Cetaceans Strandings Investigation and Co-ordination in the UK

Report to Defra for the period 1 January 2000 – 31 December 2004

defra
Department for Environment
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**TRENDS IN CETACEAN STRANDINGS AROUND
THE UK COASTLINE AND CETACEAN AND
MARINE TURTLE POST-MORTEM INVESTIGATIONS,
2000 to 2004 INCLUSIVE (CONTRACT CRO 238)**

Edited by

Paul D. Jepson

Contributing Authors

R. C. Sabin, C. J. H. Spurrier, P. D. J., Chimonides (NHM)
P. D. Jepson, R. Deaville, M. Perkins and A.A. Cunningham (ZSL)
R. J. Reid, I.A.P. Patterson, G. Foster and J. Barley (SAC)
R. Penrose (MEM)
R.J. Law (CEFAS)

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Executive summary

This report outlines the research conducted on UK cetacean strandings between 2000 and 2004 under contract to the Department for Environment, Food and Rural Affairs (Defra) (Contract no: CR0238). The main aims and objectives of the research are to monitor trends in UK cetacean strandings and to conduct standardised post-mortem investigations on stranded and by-caught cetacean and marine turtle carcasses in order to monitor the incidence of disease; determine causes of mortality; investigate potential relationships between health status and pollutant exposure; maintain a database of pooled data derived from these investigations; and maintain a national tissue archive for current and future scientific research purposes. The research contributes to the UK Government's commitment to a number of national and international conservation agreements including the EU Habitats Directive, ASCOBANS (Agreement on the Conservation of Small Cetaceans of the Baltic and North Seas) and UK Biodiversity Action Plans for cetaceans and marine turtles. The research also informs Ministers and the public on issues relating to cetacean mortality.

Trends in Cetacean Strandings 2000-2004

There were no unusual cetacean or marine turtle mass-mortality events identified in UK waters between 2000 and 2004. Data compiled by The Natural History Museum show that 3197 cetaceans of 18 species were reported stranded between 1st January 2000 and 31st December 2004. Consistent with previous years, the most common UK-stranded cetacean species in 2000-2004 was the harbour porpoise (*Phocoena phocoena*) of which 1596 were recorded. The last five years (2000-2004) have seen progressively increasing annual numbers of UK-stranded cetaceans reported, reaching a peak in 2004 when 799 individuals were recorded. The annual increase in UK cetacean strandings was predominantly due to increasing numbers of short-beaked common dolphins (*Delphinus delphis*) and harbour porpoises recorded in south-west England between January and April and a lesser increase in reported harbour porpoise strandings in Wales. Analyses of age and body-length data suggest that harbour porpoises stranding in south-west England (presumably from the Celtic Sea stock) have significantly greater body lengths compared to harbour porpoises from other UK regions, suggesting there are at least two separate stocks in UK waters.

Investigations of Disease and Mortality

The most common causes of mortality of 563 UK-stranded harbour porpoises examined at *post mortem* between 2000-2004 were attack from bottlenose dolphins (*Tursiops truncatus*) (n=128), pneumonias due to combinations of parasitic, bacterial and fungal infections (n=102), entanglement in fishing gear (bycatch) (n=93) and starvation (n=74). All cases of fatal attack from bottlenose dolphins occurred in north-east Scotland, west Wales or south-west England, where porpoises share coastal habitats with resident or transient bottlenose dolphin populations. In Scotland, the distribution of both bottlenose dolphin live-sightings and stranded harbour porpoises fatally attacked by bottlenose dolphins has occurred with increasing frