159 ng/g lipid wt (Figure 8.1). The sum PAH concentration in the single sea eagle egg that was analysed was 44.3 ng/g lipid. There was no significant difference between sum PAH concentrations in the eggs of different species or in the eggs of gannets from different colonies (Kruskall-Wallis statistic = 1.892, $P>0.05$).

There are limited data on the embryotoxicity of individual PAHs. Brunstrom (1991) assessed the embryotoxicity of 24 PAHs that were injected into chicken eggs and found that the most toxic compounds were benzo[k]fluoranthene, dibenz[a,h]anthracene and benzo[a]anthracene; LD_{50} values were 14, 39 and 79 ng/g wet weight respectively. Median lipid weight concentrations of these compounds were greatest in golden eagle eggs (Table 8.2) and the equivalent median wet weight concentrations were 0.03 ng/g at most, some two-three orders of magnitude below the reported LD_{50} values. Maximum wet weight concentrations in any sample were at least five fold lower than the LD_{50} values. Doses of approximately 36 ng/g benzo[*a*]pyrene and 270 ng/g chrysene both caused significant reduction in embryonic growth and increased incidence of abnormal chicks when applied externally to the eggshell of mallard (*Anas platyrhynchos*) eggs, (Hoffman & Gay, 1981). Maximum wet weight concentrations of these compounds in any sample analysed in the present study were at least ten fold lower.

Comparison of the summed concentrations for sixteen PAHs reported by Shore *et al.* (1999) with the equivalent concentrations in eggs from the present study suggests that total PAH accumulation may be similar across a range of species in Britain. Median summed PAH concentrations in herring gull, cormorant, shag and chough were 29, 71, 228 and 30.6 ng/g lipid, respectively (Shore *et al.*, 1999); equivalent concentrations in gannet, golden eagle and merlin were 47.4, 22.6 and 40.7 ng/g lipid.

The broad similarity between species in patterns of PAH accumulation in their eggs and in the total PAH concentrations suggests that the egg residues that have been measured may be the result of background levels of exposure to diffuse PAH sources. The eggs of species exposed to large point sources, such as birds nesting and feeding on the estuaries of industrially contaminated rivers or alongside busy roads and motorways, might be expected to have higher concentrations of PAHs than those measured in this study. As far as can be judged from the available toxicity data, the concentrations of individual compounds in the eggs analysed in the gannet, golden eagle and merlin eggs are unlikely to have embryotoxic effects. However, there is evidence that simultaneous exposure to multiple PAHs can result in additive toxicity (Brunstrom, 1992) and concurrent exposure to PAHs

and other organic pollutants, such as PCBs, may also result in enhanced toxicity (Wassenberg & Di Giulio, 2004). Thus, comparison of residues in eggs with experimental doses of single PAHs may under-estimate likely toxicity.

Overall, this pilot study has provided preliminary data on PAH concentrations in the eggs of three species of predatory bird in Britain. Regular or periodic monitoring of eggs of these species may provide information on temporal trends in general environmental concentrations of PAHs. While such trends could also possibly be monitored by measuring PAHs in other matrices, such as air or water, a key advantage of monitoring eggs is that they provide a measure of bioavailable PAHs. However, further data analysis would be required to determine what level of effort would be required to detect temporal changes in egg concentrations with an acceptable level of sensitivity. Perhaps more importantly, there is also uncertainty about whether egg concentrations in these species are likely to be sensitive indicators of changes in local or national PAH emissions. Comparison of concentrations in eggs with those in air and/or water across a range of contaminated and uncontaminated sites is needed. This would provide better information on the sensitivity of eggs monitoring to detect changes in environmental contamination.

The value of integrating any PAH analysis into the PBMS depends upon why exactly such monitoring is required. There would be value in monitoring background concentrations if there are concerns that emissions, and associated risks, may be increasing over time, or if there are policy initiatives aimed at reducing national emissions. In the absence of such policy drivers, any PAH monitoring may be better focused on regions and areas where contamination is likely to be greatest, such as in areas containing large point sources. Quantifying the degree of contamination in the eggs of species of high conservation priority in such areas would help determine if PAHs pose a threat to these species. Furthermore, measurement of PAH concentrations in the eggs of these species, and/or in the eggs of other indicator species, would determine whether (and over what spatial and temporal scale) changes in major point source emissions was associated with reduced contamination and risk to wildlife.

Table 8.2: Median concentrations (ng/g lipid) of those PAHs that had median concentrations above the detection limit in one or more species. The rank order for the size of the residue within each species is indicated. Italics indicate the PAHs that occurred in more than 50% of all the samples analysed. Data for the single sea eagle egg analysed is not shown. Data for individual PAH concentrations in each sample are given in Tables 14.12-14.15 of the Appendix.

	Median PAH concentration and rank order in					
	gannet		**golden eagle**		**merlin**	
	median	**rank**	**median**	**rank**	**median**	**rank**
1,3 & 1,7-Dimethylnaphthalene	12.3	1	8.30	2	7.58	5
2,6 & 2,7-Dimethylnaphthalene	12.3	2	10.2	1	9.12	4
1,6-Dimethylnaphthalene	9.40	3	7.10	3	2.94	7
Fluoranthene	8.01	4	6.07	4	2.02	10
Fluorene	7.39	5	5.75	5	7.30	6
2,3 & 1,4-Dimethylnaphthalene	7.13	6	5.01	6	ND	*
1,2-Dimethylnaphthalene	5.98	7	4.82	7	2.05	9
Pyrene	5.18	8	3.50	9	ND	*
2-Methylphenanthrene	4.53	9	ND	*	ND	*
1-Methylphenanthrene	2.76	10	ND	*	ND	*
Triphenylene & Chrysene	2.05	11	1.58	10	0.935	11
Anthracene	1.33	12	0.903	11	2.10	8
Benzo[ghi]fluoranthene	0.68	13	0.710	13	ND	*
Benzo[ghi]perylene	0.635	14	0.610	15	0.220	13
Benz[a]anthracene	0.586	15	0.680	14	0.266	12
Benzo[c]phenanthrene	0.315	16	0.241	18	ND	*
Cyclopenta[cd]pyrene	0.29	17	0.355	17	0.188	14
Dibenz[ah]anthracene	0.118	18	3.75	8	ND	*
2-Methylfluoranthene	ND	*	0.797	12	ND	*
Benzo[b & j & k]fluoranthene	ND	*	0.430	16	ND	*
2-Methylnaphthalene	ND	*	ND	*	38.2	1
1-Methylnaphthalene	ND	*	ND	*	26.8	2
Phenanthrene	ND	*	ND	*	15.0	3

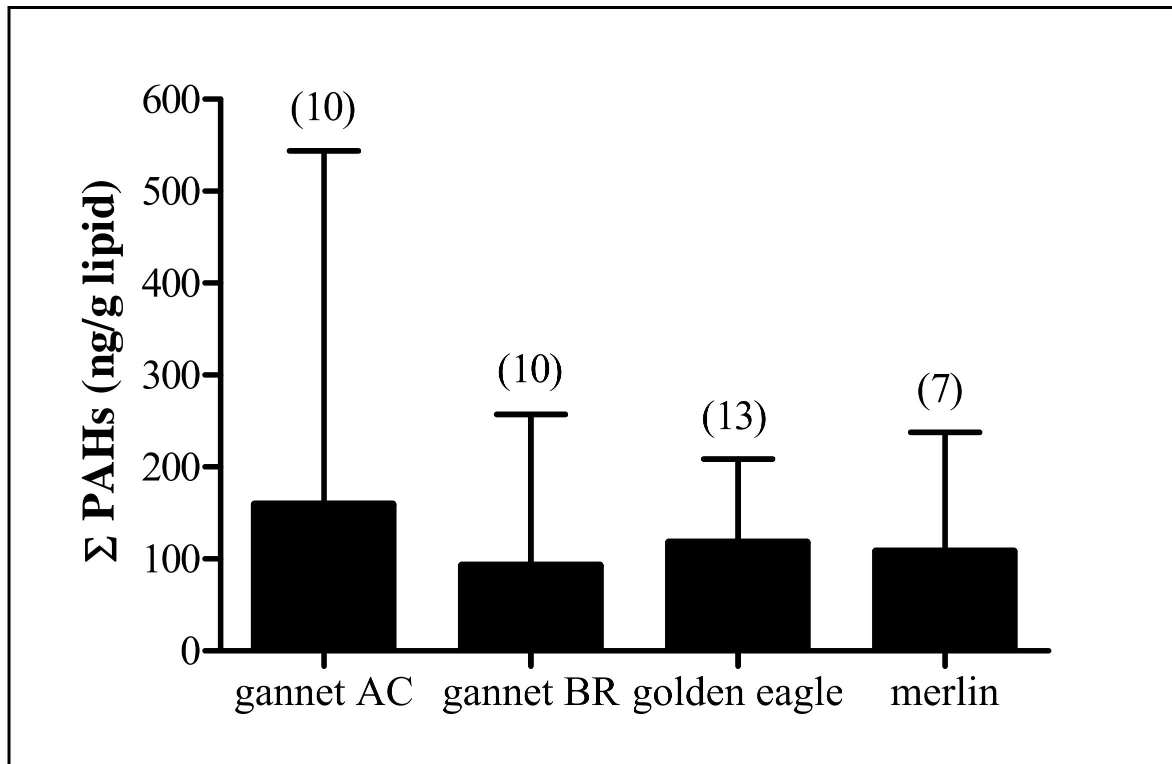
ND: not detected

Table 8.3: Sum concentrations of all determined polycyclic aromatic hydrocarbons (see Table 8.1) in gannet, golden eagle, merlin, and se eagle eggs received in 2002. ng/g lipid weight). Data for each PAH is given in Table 14.12-14.15 of the Appendix.

Egg No/	species	Year	Location	SI[1]	% lipid	Sum all PAHs
G1073	Gannet	2002	Ailsa Craig	*	3.51	88.4
G1074	Gannet	2002	Ailsa Craig	*	4.35	146
G1075	Gannet	2002	Ailsa Craig	*	3.89	78.1
G1076	Gannet	2002	Ailsa Craig	*	4.79	116
G1077	Gannet	2002	Ailsa Craig	*	3.77	475
G1078	Gannet	2002	Ailsa Craig	*	3.69	172
G1079	Gannet	2002	Ailsa Craig	*	3.38	621
G1080	Gannet	2002	Ailsa Craig	*	4.80	21.9
G1081	Gannet	2002	Ailsa Craig	*	3.30	463
G1082	Gannet	2002	Ailsa Craig	*	3.36	612
G1083	Gannet	2002	Bass Rock	2.93	4.37	556
G1084	Gannet	2002	Bass Rock	3.26	2.63	84.1
G1085	Gannet	2002	Bass Rock	2.69	3.90	55.3
G1086	Gannet	2002	Bass Rock	3.05	5.09	316
G1087	Gannet	2002	Bass Rock	2.90	4.59	65.4
G1088	Gannet	2002	Bass Rock	2.66	4.16	35.2
G1089	Gannet	2002	Bass Rock	3.00	5.03	46.8
G1090	Gannet	2002	Bass Rock	3.05	3.94	101
G1091	Gannet	2002	Bass Rock	2.87	3.84	140
G1092	Gannet	2002	Bass Rock	2.62	4.35	198
E7990	Golden Eagle	2002	Argyll	2.84	4.26	38.5
E7991	Golden Eagle	2002	Argyll	2.55	4.27	548
E7993	Golden Eagle	2002	Argyll	3.56	6.29	183.2
E7995	Golden Eagle	2002	Argyll	3.03	3.69	71.2
E7996	Golden Eagle	2002	Argyll	2.87	5.61	147.0
E7998	Golden Eagle	2002	Argyll	2.42	4.69	71.5
E8000	Golden Eagle	2002	Argyll	2.79	5.01	163.1
E8005	Golden Eagle	2002	Highland	3.54	3.92	47.6
E8010	Golden Eagle	2002	Mull	3.11	6.05	234.0
E8011	Golden Eagle	2002	Mull	*	23.72	91.4
E8055	Golden Eagle	2002	North Uist	2.39	3.21	374
E8056	Golden Eagle	2002	North Uist	4.19	4.78	60.3
E8057	Golden Eagle	2002	Skye	5.80	4.68	117.5
E7999	Merlin	2002	North Yorkshire	1.07	6.10	80.2
E8007	Merlin	2002	Lothian	*	5.45	2240
E8008	Merlin	2002	Borders	1.33	5.53	61.5
E8106	Merlin	2002	Northumberland	1.54	6.31	237.5
E8110	Merlin	2002	Northumberland	1.20	4.39	ND
E8115	Merlin	2002	Northumberland	1.33	4.76	108
E8119	Merlin	2002	Northumberland	1.23	5.23	202
E7988	Sea Eagle	2002	Mull	2.83	4.54	44.3

Values for moisture and lipid content were taken from those determined previously in the analysis of organochlorine insecticides, PCBs and mercury

Figure 8.1: **Median (and inter-quartile range) sum PAH concentration (ng/g lipid) in the eggs of gannets from Ailsa Craig (AC) and Bass Rock (BR) and in golden eagle and merlin eggs.**



9 Analysis of unknown compounds in the livers of predatory birds

9.1 Introduction

Gas-chromatography (coupled with electron capture detection—GC-ECD) analysis of the livers of birds of prey submitted to the PBMS has revealed the presence of so-called "unknown compounds". These are detected as peaks on the chromatographic trace with retention times that do not correspond to those of organochlorine insecticides or the congeners present in the Aroclor 1254 standard[2] used to quantify Aroclor-1254 matched PCB concentrations. It is possible that these "unknown" compounds are other PCB congeners that are not present in Aroclor 1254. Indeed, the reporting of "total" PCBs concentrations has assumed that these peaks are PCBs; total PCB concentrations are the sum of the concentrations of all the peaks on the chromatogram other than those that are due to the presence of organochlorine insecticides. On average, Aroclor-1254 matched PCBs have been found to account for approximately 75% of the "total" PCB concentration (Shore *et al.*, 2002), and individuals with the highest liver total PCBs concentrations also have the highest number of "unknown compounds" (Shore *et al.*, 2005c)

It is possible that these "unknown" compounds are, in fact, not PCBs but other organic xenobiotics that may or may not pose a toxic hazard to wildlife. The aim of this particular investigation was to carry out a preliminary investigation on a number of bird livers using gas chromatography coupled with mass spectrometry (GC-MS) to estimate

(i) what proportion of the "unknown" peaks that are considered to be PCBs are indeed likely to be PCB congeners
(ii) where possible to identify what non-PCB compounds may be present in the birds.

9.2 Methods

A total of 13 livers from sparrowhawks and herons were analysed in detail by GC-MS. Sample selection was based on the likelihood of containing "unknown compounds". Unknowns were most likely to be present in samples that had high total PCB concentrations and a high difference between the reported Aroclor-1254 matched PCB and the total PCB concentration. At the time of this study the latest data available was for birds collected in 2002. Liver tissue was analysed from ten birds with potentially high numbers of "unknown" compounds. Liver tissue was also analysed from three birds expected to have low numbers of "unknown compounds", and these acted essentially as analytical controls. Egg contents were not analysed because they contain high levels of cholesterol-like compounds that interfere with the GC-MS analysis.

Samples for analysis by GC-MS were extracted and cleaned up using a modification of the sample preparation method used for the analysis of bird of prey samples by GC-ECD which has been described in detail previously (Wienburg & Shore, 2004). The modifications were:

[2] Aroclor-1254 is a commercial mixture of individual PCB congeners used in some manufactured goods and processes. Matching congeners in tissue samples with those found in the Aroclor mixture is one recognised means of providing a quantitative measure of total PCBs in a sample

(i) carbon-13 labelled PCB surrogate standards were added prior to extraction.
(ii) the sample extract was not reduced to dryness after extraction.
(iii) after alumina column chromatography, the extract was reduced in volume by a factor of five
(iv) Kurdena Danish flasks were used for reducing the sample extract volumes.

Analysis of PCBs by GC-ECD was carried out as described in detail elsewhere (Wienburg & Shore, 2004). Compounds were identified by matching the retention times of chromatographic peaks from samples against the retention times of compounds in analytical standards. Standards were run for a suite of organochlorine pesticides (so that they could be identified and then discounted from this particular analysis), for Aroclor 1254, and for seven other PCB congeners. These other congeners (numbers 29, 77, 126, 169, 205, 206, 209) are routinely determined in the PBMS analysis but were not detected in our Aroclor 1254 standard. They are referred to as congener-matched PCBs for the purposes of this report. Chromatographic peaks that were not identified as organochlorine insecticides, Aroclor-1254 matched PCBs or congener-matched PCBs were classed as unknowns. Total PCB concentration, as calculated for the time-series data for the PBMS, is the sum of the Aroclor-1254 matched PCBs, the congener-matched PCBs, and the unknowns (concentrations for unknowns are calculated assuming that the response ratio is the same as for Aroclor-1254).

Samples were analysed by GC-MS using a system that comprised: an Agilent 6873 autosampler, Agilent 6890 gas chromatograph and Agilent 5973 Mass Selective Detector (MSD) with electron impact ionisation. The same capillary column was used as for the GC-ECD analysis and parameters affecting separation were adjusted to give retention times, elution orders and peak resolutions that were as close as possible to those given by the GC-ECD. PCBs were separated on a 50m x 0.22mm id x 0.25 micron HT8 column (SGE UK Ltd) fitted with a 5m x 0.22mm id methyl deactivated retention gap (SGE UK Ltd). The helium carrier gas was maintained at a constant pressure to give an initial flow of 0.9 ml min^{-1} (50°C) by electronic pressure control. Samples (60µl) were injected onto the column via a Programmable Temperature Vaporising inlet in solvent vent mode. The column oven temperature was held at 50°C for 5.18 minutes, then ramped at 45°C min^{-1} to 200°C, held for 5 minutes, then ramped at 1°C min^{-1} to 260°C, then ramped at 50°C min^{-1} to 320°C, held for 10 minutes, then ramped at 50°C min^{-1} to 325°C and held for 10 minutes. For analysis of PCBs the MSD was operated in select ion mode (SIM) at the highest resolution available. For the identification of compounds other than PCBs, the MSD was operated in scan mode (30 to 700 amu). To minimise any variation in retention time between scan and SIM analysis of the same sample that might be caused by column contamination, the SIM analysis of each sample immediately followed the scan analysis.

The analysis of PCBs by SIM was designed to enable the detection of all 209 possible congeners. The time periods for the collection of target and qualifier ions were based on: (i) GC-MS scan analysis of Aroclors 1242, 1248, 1254 and 1260, (ii) elution order data supplied by the column manufacturer for all 209 PCB congeners; (iii) individual congener standards. This method of estimating the start and stop times for the elution of PCBs of different chlorination levels carried some uncertainty and so the ion collection times were extended to minimise the chance of missing some congeners. Two ions were collected for PCBs with one and two chlorines and three ions were collected for PCBs that had between three and ten chlorines. Integration of peak areas was carried out using the Agilent Chemstation software. Peak areas and retention times were then exported to in-house software for the identification of PCBs. In summary the identification method was:

- any peak that had ion ratios that matched those found in the Aroclor standard and in the congener standard was classed as a PCB.
- peaks identified as PCBs were then examined for co-elutions and interference from ions produced by PCBs of higher chlorination level.
- PCBs were then classified by chlorination level.
- chlorination level and relative retention time (to the nearest 13C surrogate) were then used to identify the PCBs that matched those in the congener standards or in the Aroclor 1254 standard.
- Peaks identified as PCBs but that were not identified as congeners or matched PCBs were

classed as other PCBs.
- The maximum allowable difference in retention time between target and qualifier ions was set at 0.1%. The upper and lower limits for the qualifier ion ratios and relative retention time windows were 20% and 0.2% respectively. The peak area of the target ion was used for calculations.

Unknown compounds were identified by using the National Institute of Standards and Technology (NIST) spectral library. The spectral search software gives an estimate of the quality of the match between unknown and library spectra (100% is a near perfect match). Search results with a quality of over 80% have been reported.

9.3 Results

The analysis of PCBs by GC-MS was sufficiently good to detect PCBs matched to Aroclor 1254 and to specific congeners with sensitivity equivalent or better than that of the GC-ECD. Interferences with the analyses only occurred with some of the lowest chlorinated PCBs and there was no significant interference with the analysis of higher chlorinated compounds. Furthermore, the GC-MS can distinguish between co-eluting or closely eluting OC pesticides and PCBs and so some individual PCBs were detected by GC-MS but not GC-ECD.

The absolute retention times of individual compounds were different on the GC-ECD and GC-MS. Although there was a relationship between the retention times recorded by the two analytical methods, this was not sufficiently robust to match closely eluting peaks on the GC-ECD to compounds detected by GC-MS. In addition, if there were any elution order differences between the two methods, these could not be detected with confidence. Thus, it was not possible to try and match each individual peak from the GC-ECD and GC-MS chromatograms for a specific sample. The analysis of the results therefore focused on comparing the number of Aroclor 1254-matched and congener-matched congeners detected by the two analytical methods and an assessment of what proportion of unknowns were likely to be other PCB congeners.

In samples with overall high concentrations of total PCBs, analysis by GC-MS detected significantly more individual chromatographic peaks that were ascribable to Aroclor 1254 congeners than analysis by GC-ECD (student t test: $t_{(14)} = 3.32$, $P<0.01$; Figure 9.1, upper graph). As would be expected, the total number of Aroclor 1254 PCB congeners in the three samples with low PCB concentrations was smaller than in samples with high PCB concentrations, irrespective of the method of detection. However, GC-MS analysis detected almost four times as many Aroclor 1254 PCBs than the GC-ECD analysis of these low concentration samples (Figure 9.1).

The results from the analysis of the samples that contained either high or low PCB concentrations suggest that analysis was more sensitive when done by GC-MS than GC-ECD in terms of detecting individual PCB congeners present in Aroclor 1254. This may have because the limits of detection were better for GC-MS and/or co-eluting peaks that were not clearly distinguishable by GC-ECD were identified by GC-MS.

GC-MS and GC-ECD analysis both detected a number of congener-matched PCBs (Figure 9.1). The two analytical techniques detected similar numbers of these congeners in samples with high PCB concentrations but detection was better by GC-MS for samples with low PCB concentrations (Figure 9.1). This again suggested that the limits of detection for PCB congeners were better for GC-MS than GC-ECD.

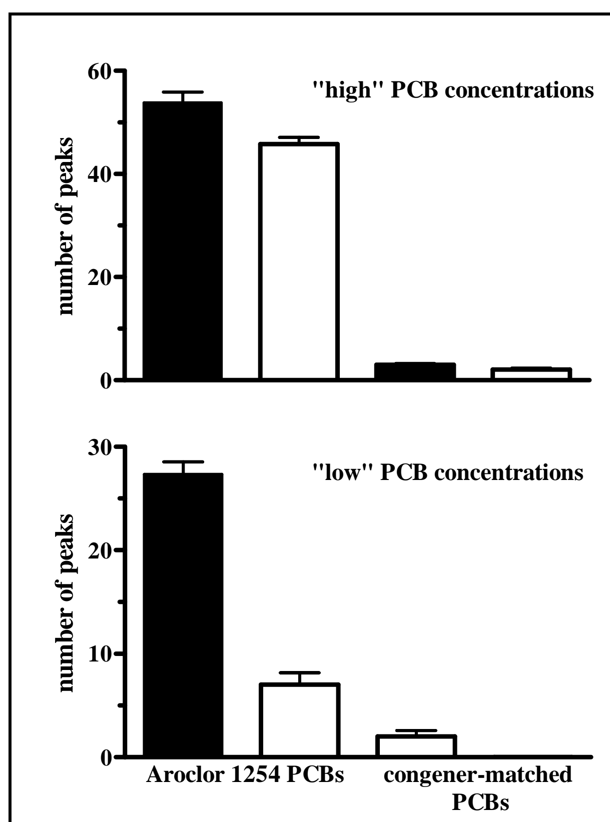


Figure 9.1: Mean ± SE number of peaks that were assigned to Aroclor 1254 PCBs and to congener matched PCBs (see text for details) as detected by GC-MS (filled histograms) and GC-ECD (open histograms) in livers that had "high" and "low" total PCB concentrations

The remaining unassigned peaks on the GC-ECD chromatogram were the unknowns. Tissues with high and low PCB concentrations had, on average, 36 and two unknown peaks, respectively (Figure 9.2). This closely corresponds to previous analyses of sparrowhawk livers with high total PCB concentrations in which the number of unknown peaks was approximately 40 (Shore *et al.*, 2005c).

The mean number of PCBs detected by GC-MS that were neither Aroclor 1254 nor congener-matched PCBs was 15.8 and 6.0 in high and low PCB concentration samples, respectively (Figure 9.2). Thus, in high PCB concentration samples, on average just under half of the unknown peaks detected by GC-ECD were likely to be other PCBs. In low PCB concentration samples, GC-MS analysis detected more non-Aroclor 1254 and non-congener matched PCBs than there were unidentified peaks detected by GC-ECD, and it is possible that most or all of the unknowns in these samples could have been other PCBs.

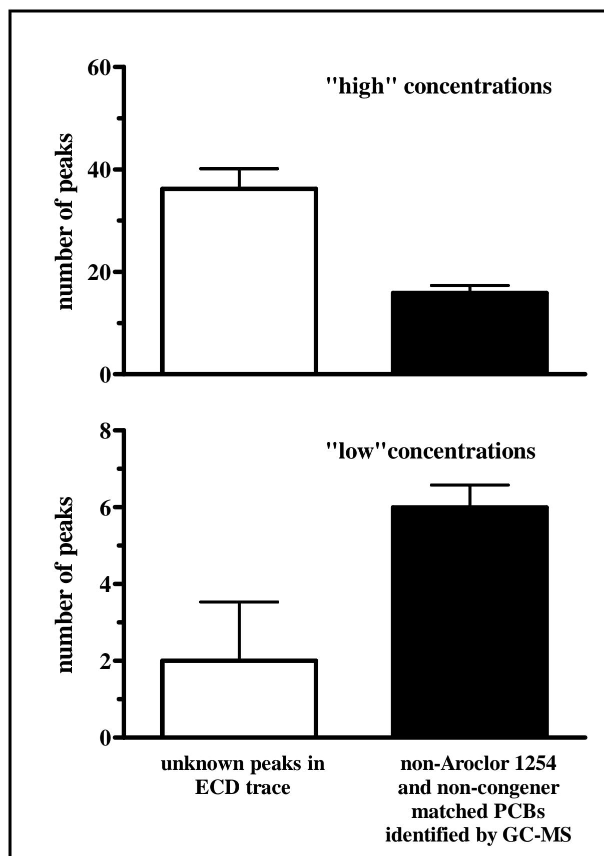


Figure 9.2: Mean ± SE number of unknowns detected by GC-ECD analysis (open bars) in livers with high and low PCB concentrations, and mean ± SE number of non-Aroclor 1254 and non-standard matched congeners identified by GC-MS (filled bars) in these samples.

When the number of GC-MS identified non-Aroclor 1254 and non-congener matched PCBs were plotted against the number of unknown peaks from the GC-ECD trace, it was apparent that there was no relationship between the two (Figure 9.3a). The number of non-Aroclor 1254 and non-congener matched PCBs that were identified by GC-MS remained constant in samples with high PCB concentrations, even though the number of unknowns on the GC-ECD traces varied more than two-fold. Overall, the percentage of unknowns in livers with high PCB concentrations that were likely to be other PCBs varied from between 80% in samples with 20-30 unknown peaks to about 40% in samples with the highest numbers of unknowns (Figure 9.3b).

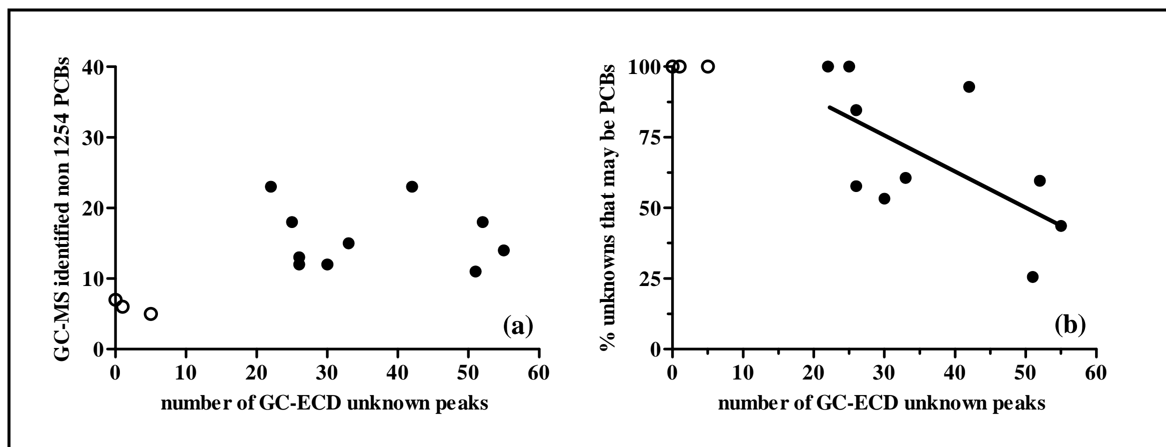


Figure 9.3: (a) Number of non-Aroclor 1254 and non-congener matched PCBs identified by GC-MS plotted against the number of unknowns in the sample as identified by GC-ECD; (b) % of unknowns that may be non-Aroclor 1254 and non-congener matched PCBs plotted against the total number of unknowns in the sample. Filled circles represent high PCB concentration samples, open circles low PCB concentration samples. The significance of the regression line plotted in graph (b) for high PCB concentration samples only is: R^2 = 0.405, $F_{(1,8)}$ = 5.451, $P<0.05$.

Spectral libraries were used in conjunction with the GC-MS in scan mode to determine whether any of the unknowns not identified as PCBs could be tentatively identified. Mirex, heptachlor epoxide, 1,1-dichloro-2,2-bis(p-chlorophenyl)ethylene (DDMU) and between one and three isomers of nonachlor were detected in one, two , five and nine livers, respectively; all were livers that had high levels of PCBs. Mirex is an organochlorine insecticide used as a stomach insecticide, mainly formulated into baits, for the control of ants, and as a flame-retardant; it is highly persistent in the environment (WHO, 1984). The other compounds are biotransformation products of one or more organochlorine insecticides and may be present in livers as a result of *in vivo* metabolism by the bird or due to exposure to environmental concentrations of the metabolite itself.

In conclusion, the results of the analysis to date suggests that in many of the bird livers typically analysed for PCBs as part of the PBMS, most (> 60%) of the unidentified peaks present on the GC-ECD trace are likely to be PCBs not present in Aroclor 1254. Other compounds that are present include metabolites of organochlorine insecticides. However, in livers with particularly high "total PCB" concentrations, more than half of the unknown peaks may be due to the presence of other compounds. Although their identity and potential toxicity is unknown, these compounds are likely to be relatively non-polar and insoluble in water; this is because of the extraction and detection methods that were used. However, it is also possible that predatory birds are exposed to a variety of other, more polar, xenobiotics that would not be detected using GC-ECD detection. Thus, while the results of the present study have advanced our understanding of chemical exposure and risk in predatory birds in the UK, they do not provide a comprehensive assessment of exposure to all environmental contaminants. This would require a much more detailed and sophisticated analysis that employed a range of analytical techniques. A possible approach to such an analysis has been described by Shore *et al.* (2005c)

10 Analysis of spatial variation of liver PCB concentrations in terrestrial predatory birds

10.1 Introduction

Liver total PCB concentrations in sparrowhawks, kestrels and herons are typically characterised by large scale variation between individuals. The causes of this variation are multiple. One likely key factor is geographical variation in dietary PCB concentrations; regional variation in environmental concentrations of other contaminants in birds monitored by the PBMS has been reported previously (eg., Erry *et al.*, 1999; Newton *et al.*, 1999a; Sparks *et al.*, 1999). Although PCBs are known to transported globally in the atmosphere (Bard, 1999; Sweetman *et al.*, 2002; Vallack *et al.*, 1998), local sources are also likely to be important; such sources include waste disposal and landfill sites and open sources of PCBs such as plastics, paints and adhesives (Hoffman *et al.*, 2001; Johnson *et al.*, 1996), especially in urban areas where the density of such sources may be high. Because the PBMS chemical monitoring of carcasses has national geographical coverage, it may prove a useful indicator of hotspots of PCB contamination in Britain that are due to major localized or regional sources.

A key problem in determining the importance of local sources of contamination is in devising statistically robust ways of identifying contamination hotspots. We previously reported a novel way of carrying out such analyses and, using a limited span (1993-1997) of data on liver PCB concentrations in sparrowhawks and kestrels, identified a small number of potential hotspots of PCB contamination (Broughton *et al.*, 2003; Shore *et al.*, 2005c). The aim in this current study was to build on our initial analyses and examine the whole time span of PCB monitoring data accumulated for sparrowhawk and kestrel carcasses to determine whether, using this much larger data set, statistically robust and consistent geographical hotspots of PCB contamination could be identified. Such information is essential for understanding the processes by which contaminants are distributed in and move through the environment and is vital for formulating effective mitigation strategies. A further aim was to determine whether liver PCB concentrations also varied with larger-scale geographical factors, such as latitude and longitude, rainfall and land cover.

10.2 Methods

The data used in this analysis was the PBMS long-term data for total PCB concentrations in sparrowhawks and kestrels. These long-term data were most recently described by Shore *et al.* (2005a), and cover the period 1963-2002.

Initially, spatial analysis of the liver PCB concentrations in sparrowhawks and kestrels was performed using the same statistical technique as described by Broughton *et al.* (2003), which looked for clusters of high PCB concentrations in birds from the same area. The area that was searched for clusters had a radius that increased at 1 km incremental steps from a minimum of 5 km to a maximum of 25 km. The only difference in the methodology was that clusters were identified on the basis of average PCB concentrations within search radii as opposed to totals (summed value of PCB concentrations for all birds within the search radii). This methodology was changed because the use of totals implies that birds with low concentrations are not important and could have come from anywhere whereas the use of average values implicitly assumes that all birds are equally important and equally as likely to have come from that location.

Semi-variograms were also constructed for both sparrowhawks and kestrels. A semi-variogram is graph where the x-axis represents how far apart each pair of points is, and the y-axis provides a measure (half the variance) of how different they are. With a "well behaved" variable, like ground height, points that are close together will have very similar values; points that are further apart will have values that are more likely to be more different. The graph will therefore show an upward trace towards some maximum value. If the liver PCB data for sparrowhawks and kestrels conformed to such a pattern, it would suggest that birds from similar locations had similar liver PCB concentrations, as would be expected if there was a strong relationship between geographical location, exposure and associated liver PCB concentration.

The variation in liver PCB concentrations was also related to the degree of urbanisation in the area around which the bird was found, and to macro-scale geographical factors, namely north-south (latitudinal) and east-west (oceanic-continental) variation. The measure of urban land area was extracted from the Land Cover Map of Great Britain (Fuller *et al.*, 1994) and consisted of the number of pixels of land class 21 (continuous urban) in the 2 km^2 (kestrel) or 10 km^2 (sparrowhawk) around where the carcass was found. The choice of 2 km^2or 10 km^2 was based on the likely foraging areas for kestrels and sparrowhawks, as used by Broughton *et al. (*2003), and assumes that each bird was found in the centre of its foraging territory. The pixel count is an indication of the intensity of land use but not a precise tally of land cover. Finally, liver residues were related to annual rainfall averaged over the period 1961 to 1990 and interpolated from Meteorological Office stations to a 5km resolution using thin plate splines (Barrow *et al.*, 1993)

10.3 Results

Spatial analysis at the micro-scale of total liver PCB concentrations in all the sparrowhawks monitored by the PBMS indicated the presence of a statistically significant cluster at a search radius of 15-20 km. This cluster was located on Merseyside and was due to very high residues in three birds analysed by the PBMS that were from this location. The cluster was essentially the same one that had been identified previously when only the data for the period 1993-1997 had been analysed (Broughton *et al.*, 2003). No other clusters were detected. Unlike the results for sparrowhawks, there were no statistically significant clusters identified for liver total PCB concentrations in kestrels.

The semi-variograms that were constructed (data not shown) for liver residues in sparrowhawks and kestrels did not exhibit the upward pattern that would be expected to occur if individuals from similar locations had similar liver PCB concentrations. This suggests that the distribution of residues in both sparrowhawks and kestrels is, in general, essentially unrelated to specific location.

There was a wide range of liver PCB concentrations across areas with varying degrees of urbanisation (Figure 10.1). A Spearman Rank correlation analysis was carried out to determine if there was any evidence for an association between the number of urban pixels and liver total PCB concentration; a ranked approach was used to avoid making any assumptions about the exact nature of any potential relationship. There was no evidence of any significant association between urban land cover and liver PCBs in sparrowhawks ($r_s = -0.008$, $P = 0.731$, $n = 1812$) but there was a positive relationship for kestrels ($r_s = +0.058$, $P = 0.043$, $n = 1229$), although this was extremely weak and only just achieved statistical significance.

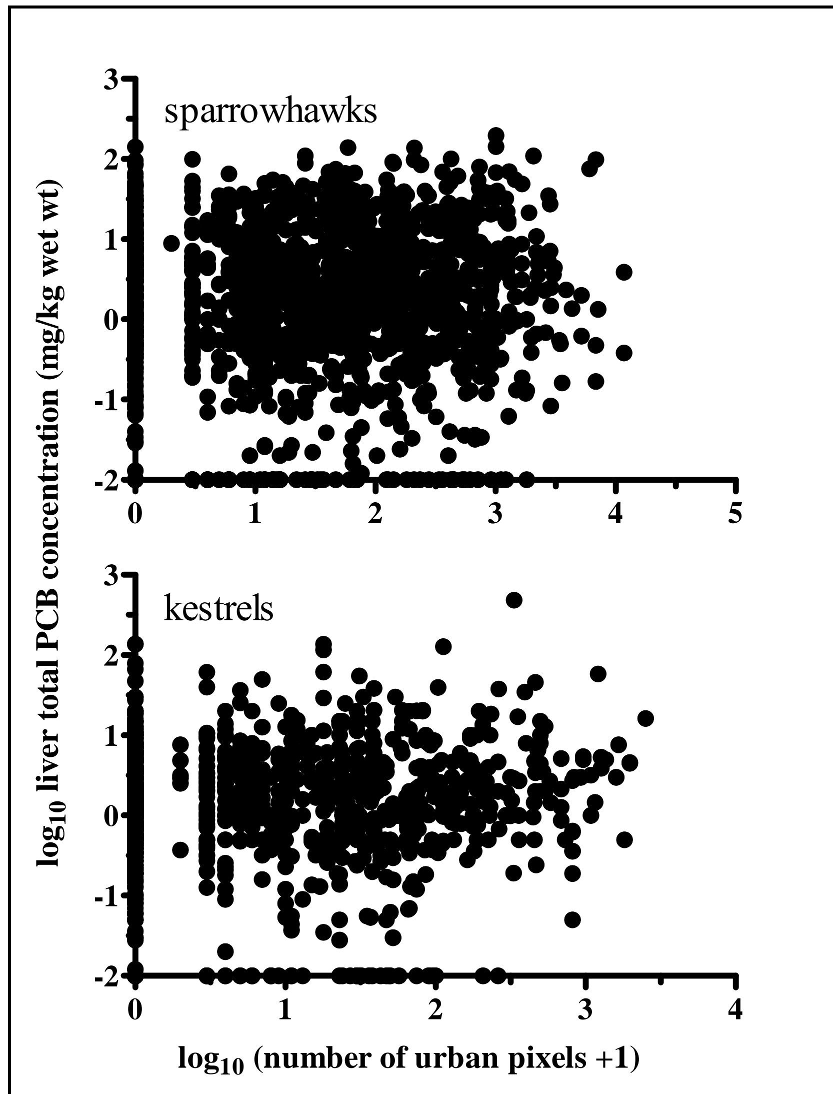


Figure 10.1: Liver PCB concentration in dead sparrowhawks and kestrels plotted against the number of urban pixels in the area surrounding where the carcass was found. Data are shown as log values for ease of visualization. Non-detected PCB concentrations were assigned a value of 0.01 mg/kg wet wt ($\log_{10}$ value of -2).

When liver PCB concentrations were related to the geographical position at which carcasses were found (Figure 10.2), there was no relationship between easting and the magnitude of the liver PCB concentration in either sparrowhawks ($r_s = 0.0007$, $P = 0.976$, $n = 1812$) or kestrels ($r_s = 0.027$, $P = 0.341$, $n = 1229$). Northings were likewise unrelated to liver PCB residues in kestrels ($r_s = 0.0025$, $P = 0.931$, $n = 1229$) but there was a weakly significant negative relationship between northings and liver PCBs in sparrowhawks ($r_s = -0.052$, $P = 0.0269$, $n = 1812$). This suggested that liver total PCB residues in sparrowhawks tended to be lower in birds that died further north.

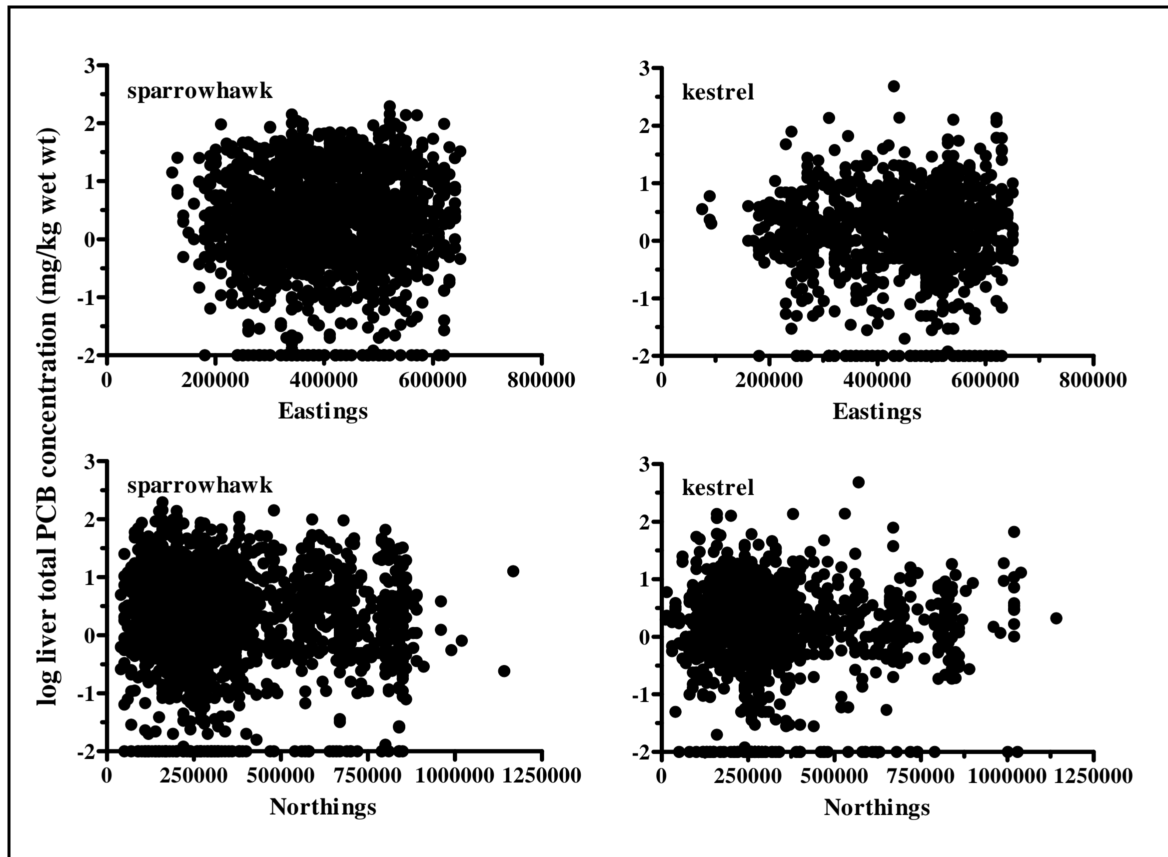


Figure 10.2: Liver PCB concentration in dead sparrowhawks and kestrels plotted against the Easting and Northing of the location where the carcass was found. PCB concentrations are shown as log values for ease of visualization and non-detected values were assigned a value of 0.01 mg/kg wet wt (log_{10} value of -2).

Liver PCB concentrations also varied with rainfall (Figure 10.3). There was a significant positive relationship between average annual rainfall and liver PCB concentrations in sparrowhawks ($r_s = 0.067$, $P = 0.005$, $n = 1779$), which indicated that liver PCBs were generally higher in sparrowhawks that died in parts of the country that had on average higher annual rainfall. A similar trend was apparent in kestrels but was not statistically significant ($r_s = 0.044$, $P = 0.132$, $n = 1185$). However, for both species, the amount of variation in liver PCB concentrations that was associated with variation in rainfall was small.

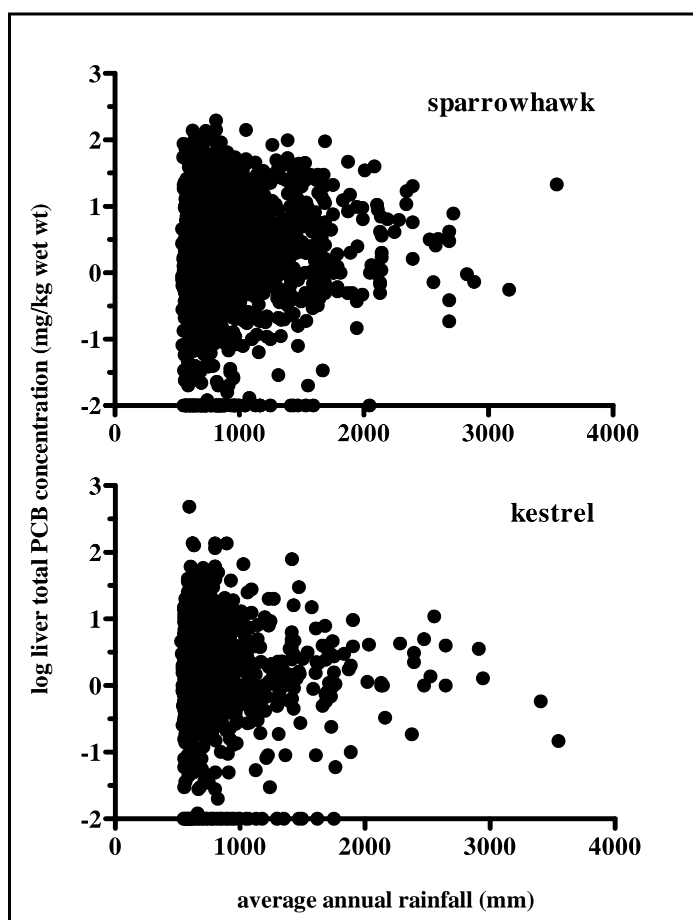


Figure 10.3: Liver PCB concentration in dead sparrowhawks and kestrels plotted against the annual average rainfall (see text for details). PCB concentrations are shown as log values for ease of visualization and non-detected values were assigned a value of 0.01 mg/kg wet wt (log_{10} value of -2).

10.4 Discussion

Overall, the results of the micro-spatial analyses suggest that there is little evidence of geographical clusters or hotspots of PCB contamination in sparrowhawks and kestrels in Britain. The only cluster detected was for sparrowhawks was on Merseyside. This is plausible given a large PCB landfill site is located in this area and has been shown to result in elevated PCB concentrations in small mammals (Johnson *et al.*, 1996). However, it was surprising the same hotspot was not identified in kestrels which, unlike sparrowhawks, feed predominantly on small mammals. In our initial analysis of a subset of data for kestrels that covered the period 1993-1997, a number of clusters were identified, including the Merseyside hotspot (Broughton *et al.*, 2003). The loss of these clusters once the data for the whole PBMS monitoring period was examined largely reflected the fact that birds from cluster areas that had died before 1993 did not have particularly high liver PCB residues.

The “disappearance” of clusters, caused by including “pre-1993 birds” in the analysis, could be taken to indicate the emergence of hotspots of *new* emissions. However, given that PCB manufacture ceased well before 1993 and PCB concentrations in predatory birds and their eggs in Britain have generally either remained stable or declined since the 1970s (Shore *et al.*, 2005a), such an explanation seems unlikely. It is possible, however, that as environmental concentrations of PCBs generally decline over time, localised hotspots, where emissions are declining more slowly than in the general environment, may become increasingly apparent. Given the environmental persistence of PCBs, it might be expected that, once identified, hotspots should remain robustly detectable over time. Although this concept could theoretically be tested by dividing the data analysis into distinct time periods (such as different decades), the probability of detecting any cluster would be reduced because the number of birds collected from each potential hotpot may be very small in any given decade. It also cannot be ruled out that some hotspots may be analysis artefacts because clusters may be more likely to be identified when the sample consists of few birds.

Further work would be needed to test the robustness and longevity of PCB hotspots in sparrowhawks and kestrels. However, the analysis reported here suggests that hotspots are relatively rare. It is possible that localised environmental PCB contamination does not significantly enhance liver PCB concentrations in predatory birds at all; transportation and deposition of PCBs from globally diffuse sources may be an equally or a more important source of contamination than local inputs. Certainly the analysis presented here suggests the PBMS is unlikely to prove a robust biomonitoring tool for detecting localized hotspots of environmental PCB contamination.

The results of the wider scale spatial analysis suggest that some of the variation in liver total PCB concentrations may be explained by certain geographical factors. The association between rainfall and liver PCB concentrations appeared to be common to both sparrowhawks and kestrels (although only statistically significant for sparrowhawks), and may be related to deposition rates of atmospheric PCBs. An association between rainfall (thought to be related to atmospheric deposition) and bioaccumulation of mercury has been demonstrated in a top mammalian predator, the Eurasian otter *Lutra lutra* (Kruuk *et al.*, 1997). However, we currently have no proof that rainfall is a proximate factor in determining liver PCB concentrations. For example, it is possible that prey availability or hunting success could be negatively associated with amount of rainfall and birds in wetter areas are more likely to starve; liver PCB concentrations are elevated in starved kestrels and sparrowhawks (Wienburg & Shore, 2004).

The other geographical factors detected as being associated with variation in liver PCBs were latitude for sparrowhawks and urban land cover for kestrels. Why and how latitude may directly or indirectly affect liver PCB concentrations in sparrowhawks, but not kestrels, is unknown. The positive association between urban land class and liver PCB concentrations in kestrels could indicate that local industrialised sources elevate PCB accumulation to some extent. This would be expected to be more evident in kestrels, which predominantly take small mammal prey that are relatively sedentary and are likely to reflect local contamination, than in sparrowhawks that feed over wider areas and take small birds that themselves are relatively mobile. However, extreme caution is needed when making such interpretations. It is also possible that kestrel prey items are more scarce in urban than non-urban areas, and elevated liver PCB residues in urban birds are simply an indication that starvation [and associated increased liver contaminant levels] is more prevalent in urban kestrels.

In conclusion, this study indicates that geographical hotspots of PCB contamination are rarely detected in the sparrowhawks and kestrels examined by the PBMS, but other larger-scale geographical factors are associated with variation in liver PCB concentrations. However, the extent of this association is limited. The correlation coefficients for rainfall, land cover and latitude, although statistically significant, were consistently low. In contrast, the nutritional state of the bird at the time of death appears to be a much more important factor influencing liver PCB concentrations. Recently, we examined the importance of intrinsic physiological factors (nutritional state, age, sex) in explaining recent and current intra-species variation in liver PCB concentrations in predatory birds in Britain (Wienburg & Shore, 2004). This involved analysing data on PCB liver concentrations in the sparrowhawks, kestrels and herons monitored as part of the PBMS. The study found that, in birds

examined between 1992 and 1997, body condition was the single most important intrinsic factor affecting liver PCB concentrations. It accounted for up to 49% of the variation in liver residues in the three species; starved birds had the highest liver concentrations. Age, sex and season were also significant factors, although of lesser importance. Given this, we would recommend that further analysis should focus on how nutritional state, rather than geographical location, influences the detection of long-term trends in liver PCB concentrations in sparrowhawks and kestrels at a national scale.

11 Summary report of the role of the PBMS in monitoring decabromodiphenylether (DBDE) concentrations in predatory birds

11.1 Introduction

Decabromodiphenylether (DBDE) is one of the most commercially important (in terms of production and use) of the brominated diphenyl ethers, a series of compounds that range from mono- to DBDE. It has been used as an additive flame retardant in many plastics, especially high-impact polystyrene, and in the treatment of textiles used in soft furnishing, automobile fabrics, and tents. Global production was estimated to be as high as some 30 000 tonnes/year (WHO, 1994). DBDE is not readily extracted from polymers, binds to particulate matter, and is poorly absorbed across the gastrointestinal tract. Therefore, the bioaccumulation potential of this compound has been assessed as low (WHO, 1994). There is little information on the toxicity of DBDE to organisms in the environment.

Although DBDE was not expected to bioaccumulate, it was detected in the eggs of a top predator, the peregrine falcon, in Sweden (Lindberg *et al.*, 2004). This finding triggered a subsequent study, initiated the UK Environment Agency and the Bromine Science and Environmental Forum (BSEF). The aim of this new study was to determine the presence and time trends of DBDE residues in predatory birds and so provide environmental information needed for the European environmental risk assessment of DBDE.

Although many of the samples analysed in this new study were from the PBMS frozen tissue and egg archive, the project did not formally form part of the activities covered under the annual Wildlife and pollution project. The analytical work was funded by the BSEF (representing the producers of DBDE), in response to a data request under the Existing Substances Regulation (EC no. 793/93) and the analytical chemical analysis was conducted by The Centre for Environment Fisheries and Aquaculture Science (CEFAS) in the UK and by the Netherlands Institute for Fisheries Research (formerly RIVO). The use of the PBMS archive, however, means that the outcome of the work are likely to be of interest to all stakeholders of the PBMS, so the results of the study are briefly summarised below.

11.2 Summary of DBDE monitoring study results

The first phase of the DBDE study involved analyzing 135 samples (muscle, liver, eggs) from various species of predatory birds. Many of the samples were from the PBMS frozen tissue and egg archive. DBDE was detected in the eggs of peregrine falcons from the UK, although generally at lower concentrations than in peregrine falcon eggs from Sweden. DBDE was also detected in the muscle and liver of peregrine falcons found dead and submitted to the PBMS. Furthermore, accumulation of DBDE was not restricted to peregrine falcons. DBDE was detected at relatively low concentrations in approximately a third of the other samples that were analysed; these included PBMS archived samples of eggs, muscle and liver from a range of terrestrial species from the UK (barn owl, kestrel, merlin, Montagu's harrier *Circus pygargus,* red kite and sparrowhawk). DBDE was also found in samples from some aquatic predatory birds (heron, great crested grebe, sea eagle) but was not detected in the majority of samples from these species. It was not detected in the tissues or eggs of golden eagles, marsh harriers *Circus aeruginosus*, ospreys *Pandion haliaetus*, or gannets from the UK (all PBMS samples).

The presence of DBDE in a wide range of birds indicated that this compound is bioavailable to higher trophic species. The relatively low concentrations that were detected suggested that bioaccumulation may be limited in extent. DBDE concentrations were generally higher in samples from terrestrial than aquatic species. This suggested that bioaccumulation of DBDE in aquatic food chains, apart from a possible uptake through the gut, is unlikely. Finally, comparison of residues in each type of sample matrix indicated that DBDE concentrations were relatively similar in eggs and tissues, and so either sample type might be useful for monitoring. Amongst the PBMS samples that were analysed, DBDE tended to be detected most frequently in samples from sparrowhawks and peregrine falcons.

In the second phase of the study, muscle samples from 64 sparrowhawks and 48 peregrine falcon eggs (all from the PBMS archive) were analysed. The samples had been collected at approximately five-year sampling intervals between 1973 and 2001 and were analysed to provide a time trend of DBDE contamination in predatory birds. The data for these samples were combined with those from sparrowhawk and peregrine falcon samples analysed in the first phase of the study.

The proportion of peregrine falcon eggs that contained detectable concentrations of DBDE doubled between the period 1973-1986 and the period 1987-2001. However, this rise did not appear to continue through the 1990s and beyond. During the 1987-2001 period, the proportion of eggs contaminated with DBDE did not increase progressively with each five year sampling interval. In terms of DBDE concentrations in peregrine falcon eggs, DBDE concentrations in eggs varied significantly between five year sampling intervals over the whole 1973-2001 period but there was no progressive increasing or decrease over time.

The pattern for DBDE in sparrowhawk muscle samples was broadly similar to that reported for peregrine falcon eggs. The proportion of sparrowhawk samples contaminated with DBDE increased significantly between 1975 and 2001. This appeared to be a stepwise increase (from 7% to 52%) that occurred sometime between 1985 and 1990 rather than any progressive increase. During the later time period (1990-2001), the proportion of sparrowhawks contaminated with DBDE did not increase significantly with each five year sampling interval. In terms of DBDE concentrations in sparrowhawks, median DBDE concentrations in birds with detectable concentrations approximately doubled between 1990 and 2001 but this increase was not statistically significant.

As a result of the outcome of this work, the BSEF, in consultation with the Environment Agency, has agreed that further studies on long-term trends of environmental concentrations of DBDE are needed. Such work should include monitoring concentrations in predatory birds. The BSEF have therefore initiated a monitoring programme that includes the analysis of addled sparrowhawk eggs collected through volunteer collectors that submit failed eggs to the PBMS. The PBMS has therefore contacted all of its regular submitters of failed eggs to ask them to submit failed sparrowhawk eggs to the PBMS wherever possible.

A scientific paper reporting the full details of both phases of the study is currently being prepared by the BSEF, in collaboration with tall participants of the study. The use of the tissue and egg archive of the PBMS in this project emphasises the international value of this resource for assessing chemical risks to predatory birds and other wildlife.

12 Cataloguing of the PBMS tissue and egg archive

12.1 Introduction

As part of the PBMS core activities, liver, kidney, brain, muscle, fat, and gizzard contents have been collected (where available) from the carcasses sent in to CEH. These samples have been stored at -20°C to form a tissue archive. We have recently started to archive bone and feather samples as well. Egg contents from the eggs submitted to the PBMS are also stored in the deep freeze archive. Archiving of material from the PBMS has been carried out since the scheme began but the majority of samples date from the 1970s onwards. The cost of accumulating this archive can be estimated from the current costs of running the PBMS and the duration of the scheme and to date is approximately £4 million.

There are three primary uses for the archive:

(i) t*o develop new monitoring*. Pilot studies using archived material can determine whether monitoring of new chemicals, or new bird species for chemicals already of concern, is merited (Shore *et al.*, 2000, 2001). Analysis of stored material also provides information on which tissues accumulate the greatest concentrations of any new chemical of interest, and so identify which tissues will be most suitable for future monitoring of this contaminant (eg., see section 11 of this report). Such analyses also provide a quantitative assessment of the variability in measured concentrations. This information is needed to determine the sampling intensity needed in any new monitoring programme and the associated power that the programme will have to detect spatial and temporal changes in contamination.

(ii) *for retrospective applied and fundamental studies* that can be used to refine our existing monitoring methodologies and to inform us about the risks posed by a range of contaminants that may be accumulated by predatory birds (for example, Erry *et al.*, 1999; Shore *et al.*, 2001; Shore *et al.*, 2006).

(iii) *contributory material for other studies* run by stakeholders and research colleagues. To date these have included studies on genetic variation in certain species and recently, initiatives to genetically profile captive-bred birds and so assist in the detection of illegal trading in wild-born individuals.

Unique traceable codes are assigned to each archived tissue and egg sample so that the sample can be related back to the source information held on each carcass or egg. Therefore, it is known from our records of tissue and eggs that have been submitted to the PBMS approximately what samples are held. However, as the archive has expanded and been used over the course of the PBMS, it has been relocated in different freezers and samples have been used partly or completely for one-off studies. Records have not been systematically kept as to how much of each sample has been used, or remains in the freezer, nor is the exact location of each sample within the freezer known. Thus, management of the archive resource and easy accessing of the samples can be problematic and time consuming.

To overcome these difficulties, an initiative to catalogue the PBMS archive has begun. The aim is to produce a readily interrogated catalogue of samples currently held, their location within the freezer, and an estimate of the amount of sample present. Here we report (i) the procedures used to catalogue the eggs contents and tissue samples in the archive; (ii) the progress that has been made to date.

12.2 Cataloguing activities

In some cases, the weight of some of the tissues for storage was recorded but this was not done consistently for all tissues or in all years in the past. The weight of stored egg samples has never been recorded.

Samples in the freezer have been manually inspected and the following information collected and recorded electronically:

- location in freezer
- date the sample was catalogued
- comments on the sample condition (eg. dried sample)
- visual estimate of the amount of sample present (weighing all the samples in the archive was not considered practically feasible).

All the egg samples, a total of 7695, have been catalogued.

A total of 9618 tissue samples have been catalogued so far. These have come from 2329 birds indicating that, on average, 4.1 tissue samples have been collected from each bird. The majority of the birds (1738, equivalent to 75% of the sample catalogued to date) are from sparrowhawks, kestrels, barn owls and herons, the main target species for the tissue analysis conducted by the PBMS. The livers of a total of 5445 birds of these species have been analysed during the lifetime of the PBMS and so the tissues of some 3700 target birds remain to be catalogued. It is estimated that the tissues of approximately 1300 other predatory birds are also contained in the archive and remain to be catalogued. In all, this equates to an estimated 20,000 tissue samples.

To ensure that new samples collected by the PBMS are catalogued before the are archived, data on stored weight and the location in the freezer has been recorded for all new samples submitted to the PBMS from the beginning of 2002. When part of a sample is taken for chemical or other analysis, the weight of the material taken for analysis is now also recorded and the amount of archives sample remaining is estimated.

In conclusion, a large amount of effort has been put in to date to cataloguing the existing archive. To date, some 17,000 samples have been archived, approximately 45% of the estimated total holding. Cataloguing will continue in future years, but at a slower rate because of limited resources and the aim is to complete the process by 2007.

13 References

Ahlborg, U.G., Becking, G.C., Birnbaum, L.S., Brouwer, A., Derks, H.J.G.M., Feeley, M., Golor, G., Hanberg, A., Larsen, J.C., Liem, A.k.D., Safe, S.H., Schlatter, C., Wærn, F., Younes, M. & Yrjänheikki, E. (1994). Toxic equivalency factors for dioxin-like PCBs. *Chemosphere* **28:** 1049-1067.

Albers, P.H. & Loughlin, T.R. (2003). Effects of PAHs on marine birds, mammals and reptiles. In *PAHs - an ecotoxicological perspective.* (ed: Douben, PET), John Wiley & Sons, Chichester, pp. 243-261.

AMAP (1998). *AMAP asssessment report: arctic pollution issues.* Arctic monitoring and assessment programme (AMAP), Oslo, Norway,

Aronson, D., McClure, P., McGinnis, P. & Woodall, G. (2001). Occurrence and comparative toxoocty of PAHs and alkylated PAHs in unused pertroleum products. *Toxicology* **164:** 156.

Ballschmiter, K. & Zell, M. (1980). Analysis of polychlorinated biphenyls (PCB) by glass capillary gas chromatography. *Fesnius Zeitschrift für Analtische Chemie* **302:** 20-31.

Bard, S.M. (1999). Global transport of anthropogenic contaminants and the consequences for the Arctic marine ecosystem. *Marine Pollution Bulletin* **38:** 356-379.

Barrow, E.M., Hulme, M. & Jiang, T. (1993). *A 1961-90 baseline climatology and future climate change scenarios for Great Britain and Europe. Part I: Great Britain baseline climatology. A report accompanying the datasets prepared for the 'Landscape Dynamics and Climate Change' TIGER IV Consortium.* Climatic Research Unit, University of East Anglia, Norwich, UK, pp. 50.

Blus, L.J. (1996). DDT, DDD and DDE in birds. In *Environmental contaminants in wildlife: interpreting tissue concentrations.* (ed: Beyer, WN, Heinz, GH & Redmon-Norwood, AW), CRC Lewis Publishers, Boca Raton, pp. 49-71.

Bosveld, A.T.C., Nieboer, R., de Bont, A., Mennen, J., Murk, A.J., Feyk, L.A., Giesy, J.P. & van den Berg, M. (2000). Biochemical and developmental effects of dietary exposure to polychlorinated biphenyls 126 and 153 in common tern chicks (Sterna hirundo). *Environmental Toxicology and Chemistry* **19:** 719-730.

Broughton, R.K., Osborn, D., Shore, R.F., Wienburg, C.L. & Wadsworth, R.A. (2003). Identifying pollution hotspots from PCB residues in birds of prey. *Environmental Toxicology and Chemistry* **22:** 2519-2524.

Brunstrom, B. (1991). Toxicity and Erod Inducing Potency Of Polychlorinated Biphenyls (PCBs) and Polycyclic Aromatic Hydrocarbons (PAHs) In Avian Embryos. *Comparative Biochemistry and Physiology C Pharmacology Toxicology & Endocrinology* **100:** 241-243.

Brunstrom, B., Broman, D. & Naf, C. (1991). Toxicity and erod inducing potency of 24 polycyclic aromatic hydrocarbons (PAHs) in chick embryos. *Archives of Toxicology* **65:** 485-489.

Brunstrom, B. (1992). Embryolethality and induction of 7 ethoxyresorufin o deethylase in chick embryos by polychlorinated biphenyls and polycyclic aromatic hydrocarbons having Ah receptor affinity. *Chemico Biological Interactions* **81:** 69-77.

Coady, K.K., Jones, P.D. & John, P. (2001). 2,3,7,8-tetrachlorodibenzo-p-dioxin equivalents in tissue samples from three species in the Denver, Colorado, USA, metropolitan area. *Environmental Toxicology and Chemistry* **20:** 2433-2442.

Environment Canada (1994). *Canadian Environmental Protection Act. Priority substances list assessment report: polycyclic aromatic hydrocarbons.* Ministry of Supply & Services, Ottawa, Canada, pp. 61.

Erry, B.V., Macnair, M.R., Meharg, A.A., Shore, R.F. & Newton, I. (1999). Arsenic residues in predatory birds from an area of Britain with naturally and anthropogenically elevated arsenic levels. *Environmental Pollution* **106:** 91-95.

Fuller, R.M., Groom, G.B. & Jones, A.R. (1994). The Land Cover Map of Great Britain: an automated classification of Landsat Thematic Mapper data. *Photogrammetric Engineering and Remote Sensing* **60:** 553-562.

Goldberg, E.D., Bowen, V.T., Farrington, J.W., Harvey, G., Martin, J.H., Parker, P.L., Riseborough, R.W., Robertson, W., Schneider, E. & Gamble, E. (1978). The Mussel Watch. *Environmental Conservation* **5:** 101-126.

Hallett, D.J. & Brecher, R.W. (1984). Cycling of polynuclear aromatic hydrocarbons in the Great Lakes ecosystem. In *Toxic contaminants in the Great Lakes.* (ed: Niagru, JO & Simmons, MS), John Wiley & Sons, New York, pp. 213-238.

Helander, B., Olsson, A., Bignert, A., Asplund, L. & Litzen, K. (2002). The role of DDE, PCB, coplanar PCB and eggshell parameters for reproduction in the white-tailed sea eagle (*Haliaeetus albicilla*) in Sweden. *Ambio* **31:** 386-403.

Hoffman, D.J. (1979). Embryotoxic and teratogenic effects of petroleum hydrocarbons in mallards (*Anas platyrynchos*). *Journal of Toxicology and Environmental Health* **5:** 835-844.

Hoffman, D.J. & Gay, M.L. (1981). Embryotoxic effects of benzo(a)pyrene, chrysene and 7,12 - dimethylbenz(a)anthracene in petroleum hydrocarbon mixtures in mallard ducks. *Journal of Toxicology and Environmental Health* **7:** 775-787.

Hoffman, D.J., Rice, C.P. & Kubiak, T.J. (1996). PCBs and dioxins in birds. In *Environmental contaminants in wildlife: interpreting tissue concentrations.* (ed: Beyer, WN, Heinz, GH & Redmon-Norwood, AW), CRC Lewis Publishers, Boca Raton, pp. 165-207.

Hoffman, D.J., Rattner, B.A., Scheunert, I. & Korte, F. (2001). Environmental contaminants. In *The ecotoxicology of wild mammals.* (ed: Shore, RF & Rattner, BA), John Wiley & Sons, Chichester, pp. 1-48.

Hunter, K. (1985). High-performance liquid chromatographic strategies for the determination and confirmation of anticoagulent rodenticide residues in animal tissues. *Journal of Chromatography* **321:** 255-272.

Johnson, M.S., Leah, R.T., Connor, L., Rae, C. & Saunders, S. (1996). Polychlorinated biphenyls in small mammals from contaminated landfill sites. *Environmental Pollution* **92:** 185-191.

Kannan, K., Kumar, K.S., Nakata, H., Falandysz, J., Oehme, G. & Masunaga, S. (2003). Polychlorinated biphenyls, dihenzo-p-dioxins, dihenzofurans, and p,p '-DDE in livers of white-tailed sea eagles from eastern Germany, 1979-1998. *Environmental Science & Technology* **37:** 1249-1255.

Kruuk, H., Conroy, J.W.H. & Webb, A. (1997). Concentrations of mercury in otters (*Lutra lutra* L) in Scotland in relation to rainfall. *Environmental Pollution* **96:** 13-18.

Lindberg, P., Sellstrom, U., Haggberg, L. & de Wit, C.A. (2004). Higher brominated diphenyl ethers and hexabromocyclododecane found in eggs of peregrine falcons (Falco peregrinus) breeding in Sweden. *Environmental Science & Technology* **38:** 93-96.

Newton, I., Bogan, J., Meek, E. & Little, B. (1982). Organochlorine compounds and shell-thinning in British merlins *Falco columbarius*. *Ibis* **124:** 328-335.

Newton, I. & Haas, M.B. (1988). Pollutants in merlin eggs and their effects on breeding. *British Birds* **81:** 258-269.

Newton, I., Haas, M.B. & Freestone, P. (1990a). Trends in organochlorine and mercury levels in gannet eggs. *Environmental Pollution* **63:** 1-12.

Newton, I., Wyllie, I. & Freestone, P. (1990b). Rodenticides in British Barn Owls. *Environmental Pollution* **68:** 101-117.

Newton, I. & Galbraith, E.A. (1991). Organochlorines and mercury in the eggs of golden eagles *Aquila chrysaetos* from Scotland. *Ibis* **133:** 115-120.

Newton, I., Wyllie, I. & Asher, A. (1992). Mortality from the pesticides aldrin and dieldrin in British sparrowhawks and kestrels. *Ecotoxicology* **1:** 31-44.

Newton, I., Wyllie, I. & Asher, A. (1993). Long-term trends in organochlorine and mercury residues in some predatory birds in Britain. *Environmental Pollution* **79:** 143-151.

Newton, I., Dale, L., Finnie, J.K., Freestone, P., Wright, J., Wyatt, C. & Wyllie, I. (1998). *Wildlife and Pollution: 1997/98. Annual Report. JNCC Report No. 285* Joint Nature Conservation Committee, Peterborough, UK, pp. 61.

Newton, I., Dale, L. & Little, B. (1999a). Trends in organochlorine and mercurial compounds in the eggs of British merlins *Falco columbarius*. *Bird Study* **46:** 356-362.

Newton, I., Shore, R.F., Wyllie, I., Birks, J.D.S. & Dale, L. (1999b). Empirical evidence of side-effects of rodenticides on some predatory birds and mammals. In *Advances in vertebrate pest management.* (ed: Cowan, DP & Feare, CJ), Filander Verlag, Fürth, pp. 347-367.

Newton, I., Afsar, A., Dale, L., Finnie, J.K., Horne, J., Shore, R.F., Wright, J., Wyatt, C. & Wyllie, I. (2000). *Wildlife and Pollution: 1998/99. Annual Report. JNCC Report No. 305* Joint Nature Conservation Committee, Peterborough, UK,

Peakall, D.B. (1996). Dieldrin and other cyclodiene pesticides in wildlife. In *Environmental contaminants in wildlife: interpreting tissue concentrations.* (ed: Beyer, WN, Heinz, GH & Redmon-Norwood, AW), CRC Lewis Publishers, Boca Raton, pp. 73-97.

Senthilkumar, K., Iseki, N., Hayama, S., Nakanishi, J. & Masunaga, S. (2002). Polychlorinated dibenzo-p-dioxins, dibenzofurans, and dioxin- like polychlorinated biphenyls in livers of birds from Japan. *Archives of Environmental Contamination and Toxicology* **42:** 244-255.

Shore, R.F., Wright, J., Horne, J.A. & Sparks, T.H. (1999). Polycyclic aromatic hydrocarbon (PAH) residues in the eggs of coastal-nesting birds from Britain. *Marine Pollution Bulletin* **38:** 509-513.

Shore, R.F., Afsar, A., Horne, J.A. & Wright, J. (2000). *Rodenticide and lead concentrations in red kites Milvus milvus* Centre for Ecology & Hydrology Contract Report to English Nature, pp. 17.

Shore, R.F., Malcolm, H.M., Horne, J.A., Turner, S. & Wienburg, C.L. (2001). *Rodenticide residues in the kestrel Falco tinnunculus* Centre for Ecology & Hydrology Contract Report to English Nature, English Nature, Peterborough, UK, pp. 17.

Shore, R.F., Wienburg, C.L., Dobson, S. & Osborn, D. (2002). *Proposals for potential modifications to the Bird of Prey Monitoring Scheme. JNCC Report No. 320* Joint Nature Conservation Committee, Peterborough, UK, pp. 32.

Shore, R.F., Birks, J.D.S., Afsar, A., Wienburg, C.L. & Kitchener, A.C. (2003). Spatial and temporal analysis of second-generation rodenticide residues in polecats (*Mustela putorius*) from throughout their range in Britain, 1992-1999. *Environmental Pollution* **122:** 183-193.

Shore, R.F., Malcolm, H.M., Wienburg, C.L., Turk, A., Walker, L.A. & Horne, J.A. (2005a). *Wildlife and pollution: 2000/01 Annual report. JNCC Report, No. 351.* Joint Nature Conservation Committee, Peterborough, UK, pp. 49.

Shore, R.F., Malcolm, H.M., Wienburg, C.L., Turk, A., Walker, L.A. & Horne, J.A. (2005b). *Wildlife and pollution: 2001/02 Annual report. JNCC Report, No. 352.* Joint Nature Conservation Committee, Peterborough, UK, pp. 26.

Shore, R.F., Osborn, D., Wienburg, C.L., Sparks, T.H., Broughton, R. & Wadsworth, R. (2005c). *Potential modifications to the Predatory Bird Monitoring Scheme (PBMS): second report. JNCC Report No. 353* Joint Nature Conservation Committee, Peterborough, UK, pp. 58.

Shore, R.F., Malcolm, H.M., McLennan, D., Turk, A., Walker, L.A., Wienburg, C.L. & A., B. (2006). Did Foot and Mouth Disease control operations affect rodenticide exposure in raptors? *Journal of Wildlife Management* **70:**588-593.

Sparks, T.H., Newton, I., Dale, L. & Osborn, D. (1999). Organochlorines in bird of prey eggs: a terrestrial case study. In *Statistics in Ecotoxicology.* (ed: Sparks, TH), John Wiley & Sons, Chichester, pp. 211-245.

Sweetman, A.J., Cousins, I.T., Seth, R., Jones, K.C. & Mackay, D. (2002). A dynamic level IV multimedia environmental model: Application to the fate of polychlorinated biphenyls in the United Kingdom over a 60-year period. *Environmental Toxicology and Chemistry* **21:** 930-940.

Thompson, D.R. (1996). Mercury in birds and terrestrial mammals. In *Environmental contaminants in wildlife: interpreting tissue concentrations.* (ed: Beyer, WN, Heinz, GH & Redmon-Norwood, AW), CRC Lewis Publishers, Boca Raton, pp. 341-356.

Vallack, H.W., Bakker, D.J., Brandt, I., Brostrom-Lunden, E., Brouwer, A., Bull, K.R., Gough, C., Guardans, R., Holoubek, I., Jansson, B., Koch, R., Kuylenstierna, J., Lecloux, A., Mackay, D., McCutcheon, P., Mocarelli, P. & Taalman, R.D.F. (1998). Controlling persistent organic pollutants - what next? *Environmental Toxicology and Pharmacology* **6:** 143-175.

Van den Berg, M., Birnbaum, L., Bosveld, A.T.C., Brunstrom, B., Cook, P., Feeley, M., Giesy, J.P., Hanberg, A., Hasegawa, R., Kennedy, S.W., Kubiak, T., Larsen, J.C., van Leeuwen, F.X.R., Liem, A.K.D., Nolt, C., Peterson, R.E., Poellinger, L., Safe, S., Schrenk, D., Tillitt, D., Tysklind, M., Younes, M., Waern, F. & Zacharewski, T. (1998). Toxic equivalency factors (TEFs) for PCBs, PCDDs, PCDFs for humans and wildlife. *Environmental Health Perspectives* **106:** 775-792.
http://ehpnet771.niehs.nih.gov/docs/1998/1106p1775-1792vandenberg/abstract.html.

Wassenberg, D.M. & Di Giulio, R.T. (2004). Synergistic embryotoxicity of polycyclic aromatic hydrocarbon aryl hydrocarbon receptor Agonists with cytochrome P4501A inhibitors in Fundulus heteroclitus. *Environmental Health Perspectives* **112:** 1658-1664.

WHO (1984). *Environmental Health Criteria 44: Mirex.* WHO, Geneva,

WHO (1994). *Environmental Health Criteria 162: Brominated diphenyl ethers.* WHO, Geneva, pp. 347.

WHO (1998). *Environmental Health Criteria 202: Selected non-heterocyclic polycyclic aromatic hydrocarbons.* WHO, Geneva, pp. 883.

Wiemeyer, S.N. (1996). Other organochlorine pesticides in birds. In *Environmental contaminants in wildlife: interpreting tissue concentrations.* (ed: Beyer, WN, Heinz, GH & Redmon-Norwood, AW), CRC Lewis Publishers, Boca Raton, pp. 99-115.

Wienburg, C.L. & Shore, R.F. (2004). Factors influencing PCB concentrations in the livers of predatory birds in Britain. *Environmental Pollution* **132:** 41-50.

14 Appendix

In the following tables, PCB congeners are identified using the nomenclature of Ballschmiter & Zell (1980) and the Toxic Equivalency Factors (TEFs) are from Ahlborg *et al. (*1994) and Van den Berg *et al. (*1998). Toxic Equivalent (TEQ) concentrations for individual congeners are calculated as the product of the congener concentration and the congener-specific TEF value. Non-detected values for specific congeners are assigned a concentration value of zero when calculating the TEQ; sum TEQ concentrations are the sum of the congener specific concentrations. Lipid wt concentrations for PCB congeners and TEQs can be calculated by multiplying the wet wt concentrations by the conversion factor (CF).

The concentrations of all the PAHs that were determined are given in Tables 14.12 – 14.15. The only data not included in these tables are those for naphthalene (not detected in any eggs), 1-Methylfluoranthene (detected in only one gannet egg (G1083) at a concentration of 3.95 ng/g lipid wt.) and Dibenzothiophene (detected in only one merlin gannet (E8007) at a concentration of 29.5 ng/g lipid wt.)

Table 14.1: Congener specific TEFs and PCB congener concentrations (µg/g wet wt) in the livers of sparrowhawks and herons received in 2002

	TEF	**Liver PCB congener concentration in birds**												
		13668	**13699**	**13701**	**13706**	**13707**	**13712**	**13717**	**13725**	**13737**	**13747**	**13748**	**13751**	**13755**
CF		30.15	34.14	26.83	26.44	37.94	31.25	33.14	32.19	40.09	23.49	38.62	23.53	33.07
8		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
18		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
29		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
28		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
31		ND	0.117	ND	ND	ND	ND	ND	0.033	ND	ND	0.014	ND	ND
52		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
101		ND	0.047	ND	ND	ND	ND	ND	ND	0.043	ND	ND	ND	0.073
81	0.1	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	1.48
77	0.05	ND	0.030	ND	ND	ND	ND	ND	ND	0.053	ND	ND	ND	ND
149		ND	0.061	0.040	ND	ND	ND	ND	ND	0.061	ND	ND	0.042	0.230
123	0.00001	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
118	0.00001	ND	0.128	0.138	0.014	ND	0.036	ND	0.019	0.137	ND	0.017	0.048	0.234
114	0.0001	ND	0.119	0.148	0.013	ND	0.019	ND	0.019	0.090	ND	0.021	0.100	0.312
153		0.037	0.658	0.874	0.066	ND	0.131	ND	0.119	0.490	0.064	0.121	0.903	2.65
141		ND	0.028	ND	ND	ND	ND	ND	ND	ND	0.055	ND	ND	0.089
105	0.0001	ND	0.020	0.030	ND	ND	ND	ND	ND	ND	ND	ND	0.049	0.044
163		0.033	0.189	0.183	0.017	ND	0.030	ND	0.025	0.136	ND	0.031	0.210	0.156
138		ND	0.276	0.293	0.029	ND	0.060	ND	0.057	0.222	0.030	0.049	0.355	0.607
187		0.029	0.819	0.633	0.035	ND	0.075	ND	0.071	0.447	0.055	0.123	0.700	3.21
183		ND	0.113	0.124	ND	ND	ND	ND	0.016	0.070	ND	0.018	0.116	0.304
126	0.1	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
128		ND	0.070	0.074	ND	ND	0.019	ND	ND	0.052	ND	ND	0.056	0.224
167	0.00001	ND	0.023	0.049	ND	ND	ND	ND	ND	ND	ND	ND	ND	0.059
171		ND	0.031	0.035	ND	ND	ND	ND	ND	0.034	ND	ND	0.049	0.086
156	0.0001	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
199		ND	0.120	0.146	ND	ND	0.018	ND	0.016	0.070	ND	0.017	0.097	0.299
157	0.0001	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	0.033
180		0.022	0.659	0.832	0.059	ND	0.081	ND	0.085	0.377	0.046	0.093	0.694	2.65
201		ND	ND	0.220	ND	ND	0.019	ND	0.027	0.120	ND	0.031	0.172	0.622
170		ND	ND	0.239	0.017	ND	0.025	ND	0.027	0.132	ND	0.030	0.254	0.732
169	0.001	ND	0.173	0.149	ND	ND	ND	ND	0.016	0.071	ND	0.019	0.104	0.373
189	0.00001	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	0.057
194		ND	0.131	ND	0.024	ND	0.024	ND	ND	0.066	0.026	0.022	0.036	0.268
205		ND	ND	ND	ND	ND	ND	ND	0.023	ND	ND	ND	ND	0.017
206		ND	0.079	ND	ND	ND	ND	ND	ND	0.025	0.070	0.010	ND	0.102
209		ND	ND	ND	ND	ND	ND	ND	ND	0.026	ND	ND	ND	0.039
Sum		0.121	3.89	4.21	0.274	ND	0.538	ND	0.553	2.72	0.346	0.615	3.987	15.0

ND is not detected. CF is conversion factor needed to convert from wet wt to lipid wt concentrations

Table 14.1 cont:

	TEF	**Liver PCB congener concentration in birds**												
		13759	**13765**	**13788**	**13792**	**13808**	**13812**	**13816**	**13822**	**13826**	**13839**	**13844**	**13845**	**13849**
CF		42.39	25.88	28.21	21.09	48.92	34.29	26.99	26.81	27.31	42.27	26.91	48.38	35.10
8		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
18		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
29		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
28		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
31		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
52		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
101		0.071	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
81	0.1	1.57	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
77	0.05	0.014	ND	ND	ND	ND	ND	ND	ND	ND	0.018	ND	ND	ND
149		0.074	ND	ND	ND	ND	ND	ND	ND	ND	0.047	ND	ND	ND
123	0.00001	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
118	0.00001	0.214	ND	0.093	ND	0.034	0.029	0.032	0.020	ND	0.120	ND	ND	ND
114	0.0001	0.173	ND	0.079	ND	0.028	ND	0.018	0.013	ND	0.149	ND	ND	ND
153		1.09	ND	0.529	0.075	0.143	0.091	0.138	0.123	0.065	0.850	ND	0.058	0.022
141		0.024	0.034	ND	ND	ND	ND	ND	ND	ND	ND	0.029	ND	ND
105	0.0001	0.032	ND	ND	ND	ND	ND	0.017	ND	ND	0.025	ND	ND	ND
163		0.243	ND	0.112	0.017	0.045	0.029	0.035	0.027	ND	0.196	ND	ND	ND
138		0.409	0.025	0.198	0.027	0.055	0.045	0.057	0.041	0.024	0.336	ND	0.017	ND
187		1.03	ND	0.249	0.041	0.133	0.079	0.146	0.072	0.060	0.689	ND	0.047	ND
183		0.166	ND	0.065	ND	0.020	ND	0.032	0.020	ND	0.171	ND	ND	ND
126	0.1	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
128		0.118	ND	0.059	ND	ND	ND	0.020	ND	ND	0.062	ND	ND	ND
167	0.00001	0.036	ND	0.026	ND	ND	ND	0.016	ND	ND	0.023	ND	ND	ND
171		0.046	ND	ND	ND	ND	ND	0.020	ND	ND	0.044	ND	ND	ND
156	0.0001	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
199		0.168	ND	0.079	ND	0.030	ND	0.028	0.026	ND	0.145	ND	ND	ND
157	0.0001	0.018	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
180		0.937	ND	0.401	0.061	0.144	0.079	0.191	0.134	0.060	0.854	ND	0.063	ND
201		0.367	ND	0.088	ND	0.034	0.028	0.067	0.041	ND	0.325	ND	0.017	ND
170		0.262	ND	0.127	0.019	0.049	ND	0.054	0.045	0.019	0.257	ND	ND	ND
169	0.001	ND	ND	0.074	ND	0.026	ND	0.053	0.024	ND	ND	ND	ND	ND
189	0.00001	ND	ND	ND	ND	ND	ND	ND	ND	ND	0.022	ND	ND	ND
194		0.161	ND	0.074	0.022	0.038	ND	ND	0.031	0.036	0.182	ND	0.025	ND
205		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
206		0.106	ND	0.030	ND	0.013	ND	ND	0.010	ND	0.112	ND	ND	ND
209		0.042	ND	ND	ND	ND	ND	ND	ND	ND	0.055	ND	ND	ND
Sum		7.37	0.059	2.28	0.261	0.792	0.380	0.922	0.627	0.264	4.68	0.029	0.226	0.022

ND is not detected. CF is conversion factor needed to convert from wet wt to lipid wt concentrations

Table 14.1 cont:

	TEF	Liver PCB congener concentration in birds												
		13850	**13858**	**13861**	**13867**	**13879**	**13894**	**13903**	**13905**	**13919**	**13925**	**13947**	**13977**	**13993**
CF		42.70	34.04	32.95	36.73	30.97	30.23	33.18	42.95	30.80	38.85	33.86	29.76	26.16
8		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
18		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
29		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
28		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
31		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
52		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
101		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
81	0.1	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
77	0.05	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
149		ND	ND	ND	ND	ND	ND	ND	ND	ND	0.017	ND	ND	ND
123	0.00001	ND	ND	ND	ND	ND	ND	ND	ND	ND	0.030	ND	ND	ND
118	0.00001	0.017	ND	ND	0.057	0.011	0.017	ND	0.030	ND	0.095	ND	ND	0.016
114	0.0001	0.018	ND	ND	0.053	0.014	ND	ND	0.024	ND	0.073	ND	ND	ND
153		0.100	0.071	0.037	0.364	0.074	0.056	ND	0.158	ND	0.443	ND	0.069	0.081
141		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
105	0.0001	ND	0.025	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
163		0.027	0.020	ND	0.058	0.020	ND	ND	0.036	ND	0.113	ND	0.022	0.020
138		0.037	0.032	0.013	0.124	0.029	0.019	ND	0.052	ND	0.169	ND	0.024	0.031
187		0.072	0.049	0.033	0.176	0.061	0.032	ND	0.149	ND	0.423	ND	ND	0.094
183		ND	ND	ND	0.042	0.012	ND	ND	0.020	ND	0.080	ND	ND	ND
126	0.1	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
128		ND	ND	ND	0.035	ND	ND	ND	ND	ND	0.051	ND	ND	ND
167	0.00001	ND	ND	ND	0.019	ND	ND	ND	ND	ND	0.029	ND	ND	ND
171		ND	ND	ND	ND	ND	ND	ND	ND	ND	0.021	ND	ND	ND
156	0.0001	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
199		ND	ND	ND	0.050	ND	ND	ND	0.024	ND	0.102	ND	ND	ND
157	0.0001	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
180		0.071	0.056	0.033	0.251	0.067	0.037	ND	ND	ND	0.525	ND	0.069	0.080
201		0.021	ND	ND	0.084	0.019	ND	ND	0.055	ND	0.200	ND	0.018	0.037
170		0.019	0.018	ND	0.083	0.018	ND	ND	0.035	ND	0.148	ND	0.021	0.021
169	0.001	ND	ND	ND	0.070	0.015	ND	ND	ND	ND	0.143	ND	ND	0.025
189	0.00001	ND	ND	ND	ND	ND	ND	ND	ND	ND	0.019	ND	ND	ND
194		0.025	ND	0.020	0.059	0.021	0.017	ND	ND	ND	0.119	ND	0.023	0.027
205		ND	ND	ND	ND	ND	ND	ND	ND	ND	0.293	ND	ND	ND
206		ND	ND	ND	0.026	0.008	ND	ND	ND	ND	0.074	ND	ND	ND
209		ND	ND	ND	ND	ND	ND	ND	ND	ND	0.066	ND	ND	ND
Sum		0.405	0.272	0.137	1.55	0.370	0.178	ND	0.584	ND	3.23	ND	0.246	0.431

ND is not detected. CF is conversion factor needed to convert from wet wt to lipid wt concentrations

Table 14.1 cont:

	TEF	Liver PCB congener concentration in birds	
		13670	**13909**
CF		16.60	31.41
8		ND	ND
18		ND	ND
29		ND	ND
28		ND	ND
31		ND	ND
52		ND	ND
101		ND	ND
81	0.1	ND	ND
77	0.05	ND	ND
149		ND	ND
123	0.00001	ND	ND
118	0.00001	0.015	ND
114	0.0001	ND	ND
153		0.035	0.015
141		ND	ND
105	0.0001	ND	ND
163		0.013	ND
138		0.020	ND
187		0.023	0.019
183		ND	ND
126	0.1	ND	ND
128		ND	ND
167	0.00001	ND	ND
171		ND	ND
156	0.0001	ND	ND
199		ND	ND
157	0.0001	ND	ND
180		0.019	ND
201		ND	ND
170		ND	ND
169	0.001	ND	ND
189	0.00001	ND	ND
194		ND	0.014
205		ND	ND
206		ND	ND
209		ND	ND
Sum		0.125	0.048

ND is not detected. CF is conversion factor needed to convert from wet wt to lipid wt concentrations

Table 14.2: Congener specific and sum TEQ concentrations (pg/g wet wt) in the livers of sparrowhawks and herons received in 2002

	TEQ concentration in livers of birds												
	13668	**13699**	**13701**	**13706**	**13707**	**13712**	**13717**	**13725**	**13737**	**13747**	**13748**	**13751**	**13755**
CF	30.15	34.14	26.83	26.44	37.94	31.25	33.14	32.19	40.09	23.49	38.62	23.53	33.07
77	ND	1501	ND	ND	ND	ND	ND	ND	2625	ND	ND	ND	ND
81	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	148252
105	ND	1.97	2.97	ND	ND	ND	ND	ND	ND	ND	ND	4.91	4.44
114	ND	11.9	14.8	1.30	ND	1.94	ND	1.85	9.00	ND	2.10	10.0	31.2
118	ND	1.28	1.38	0.137	ND	0.360	ND	0.192	1.37	ND	0.167	0.480	2.34
123	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
126	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
156	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
157	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	3.27
167	ND	0.225	0.491	ND	ND	ND	ND	ND	ND	ND	ND	ND	0.590
169	ND	173	149	ND	ND	ND	ND	16.4	71.5	ND	19.5	104	373
189	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	0.573
Sum	ND	1689	169	1.43	ND	2.30	ND	18.4	2707	ND	21.7	119	148668

ND is not detected. CF is conversion factor needed to convert from wet wt to lipid wt concentrations

Table 14.2: continued

	TEQ concentration in livers of birds												
	13759	**13765**	**13788**	**13792**	**13808**	**13812**	**13816**	**13822**	**13826**	**13839**	**13844**	**13845**	**13849**
CF	42.39	25.88	28.21	21.09	48.92	34.29	26.99	26.81	27.31	42.27	26.91	48.38	35.10
77	714	ND	ND	ND	ND	ND	ND	ND	ND	882	ND	ND	ND
81	157331	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
105	3.24	ND	ND	ND	ND	ND	1.70	ND	ND	2.53	ND	ND	ND
114	17.3	ND	7.88	ND	2.78	ND	1.83	1.29	ND	14.9	ND	ND	ND
118	2.14	ND	0.934	ND	0.337	0.292	0.325	0.197	ND	1.20	ND	ND	ND
123	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
126	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
156	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
157	1.83	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
167	0.355	ND	0.258	ND	ND	ND	0.157	ND	ND	0.230	ND	ND	ND
169	ND	ND	73.7	ND	26.4	ND	52.9	24.4	ND	ND	ND	ND	ND
189	ND	ND	ND	ND	ND	ND	ND	ND	ND	0.215	ND	ND	ND
Sum	158070	ND	82.7	ND	29.5	0.292	56.9	25.9	ND	901	ND	ND	ND

ND is not detected. CF is conversion factor needed to convert from wet wt to lipid wt concentrations

Table 14.2: continued

	TEQ concentration in livers of birds												
	13850	**13858**	**13861**	**13867**	**13879**	**13894**	**13903**	**13905**	**13919**	**13925**	**13947**	**13977**	**13993**
CF	42.70	34.04	32.95	36.73	30.97	30.23	33.18	42.95	30.80	38.85	33.86	29.76	26.16
77	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
81	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
105	ND	2.50	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
114	1.84	ND	ND	5.33	1.39	ND	ND	2.39	ND	7.26	ND	ND	ND
118	0.166	ND	ND	0.569	0.115	0.169	ND	0.298	ND	0.945	ND	ND	0.156
123	ND	ND	ND	ND	ND	ND	ND	ND	ND	0.297	ND	ND	ND
126	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
156	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
157	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
167	ND	ND	ND	0.185	ND	ND	ND	ND	ND	0.285	ND	ND	ND
169	ND	ND	ND	70.3	15.1	ND	ND	ND	ND	143	ND	ND	24.7
189	ND	ND	ND	ND	ND	ND	ND	ND	ND	0.188	ND	ND	ND
Sum	2.00	2.50	ND	76.4	16.6	0.169	ND	2.69	ND	152	ND	ND	24.9

ND is not detected. CF is conversion factor needed to convert from wet wt to lipid wt concentrations

Table 14.2: continued

	TEQ concentration in livers of birds	
	13850	**13858**
CF	42.70	34.04
77	ND	ND
81	ND	ND
105	ND	2.50
114	1.84	ND
118	0.166	ND
123	ND	ND
126	ND	ND
156	ND	ND
157	ND	ND
167	ND	ND
169	ND	ND
189	ND	ND
Sum	2.00	2.50

ND is not detected. CF is conversion factor needed to convert from wet wt to lipid wt concentrations

Table 14.3: Congener specific TEFs and PCB congener concentrations (µg/g wet wt) for merlin eggs received in 2002

	TEF	PCB congener concentration in merlin eggs						
		E7999	E8007	E8008	E8106	E8110	E8115	E8119
CF		*16.39*	*18.35*	*18.10*	*15.85*	*22.78*	*20.99*	*19.10*
8		ND	ND	0.016	ND	0.023	ND	ND
18		ND	ND	ND	ND	ND	ND	ND
29		ND	ND	ND	ND	ND	ND	ND
28		ND	ND	ND	ND	ND	ND	ND
31		ND	ND	ND	ND	ND	ND	ND
52		ND	ND	ND	ND	0.037	ND	ND
101		ND	ND	ND	ND	ND	ND	ND
81	0.1	ND	ND	ND	ND	ND	ND	ND
77	0.05	ND	ND	ND	ND	ND	ND	ND
149		ND	ND	ND	ND	ND	0.006	ND
123	0.00001	ND	ND	ND	ND	ND	ND	ND
118	0.00001	ND	0.048	0.038	0.047	0.054	0.070	0.069
114	0.0001	ND	0.027	0.022	0.032	0.036	0.045	0.041
153		ND	0.177	0.163	0.215	0.216	0.298	0.311
141		ND	ND	ND	ND	ND	ND	ND
105	0.0001	ND	0.016	0.015	0.013	0.014	0.023	0.019
163		ND	0.026	0.034	0.044	0.047	0.036	0.048
138		ND	0.063	0.059	0.056	0.078	0.082	0.085
187		ND	0.055	0.069	0.081	0.087	0.078	0.092
183		ND	0.018	0.014	0.019	0.020	0.028	0.026
126	0.1	ND	ND	ND	ND	ND	0.003	0.004
128		ND	0.011	0.007	0.008	0.015	0.018	0.012
167	0.00001	ND	0.012	0.011	0.016	0.010	0.025	0.016
171		ND	0.005	0.005	0.008	0.005	0.009	0.007
156	0.0001	ND	ND	ND	ND	ND	ND	ND
199		ND	0.029	0.022	0.032	0.027	0.059	0.043
157	0.0001	ND	0.005	ND	ND	ND	0.008	0.004
180		ND	0.097	0.097	0.130	0.114	0.166	0.168
201		ND	0.030	0.020	0.036	0.031	0.049	0.039
170		ND	0.041	0.035	0.047	0.040	0.080	0.057
169	0.001	ND	0.030	0.022	0.045	0.029	0.062	0.045
189	0.00001	ND	0.010	ND	0.008	0.005	ND	0.008
194		ND	0.032	0.024	0.043	0.026	0.066	0.043
205		ND	ND	ND	0.005	ND	ND	ND
206		ND	0.015	0.009	0.023	0.014	0.033	0.021
209		ND	0.021	ND	0.024	0.011	0.022	0.018
Sum		ND	0.772	0.681	0.931	0.939	1.267	1.176

ND is not detected. CF is conversion factor needed to convert from wet wt to lipid wt concentrations

Table 14.4: Congener specific and sum TEQ concentrations (pg/g wet wt) for merlin eggs received in 2002

	TEQ concentration in merlin eggs						
	E7999	**E8007**	**E8008**	**E8106**	**E8110**	**E8115**	**E8119**
CF	*16.39*	*18.35*	*18.10*	*15.85*	*22.78*	*20.99*	*19.10*
77	ND	ND	ND	ND	ND	ND	ND
81	ND	ND	ND	ND	ND	ND	ND
105	ND	1.61	1.47	1.29	1.39	2.31	1.88
114	ND	2.73	2.23	3.16	3.58	4.55	4.06
118	ND	0.484	0.376	0.473	0.536	0.704	0.693
123	ND	ND	ND	ND	ND	ND	ND
126	ND	ND	ND	ND	ND	342.16	419.96
156	ND	ND	ND	ND	ND	ND	ND
157	ND	0.518	ND	ND	ND	0.765	0.407
167	ND	0.120	0.112	0.158	0.103	0.247	0.157
169	ND	30.3	22.1	44.9	29.5	61.9	45.2
189	ND	0.103	ND	0.078	0.048	ND	0.075
Sum	ND	35.8	26.2	50.0	35.1	413	472

ND is not detected. CF is conversion factor needed to convert from wet wt to lipid wt concentrations

Table 14.5: Congener specific TEFs and PCB congener concentrations (µg/g wet wt) for golden eagle eggs received in 2002

	TEF	PCB congener concentration in golden eagle eggs												
		E8005	**E8010**	**E8011**	**E8055**	**E8056**	**E8057**	**E7990**	**E7991**	**E7993**	**E7995**	**E7996**	**E7998**	**E8000**
CF		25.52	16.54	4.22	31.16	20.93	21.36	23.48	23.43	15.89	27.11	17.81	21.31	19.97
8		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
18		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
29		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
28		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
31		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
52		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
101		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
81	0.1	ND	0.397	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
77	0.05	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
149		ND	ND	ND	0.017	0.026	ND	ND	ND	ND	ND	ND	ND	ND
123	0.00001	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
118	0.00001	ND	ND	ND	0.282	0.021	ND	0.016	ND	ND	0.012	ND	ND	ND
114	0.0001	ND	ND	ND	0.027	ND	ND	ND	ND	ND	ND	ND	ND	ND
153		ND	0.050	0.160	1.693	0.132	ND	0.035	0.037	0.043	0.073	0.015	0.028	0.038
141		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
105	0.0001	ND	0.020	ND	0.059	0.015	ND	ND	0.017	ND	0.050	ND	ND	ND
163		ND	ND	ND	0.021	ND	ND	ND	ND	ND	ND	ND	ND	ND
138		ND	0.017	ND	0.405	0.040	0.019	ND	ND	ND	0.023	ND	ND	ND
187		ND	ND	ND	0.047	0.020	ND	ND	ND	ND	ND	ND	ND	ND
183		ND	ND	ND	0.179	0.012	ND	ND	ND	ND	ND	ND	ND	ND
126	0.1	ND	ND	ND	0.014	ND	ND	ND	ND	ND	ND	ND	ND	ND
128		ND	ND	ND	0.065	ND	ND	ND	ND	ND	ND	ND	ND	ND
167	0.00001	ND	ND	ND	0.081	ND	ND	ND	ND	ND	ND	ND	ND	ND
171		ND	ND	ND	0.032	ND	ND	ND	ND	ND	ND	ND	ND	ND
156	0.0001	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
199		ND	ND	ND	0.124	ND	ND	ND	ND	ND	ND	ND	ND	ND
157	0.0001	ND	ND	ND	0.020	ND	ND	ND	ND	ND	ND	ND	ND	ND
180		ND	0.042	0.128	1.648	0.081	0.044	0.047	0.037	0.032	0.060	ND	0.034	0.040
201		ND	ND	ND	0.019	ND	ND	ND	ND	ND	ND	ND	ND	ND
170		ND	ND	ND	0.379	0.025	0.017	ND	ND	ND	0.018	ND	ND	ND
169	0.001	ND	ND	ND	0.093	ND	ND	ND	ND	ND	ND	ND	ND	ND
189	0.00001	ND	ND	ND	0.036	ND	ND	ND	ND	ND	ND	ND	ND	ND
194		ND	0.023	ND	0.329	0.023	0.027	0.026	0.022	0.031	0.026	ND	0.019	0.033
205		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
206		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
209		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
Sum		ND	0.547	0.288	5.568	0.395	0.108	0.124	0.113	0.106	0.262	0.015	0.080	0.111

ND is not detected. CF is conversion factor needed to convert from wet wt to lipid wt concentrations

Table 14.6: Congener specific and sum TEQ concentrations (pg/g wet wt) for golden eagle eggs received in 2002

	TEQ concentration in golden eagle eggs												
	E8005	**E8010**	**E8011**	**E8055**	**E8056**	**E8057**	**E7990**	**E7991**	**E7993**	**E7995**	**E7996**	**E7998**	**E8000**
CF	25.52	16.54	4.22	31.16	20.93	21.36	23.48	23.43	15.89	27.11	17.81	21.31	19.97
77	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
81	ND	39650	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
105	ND	1.96	ND	5.88	1.47	ND	ND	1.70	ND	5.04	ND	ND	ND
114	ND	ND	ND	2.71	ND	ND	ND	ND	ND	ND	ND	ND	ND
118	ND	ND	ND	2.82	0.213	ND	0.165	ND	ND	0.118	ND	ND	ND
123	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
126	ND	ND	ND	1425	ND	ND	ND	ND	ND	ND	ND	ND	ND
156	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
157	ND	ND	ND	2.04	ND	ND	ND	ND	ND	ND	ND	ND	ND
167	ND	ND	ND	0.813	ND	ND	ND	ND	ND	ND	ND	ND	ND
169	ND	ND	ND	93.1	ND	ND	ND	ND	ND	ND	ND	ND	ND
189	ND	ND	ND	0.356	ND	ND	ND	ND	ND	ND	ND	ND	ND
Sum	ND	39652	ND	1533	1.68	ND	ND	1.70	ND	5.16	ND	ND	ND

ND is not detected. CF is conversion factor needed to convert from wet wt to lipid wt concentrations

Table 14.7: Congener specific TEFs and PCB congener concentrations (µg/g wet wt) for gannet eggs from Ailsa Craig received in 2002

	TEF	PCB congener concentration in gannet eggs from Ailsa Craig									
		G1073	G1074	G1075	G1076	G1077	G1078	G1079	G1080	G1081	G1082
CF		28.51	30.30	25.70	23.00	26.56	23.53	25.39	20.87	27.11	29.58
8		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
18		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
29		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
28		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
31		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
52		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
101		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
81	0.1	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
77	0.05	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
149		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
123	0.00001	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
118	0.00001	ND	0.023	0.025	0.035	0.038	0.024	0.042	0.044	0.030	0.033
114	0.0001	ND	ND	ND	0.020	0.021	0.012	0.024	0.019	ND	ND
153		0.058	0.071	0.093	0.149	0.158	0.104	0.203	0.143	0.051	0.121
141		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
105	0.0001	0.033	ND	ND	0.021	0.020	0.016	0.018	0.021	ND	0.041
163		ND	ND	ND	0.021	0.020	0.011	0.025	0.061	ND	ND
138		0.029	0.035	0.050	0.080	0.086	0.054	0.098	ND	0.031	0.063
187		ND	0.021	0.029	0.056	0.064	0.035	0.074	0.051	ND	0.047
183		ND	ND	ND	ND	ND	0.013	0.022	ND	ND	ND
126	0.1	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
128		ND	ND	ND	0.021	ND	ND	0.021	ND	ND	ND
167	0.00001	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
171		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
156	0.0001	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
199		ND	ND	ND	ND	ND	0.011	0.017	ND	ND	ND
157	0.0001	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
180		0.054	0.056	0.067	0.089	0.113	0.078	0.128	0.087	0.034	0.083
201		ND	ND	ND	ND	ND	ND	0.016	ND	ND	ND
170		ND	0.020	0.022	0.031	0.041	0.027	0.048	0.030	ND	0.029
169	0.001	ND	ND	ND	ND	ND	0.014	0.017	ND	ND	ND
189	0.00001	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
194		0.025	0.025	0.029	0.031	0.031	0.021	ND	0.029	0.023	0.027
205		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
206		ND	ND	ND	ND	ND	0.009	ND	ND	ND	ND
209		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
Sum		0.199	0.251	0.315	0.554	0.591	0.430	0.754	0.485	0.169	0.444

ND is not detected. CF is conversion factor needed to convert from wet wt to lipid wt concentrations

Table 14.8: Congener specific TEFs and PCB congener concentrations (µg/g wet wt) for gannet eggs from Bass Rock received in 2002

	TEF	PCB congener concentration in gannet eggs from Bass Rock									
		G1083	**G1084**	**G1085**	**G1086**	**G1087**	**G1088**	**G1089**	**G1090**	**G1091**	**G1092**
CF		20.81	25.67	21.77	29.74	22.89	19.64	38.09	24.05	19.87	26.06
8		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
18		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
29		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
28		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
31		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
52		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
101		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
81	0.1	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
77	0.05	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
149		ND	ND	0.038	0.023	0.056	ND	ND	ND	0.036	0.023
123	0.00001	ND	ND	ND	ND	ND	ND	ND	ND	0.020	ND
118	0.00001	0.104	0.032	0.085	0.069	0.095	0.077	0.049	0.035	0.139	0.080
114	0.0001	0.046	0.016	0.056	0.031	0.059	0.045	0.025	0.016	0.074	0.042
153		0.316	0.133	0.400	0.238	0.439	0.342	0.222	0.159	0.542	0.309
141		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
105	0.0001	0.037	ND	0.033	0.027	0.044	0.034	0.021	0.019	0.043	0.033
163		0.048	0.030	0.060	0.033	0.068	0.046	0.022	ND	0.079	0.040
138		0.170	0.070	0.199	0.134	0.223	0.176	0.105	0.078	0.277	0.173
187		0.091	0.042	0.144	0.083	0.161	0.107	0.066	0.046	0.172	0.106
183		0.021	ND	0.041	0.029	0.046	0.032	ND	ND	0.045	0.038
126	0.1	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
128		0.044	ND	0.045	0.033	0.050	0.039	ND	ND	0.060	0.033
167	0.00001	ND	ND	ND	ND	ND	ND	ND	ND	0.020	0.013
171		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
156	0.0001	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
199		0.022	ND	0.032	ND	0.034	0.025	0.020	ND	0.041	0.025
157	0.0001	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
180		0.117	0.065	0.209	0.118	0.227	0.169	0.160	0.095	0.239	0.165
201		ND	ND	0.027	ND	0.030	0.019	ND	ND	0.031	0.021
170		0.046	0.026	0.081	0.046	0.089	0.065	0.056	0.037	0.095	0.061
169	0.001	ND	ND	ND	ND	0.026	0.021	0.021	ND	0.033	0.023
189	0.00001	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
194		ND	0.022	0.038	0.027	0.039	0.032	0.036	0.027	0.062	ND
205		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
206		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
209		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
Sum		1.06	0.437	1.49	0.892	1.69	1.23	0.805	0.513	2.01	1.19

ND is not detected. CF is conversion factor needed to convert from wet wt to lipid wt concentrations

Table 14.9: Congener specific and sum TEQ concentrations (pg/g wet wt) for the gannet eggs from Ailsa Craig received in 2002

	TEQ concentration in gannet eggs from Ailsa Craig									
	G1073	**G1074**	**G1075**	**G1076**	**G1077**	**G1078**	**G1079**	**G1080**	**G1081**	**G1082**
CF	28.51	30.30	25.70	23.00	26.56	23.53	25.39	20.87	27.11	29.58
77	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
81	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
105	3.27	ND	ND	2.09	1.95	1.60	1.79	2.12	ND	4.05
114	ND	ND	ND	1.97	2.09	1.15	2.37	1.90	ND	ND
118	ND	0.228	0.249	0.349	0.378	0.238	0.423	0.443	0.304	0.329
123	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
126	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
156	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
157	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
167	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
169	ND	ND	ND	ND	ND	13.9	16.8	ND	ND	ND
189	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
Sum	3.27	0.228	0.249	4.40	4.42	16.9	21.4	4.46	0.304	4.38

ND is not detected. CF is conversion factor needed to convert from wet wt to lipid wt concentrations

Table 14.10: Congener specific and sum TEQ concentrations (pg/g wet wt) for the gannet eggs from Bass Rock received in 2002

	TEQ concentration in gannet eggs from Bass Rock									
	G1083	**G1084**	**G1085**	**G1086**	**G1087**	**G1088**	**G1089**	**G1090**	**G1091**	**G1092**
CF	20.81	25.67	21.77	29.74	22.89	19.64	38.09	24.05	19.87	26.06
77	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
81	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
105	3.66	ND	3.29	2.72	4.41	3.42	2.10	1.86	4.28	3.30
114	4.58	1.63	5.55	3.14	5.94	4.50	2.54	1.62	7.44	4.24
118	1.04	0.315	0.851	0.693	0.947	0.765	0.495	0.349	1.39	0.795
123	ND	ND	ND	ND	ND	ND	ND	ND	0.201	ND
126	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
156	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
157	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
167	ND	ND	ND	ND	ND	ND	ND	ND	0.199	0.133
169	ND	ND	ND	ND	26.1	21.0	20.9	ND	32.6	23.1
189	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
Sum	9.28	1.94	9.69	6.55	37.4	29.7	26.0	3.83	46.1	31.6

ND is not detected. CF is conversion factor needed to convert from wet wt to lipid wt concentrations

Table 14.11: Congener specific and sum PCB and TEQ wet wt concentrations for the sea eagle egg received in 2002

	TEF	PCB concentration (µg/g wet wt)	TEQ concentration (pg/g wet wt)
CF		22.03	22.03
8		ND	
18		ND	
29		ND	
28		ND	
31		ND	
52		ND	
101		ND	
81	0.1	ND	ND
77	0.05	ND	ND
149		0.058	
123	0.00001	ND	ND
118	0.00001	0.413	4.13
114	0.0001	0.072	7.16
153		2.38	
141		ND	
105	0.0001	0.114	11.4
163		0.078	
138		0.807	
187		0.158	
183		0.264	
126	0.1	ND	ND
128		0.138	
167	0.00001	0.095	0.950
171		0.051	
156	0.0001	ND	ND
199		0.170	
157	0.0001	0.027	2.72
180		2.13	
201		0.044	
170		0.566	
169	0.001	0.232	232
189	0.00001	0.080	0.799
194		0.300	
205		ND	
206		0.099	
209		0.058	
Congener Sum		8.34	260

Table 14.12: Concentrations (ng/g lipid wt) of PAHs in gannet eggs from Ailsa Craig

	Concentrations of individual PAHs in eggs									
	G1073	**G1074**	**G1075**	**G1076**	**G1077**	**G1078**	**G1079**	**G1080**	**G1081**	**G1082**
2-Methylnaphthalene	ND	32.0	ND	ND	49.7	ND	40.3	ND	47.7	75.7
1-Methylnaphthalene	ND	21.4	ND	17.2	37.4	ND	27.8	ND	33.1	55.2
2-Ethylnaphthalene	ND	ND	ND	ND	3.90	ND	4.87	ND	5.33	6.18
1-Ethylnaphthalene	ND	ND	ND	ND	ND	ND	4.45	ND	ND	ND
2,6 & 2,7-Dimethylnaphthalene	5.18	11.4	11.2	12.3	27.3	10.9	35.2	5.51	38.5	44.0
1,3 & 1,7-Dimethylnaphthalene	5.06	12.5	9.49	10.3	28.7	9.22	31.4	4.54	34.1	46.4
1,6-Dimethylnaphthalene	5.44	9.50	8.14	9.21	26.9	7.74	30.4	4.27	33.2	44.9
2,3 & 1,4-Dimethylnaphthalene	ND	7.13	5.70	6.36	22.6	6.11	22.5	2.87	23.0	34.9
1,5-Dimethylnaphthalene	ND	ND	ND	ND	7.92	ND	8.22	ND	8.90	13.4
Acenaphthylene	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
1,2-Dimethylnaphthalene	ND	5.77	5.17	5.98	20.5	5.06	18.9	2.83	19.9	30.2
1,8-Dimethylnaphthalene	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
Acenaphthene	ND	ND	ND	4.81	17.5	ND	14.6	ND	15.6	24.2
2,3,5-Trimethylnaphthalene	ND	ND	5.07	ND	ND	ND	ND	ND	ND	52.6
Fluorene	8.81	9.13	ND	8.83	31.0	ND	ND	ND	ND	ND
Phenanthrene	27.7	23.7	ND	20.0	105	26.3	179	ND	101	90.9
Anthracene	2.54	1.07	1.31	1.22	4.18	1.89	9.57	ND	4.44	4.53
2-Methylphenanthrene	6.13	3.74	4.05	4.53	38.3	5.33	77.1	ND	41.8	40.5
1-Methylphenanthrene	3.15	ND	ND	2.70	20.9	3.13	52.3	ND	23.8	22.3
3,6-Dimethylphenanthrene	ND	0.869	ND	ND	ND	ND	17.8	ND	ND	ND
Fluoranthene	10.2	4.29	6.23	5.34	13.4	16.0	14.5	ND	15.9	11.8
9,10-Dimethylphenanthrene	ND	ND	0.357	ND	ND	ND	0.999	ND	ND	0.718
Pyrene	5.20	ND	6.87	4.19	6.27	19.2	8.20	ND	8.83	6.10
2-Methylfluoranthene	ND	ND	1.67	ND	ND	ND	3.35	1.04	2.27	ND
Benzo[a]fluorene	ND	ND	ND	ND	1.01	1.97	3.61	ND	1.45	ND
Benzo[b]fluorene	ND	ND	ND	ND	ND	ND	0.781	ND	0.857	ND
1-Methylpyrene	ND	ND	0.346	ND	ND	ND	ND	ND	0.408	ND
Benzo[ghi]fluoranthene	0.792	0.316	1.47	0.436	0.719	4.06	0.555	0.245	0.680	0.600
Benzo[c]phenanthrene	0.347	0.093	0.722	0.166	0.280	1.37	0.263	ND	0.359	0.315
Cyclopenta[cd]pyrene	0.290	0.178	0.309	0.203	0.262	2.03	0.271	0.165	0.357	0.188
Benz[a]anthracene	ND	ND	2.74	0.586	0.799	5.70	1.11	ND	ND	0.941
Triphenylene & Chrysene	3.04	0.842	3.01	1.46	1.98	8.29	4.96	ND	2.05	2.31
3-Methylchrysene	ND	ND	ND	ND	ND	1.44	3.87	ND	ND	ND
2-Methylchrysene	ND	ND	0.163	ND	0.179	0.592	1.17	ND	ND	ND
5-Methylchrysene	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
4 & 6-Methylchrysene	ND	ND	ND	ND	ND	ND	1.23	ND	ND	ND
1-Methylchrysene	ND	ND	ND	ND	0.291	ND	ND	ND	ND	ND
Benzo[b & j & k]fluoranthene	1.41	ND	2.77	ND	1.71	7.64	1.06	ND	ND	ND
Benzo[e]pyrene	0.746	ND	ND	ND	0.913	4.25	0.658	ND	ND	0.445
Benzo[a]pyrene	ND	ND	ND	ND	0.671	10.5	ND	ND	ND	1.28
Perylene	0.486	ND	ND	ND	0.406	ND	0.355	ND	ND	ND
Ideno[1,2,3-cd]pyrene	ND	ND	ND	ND	ND	8.84	ND	ND	ND	ND
Dibenz[ah]anthracene	0.111	0.209	0.118	ND	0.187	1.44	ND	ND	ND	0.214
Benzo[ghi]perylene	1.43	1.38	1.10	0.388	3.13	ND	ND	ND	ND	0.872
Anthanthrene	ND	0.193	0.132	ND	0.173	ND	ND	ND	ND	ND
Dibenzo[a,i]pyrene	ND	ND	ND	ND	ND	0.778	ND	0.416	ND	ND
Coronene	0.330	ND	ND	ND	0.689	2.11	ND	ND	ND	ND
Dibenzo[a,e]pyrene	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
Dibenzo[a,h]pyrene	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND

ND: not detected

Table 14.13: Concentrations (ng/g lipid wt) of PAHs in gannet eggs from Bass Rock

	Concentrations of individual PAHs in eggs									
	G1083	**G1084**	**G1085**	**G1086**	**G1087**	**G1088**	**G1089**	**G1090**	**G1091**	**G1092**
2-Methylnaphthalene	ND	ND	ND	121	ND	ND	ND	ND	ND	ND
1-Methylnaphthalene	31.7	ND	ND	76.0	ND	ND	ND	ND	21.3	ND
2-Ethylnaphthalene	17.0	ND	ND	5.61	ND	ND	ND	ND	ND	3.17
1-Ethylnaphthalene	12.9	ND	ND	5.17	ND	ND	ND	ND	ND	ND
2,6 & 2,7-Dimethylnaphthalene	25.4	14.2	11.4	26.9	10.4	2.50	8.72	6.99	15.3	19.5
1,3 & 1,7-Dimethylnaphthalene	38.6	12.3	9.66	23.9	8.94	ND	7.27	6.81	12.7	17.0
1,6-Dimethylnaphthalene	32.1	9.40	8.30	15.8	7.25	2.22	6.48	5.76	10.5	13.9
2,3 & 1,4-Dimethylnaphthalene	56.3	7.46	5.85	12.0	5.29	1.82	4.61	4.78	8.83	10.5
1,5-Dimethylnaphthalene	23.3	ND	ND	4.26	ND	ND	ND	ND	ND	3.55
Acenaphthylene	9.36	ND	ND	4.74	ND	ND	ND	ND	ND	ND
1,2-Dimethylnaphthalene	39.5	6.35	5.34	11.0	4.96	1.74	4.02	4.82	7.04	9.38
1,8-Dimethylnaphthalene	10.7	ND	ND	ND	ND	ND	ND	ND	ND	ND
Acenaphthene	9.35	ND	ND	ND	ND	ND	3.94	ND	6.08	19.7
2,3,5-Trimethylnaphthalene	15.1	ND	ND	ND	5.32	ND	ND	ND	ND	16.9
Fluorene	9.26	ND	7.39	9.13	8.11	ND	ND	7.95	ND	20.2
Phenanthrene	ND	ND	ND	ND	ND	ND	ND	ND	ND	36.5
Anthracene	5.37	2.10	ND	ND	1.05	1.39	1.06	ND	1.65	1.32
2-Methylphenanthrene	5.13	ND	ND	ND	ND	4.37	ND	ND	4.55	5.90
1-Methylphenanthrene	4.51	ND	ND	ND	ND	5.32	ND	ND	2.76	2.83
3,6-Dimethylphenanthrene	6.30	ND	ND	ND	0.904	ND	ND	ND	ND	ND
Fluoranthene	5.73	8.01	4.03	ND	7.02	8.27	5.14	13.6	10.3	11.3
9,10-Dimethylphenanthrene	7.80	0.649	ND	ND	ND	ND	0.351	ND	1.26	0.901
Pyrene	5.18	6.68	ND	ND	ND	ND	3.42	7.87	9.33	ND
2-Methylfluoranthene	3.70	4.28	ND	ND	ND	ND	ND	ND	ND	ND
Benzo[a]fluorene	4.65	ND	ND	ND	ND	3.23	ND	ND	2.54	0.963
Benzo[b]fluorene	5.07	ND	ND	ND	ND	ND	ND	1.90	ND	ND
1-Methylpyrene	6.83	ND	ND	ND	ND	ND	ND	0.580	0.657	0.292
Benzo[ghi]fluoranthene	3.76	1.15	0.482	0.163	0.754	0.631	0.526	2.69	1.62	0.689
Benzo[c]phenanthrene	17.4	0.663	0.211	0.107	0.294	0.359	0.204	1.01	0.638	0.349
Cyclopenta[cd]pyrene	6.55	0.864	0.118	0.076	0.740	0.504	0.346	0.228	0.792	0.333
Benz[a]anthracene	5.72	1.70	0.640	ND	ND	ND	ND	18.9	2.94	ND
Triphenylene & Chrysene	11.4	3.43	ND	ND	2.88	ND	ND	ND	5.24	2.44
3-Methylchrysene	4.54	ND	ND	ND	ND	ND	ND	ND	ND	ND
2-Methylchrysene	5.47	ND	ND	ND	ND	0.729	ND	ND	ND	ND
5-Methylchrysene	4.64	ND	ND	ND	ND	ND	ND	ND	ND	ND
4 & 6-Methylchrysene	10.5	ND	ND	ND	ND	ND	ND	0.237	ND	ND
1-Methylchrysene	5.53	ND	ND	ND	ND	ND	ND	0.492	ND	ND
Benzo[b & j & k]fluoranthene	22.9	ND	ND	ND	0.733	ND	ND	11.6	2.53	ND
Benzo[e]pyrene	6.73	ND	ND	ND	ND	ND	ND	3.36	1.08	ND
Benzo[a]pyrene	5.46	2.03	1.35	ND	ND	1.01	ND	ND	0.979	ND
Perylene	5.07	0.404	ND	ND	ND	ND	ND	0.477	ND	ND
Ideno[1,2,3-cd]pyrene	4.00	ND	ND	ND	ND	ND	ND	ND	3.80	ND
Dibenz[ah]anthracene	4.30	ND	ND	ND	0.128	0.177	0.097	0.591	1.78	ND
Benzo[ghi]perylene	4.62	2.46	0.476	ND	0.635	0.975	0.416	ND	3.42	0.469
Anthanthrene	4.54	ND	ND	ND	ND	ND	ND	ND	ND	ND
Dibenzo[a,i]pyrene	13.7	ND	ND	ND	ND	ND	ND	ND	ND	ND
Coronene	4.45	ND	ND	ND	ND	ND	ND	ND	ND	ND
Dibenzo[a,e]pyrene	5.05	ND	ND	ND	ND	ND	ND	ND	ND	ND
Dibenzo[a,h]pyrene	4.12	ND	ND	ND	ND	ND	0.217	ND	ND	ND

ND: not detected

Table 14.14: Concentrations (ng/g lipid wt) of PAHs in golden eagle and sea eagle (E7988 only) eggs

	Concentrations of individual PAHs in eggs						
	E7990	**E7991**	**E7993**	**E7995**	**E7996**	**E7998**	**E8000**
2-Methylnaphthalene	ND	ND	69.6	ND	45.9	ND	60.3
1-Methylnaphthalene	ND	ND	45.0	ND	31.3	ND	39.1
2-Ethylnaphthalene	ND	ND	3.63	ND	2.99	ND	2.89
1-Ethylnaphthalene	ND	ND	ND	ND	4.06	ND	ND
2,6 & 2,7-Dimethylnaphthalene	6.35	8.48	17.6	10.2	15.2	5.73	14.7
1,3 & 1,7-Dimethylnaphthalene	5.15	7.09	14.4	8.25	13.7	5.20	12.8
1,6-Dimethylnaphthalene	4.53	6.02	10.9	7.36	ND	3.56	9.15
2,3 & 1,4-Dimethylnaphthalene	ND	4.33	ND	5.42	9.34	ND	7.39
1,5-Dimethylnaphthalene	ND	ND	2.48	ND	2.88	ND	ND
Acenaphthylene	ND	ND	ND	ND	ND	ND	ND
1,2-Dimethylnaphthalene	3.03	4.06	6.98	4.82	7.46	2.75	6.00
1,8-Dimethylnaphthalene	ND	ND	0.181	ND	ND	ND	ND
Acenaphthene	ND	ND	3.78	ND	3.72	ND	3.83
2,3,5-Trimethylnaphthalene	ND	4.17	ND	ND	ND	ND	ND
Fluorene	ND	ND	6.64	8.46	ND	ND	5.75
Phenanthrene	ND	ND	ND	ND	ND	ND	ND
Anthracene	1.01	1.06	ND	1.38	0.903	ND	ND
2-Methylphenanthrene	ND	ND	ND	ND	ND	ND	ND
1-Methylphenanthrene	ND	ND	ND	ND	ND	ND	ND
3,6-Dimethylphenanthrene	ND	0.776	ND	0.833	ND	ND	ND
Fluoranthene	6.07	18.6	ND	10.9	6.16	5.70	ND
9,10-Dimethylphenanthrene	ND	ND	ND	ND	0.275	ND	ND
Pyrene	ND	21.0	ND	5.04	ND	9.56	ND
2-Methylfluoranthene	1.53	3.08	0.445	1.42	0.797	0.583	ND
Benzo[a]fluorene	1.20	4.98	ND	0.992	ND	ND	ND
Benzo[b]fluorene	0.817	ND	ND	ND	ND	ND	ND
1-Methylpyrene	ND	2.27	ND	ND	ND	0.336	ND
Benzo[ghi]fluoranthene	0.941	22.4	0.199	0.700	0.349	1.51	0.120
Benzo[c]phenanthrene	0.235	5.40	0.071	0.254	0.157	0.477	0.114
Cyclopenta[cd]pyrene	0.446	5.14	0.097	0.484	0.119	1.01	ND
Benz[a]anthracene	1.14	29.8	0.254	0.681	0.417	1.91	0.308
Triphenylene & Chrysene	1.94	49.1	0.605	1.89	1.21	3.13	0.648
3-Methylchrysene	0.566	ND	ND	ND	ND	ND	ND
2-Methylchrysene	0.265	8.69	ND	ND	ND	ND	ND
5-Methylchrysene	ND	ND	ND	ND	ND	ND	ND
4 & 6-Methylchrysene	ND	4.54	ND	ND	0.069	0.099	ND
1-Methylchrysene	ND	ND	ND	ND	ND	0.119	ND
Benzo[b & j & k]fluoranthene	1.11	58.2	ND	ND	ND	3.63	ND
Benzo[e]pyrene	ND	28.4	ND	ND	ND	2.34	ND
Benzo[a]pyrene	ND	77.9	ND	ND	ND	3.46	ND
Perylene	ND	ND	ND	ND	ND	ND	ND
Ideno[1,2,3-cd]pyrene	2.03	58.2	ND	ND	ND	5.58	ND
Dibenz[ah]anthracene	0.140	9.74	ND	ND	ND	0.810	ND
Benzo[ghi]perylene	ND	92.6	0.338	0.777	ND	11.0	ND
Anthanthrene	ND	8.53	ND	ND	ND	ND	ND
Dibenzo[a,i]pyrene	ND	ND	ND	1.41	ND	ND	ND
Coronene	ND	3.77	ND	ND	ND	2.55	ND
Dibenzo[a,e]pyrene	ND	ND	ND	ND	ND	0.418	ND
Dibenzo[a,h]pyrene	ND	ND	ND	ND	ND	ND	ND

ND: not detected

Table 14.14 cont.. Concentrations (ng/g lipid wt) of PAHs in golden eagle and sea eagle (E7988 only) eggs

	Concentrations of individual PAHs in eggs						
	E8005	**E8010**	**E8011**	**E8055**	**E8056**	**E8057**	**E7988**
2-Methylnaphthalene	ND	73.7	21.8	120	ND	ND	ND
1-Methylnaphthalene	ND	46.0	13.6	ND	ND	ND	ND
2-Ethylnaphthalene	ND	4.04	1.53	7.43	ND	ND	ND
1-Ethylnaphthalene	ND	ND	ND	ND	ND	ND	ND
2,6 & 2,7-Dimethylnaphthalene	ND	21.3	10.1	45.5	9.93	10.5	9.22
1,3 & 1,7-Dimethylnaphthalene	8.30	17.8	8.26	33.5	8.26	9.11	6.23
1,6-Dimethylnaphthalene	12.8	12.0	7.10	23.4	ND	6.66	ND
2,3 & 1,4-Dimethylnaphthalene	4.79	ND	5.01	20.2	5.17	5.71	4.91
1,5-Dimethylnaphthalene	ND	2.60	ND	ND	ND	ND	ND
Acenaphthylene	ND	ND	1.14	ND	ND	ND	ND
1,2-Dimethylnaphthalene	ND	8.53	4.09	16.4	4.44	5.23	ND
1,8-Dimethylnaphthalene	ND	ND	0.043	0.456	ND	ND	ND
Acenaphthene	ND	4.35	3.75	9.47	5.69	7.21	ND
2,3,5-Trimethylnaphthalene	ND	ND	ND	ND	5.19	ND	4.27
Fluorene	ND	8.43	ND	16.8	6.99	10.2	7.40
Phenanthrene	ND	12.6	10.4	24.0	ND	25.4	ND
Anthracene	ND	0.700	0.472	3.27	1.13	1.55	1.23
2-Methylphenanthrene	ND	3.97	1.22	ND	ND	5.53	ND
1-Methylphenanthrene	ND	3.17	0.590	ND	ND	3.28	ND
3,6-Dimethylphenanthrene	ND	ND	ND	ND	0.746	ND	ND
Fluoranthene	8.39	4.58	1.15	13.3	5.29	9.64	5.00
9,10-Dimethylphenanthrene	ND	ND	ND	ND	0.325	0.324	3.04
Pyrene	5.07	3.50	0.781	23.0	ND	4.87	ND
2-Methylfluoranthene	0.799	ND	ND	ND	1.40	1.79	ND
Benzo[a]fluorene	ND	1.00	ND	ND	ND	1.12	ND
Benzo[b]fluorene	ND	ND	ND	ND	ND	ND	2.04
1-Methylpyrene	ND	0.223	ND	1.01	ND	ND	ND
Benzo[ghi]fluoranthene	0.932	0.707	0.067	2.74	0.56	1.10	0.222
Benzo[c]phenanthrene	0.372	0.282	ND	2.59	0.165	0.241	ND
Cyclopenta[cd]pyrene	0.456	0.355	0.034	ND	0.288	0.895	0.241
Benz[a]anthracene	1.38	1.03	0.059	ND	ND	0.757	ND
Triphenylene & Chrysene	3.25	2.13	ND	ND	1.34	1.58	ND
3-Methylchrysene	ND	ND	ND	ND	ND	0.516	ND
2-Methylchrysene	ND	ND	ND	ND	0.121	0.252	ND
5-Methylchrysene	ND	ND	ND	ND	ND	ND	ND
4 & 6-Methylchrysene	ND	ND	ND	ND	ND	ND	ND
1-Methylchrysene	ND	ND	ND	ND	ND	ND	ND
Benzo[b & j & k]fluoranthene	ND	0.431	ND	1.96	0.613	0.579	ND
Benzo[e]pyrene	ND	ND	ND	1.01	0.310	0.401	0.450
Benzo[a]pyrene	ND	ND	ND	2.44	ND	ND	ND
Perylene	ND	ND	ND	ND	ND	ND	ND
Ideno[1,2,3-cd]pyrene	ND	ND	ND	2.04	ND	ND	ND
Dibenz[ah]anthracene	ND	ND	ND	0.127	0.270	0.152	ND
Benzo[ghi]perylene	0.611	0.514	0.068	2.97	1.32	1.82	ND
Anthanthrene	ND	ND	ND	0.303	0.157	0.381	ND
Dibenzo[a,i]pyrene	ND	ND	0.052	ND	ND	ND	ND
Coronene	0.441	ND	ND	ND	0.581	0.484	ND
Dibenzo[a,e]pyrene	ND	ND	ND	ND	ND	ND	ND

Dibenzo[a,h]pyrene	ND	ND	ND	ND	ND	0.161	ND

ND: not detected

Table 14.15: Concentrations (ng/g lipid wt) of PAHs in merlin eggs

	Concentrations of individual PAHs in eggs						
	E7999	**E8007**	**E8008**	**E8106**	**E8110**	**E8115**	**E8119**
2-Methylnaphthalene	28.2	104	ND	84.7	ND	38.2	50.7
1-Methylnaphthalene	19.8	65.0	ND	53.1	ND	26.7	35.0
2-Ethylnaphthalene	ND	11.5	ND	4.23	ND	ND	3.97
1-Ethylnaphthalene	ND	5.38	ND	ND	ND	ND	3.39
2,6 & 2,7-Dimethylnaphthalene	9.12	70.9	3.99	21.2	ND	ND	18.9
1,3 & 1,7-Dimethylnaphthalene	7.58	56.7	3.23	18.3	ND	ND	17.0
1,6-Dimethylnaphthalene	6.32	57.1	2.94	12.4	ND	ND	ND
2,3 & 1,4-Dimethylnaphthalene	4.88	48.5	ND	9.61	ND	ND	ND
1,5-Dimethylnaphthalene	ND	ND	ND	2.82	ND	ND	3.35
Acenaphthylene	ND	8.32	ND	3.47	ND	ND	5.15
1,2-Dimethylnaphthalene	4.29	ND	2.05	8.58	ND	ND	9.50
1,8-Dimethylnaphthalene	ND	ND	ND	ND	ND	ND	ND
Acenaphthene	ND	216	ND	6.88	ND	ND	5.45
2,3,5-Trimethylnaphthalene	ND	74.5	ND	ND	ND	ND	ND
Fluorene	ND	314	ND	7.80	ND	7.34	11.5
Phenanthrene	ND	983	27.7	ND	ND	15.2	20.2
Anthracene	ND	117	2.83	0.750	ND	2.14	2.20
2-Methylphenanthrene	ND	75.6	4.17	ND	ND	ND	4.49
1-Methylphenanthrene	ND	ND	2.57	ND	ND	ND	ND
3,6-Dimethylphenanthrene	ND	ND	ND	ND	ND	ND	ND
Fluoranthene	ND	ND	5.81	2.02	ND	7.07	7.96
9,10-Dimethylphenanthrene	ND	ND	1.42	ND	ND	ND	0.312
Pyrene	ND	ND	3.41	ND	ND	4.88	ND
2-Methylfluoranthene	ND	ND	ND	ND	ND	ND	ND
Benzo[a]fluorene	ND	ND	ND	ND	ND	ND	ND
Benzo[b]fluorene	ND	ND	ND	ND	ND	0.907	ND
1-Methylpyrene	ND	0.323	ND	ND	ND	0.388	0.262
Benzo[ghi]fluoranthene	ND	ND	0.274	0.301	ND	0.743	ND
Benzo[c]phenanthrene	ND	ND	ND	0.122	ND	0.306	ND
Cyclopenta[cd]pyrene	ND	0.291	0.188	0.210	ND	0.265	0.175
Benz[a]anthracene	ND	0.499	ND	0.266	ND	1.05	0.619
Triphenylene & Chrysene	ND	1.83	0.935	0.487	ND	2.54	1.42
3-Methylchrysene	ND	ND	ND	ND	ND	ND	ND
2-Methylchrysene	ND	ND	ND	ND	ND	ND	ND
5-Methylchrysene	ND	ND	ND	ND	ND	0.007	ND
4 & 6-Methylchrysene	ND	ND	ND	ND	ND	ND	ND
1-Methylchrysene	ND	ND	ND	ND	ND	ND	ND
Benzo[b & j & k]fluoranthene	ND	ND	ND	ND	ND	ND	ND
Benzo[e]pyrene	ND	ND	ND	ND	ND	ND	ND
Benzo[a]pyrene	ND	0.606	ND	ND	ND	ND	ND
Perylene	ND	ND	ND	ND	ND	ND	ND
Ideno[1,2,3-cd]pyrene	ND	ND	ND	ND	ND	ND	ND
Dibenz[ah]anthracene	ND	ND	ND	ND	ND	ND	ND
Benzo[ghi]perylene	ND	0.551	ND	0.220	ND	0.485	0.307
Anthanthrene	ND	ND	ND	ND	ND	ND	ND
Dibenzo[a,i]pyrene	ND	ND	ND	ND	ND	ND	ND
Coronene	ND	ND	ND	ND	ND	ND	ND
Dibenzo[a,e]pyrene	ND	ND	ND	ND	ND	ND	ND

Dibenzo[a,h]pyrene	ND	ND	ND	ND	ND	ND	ND

ND: not detected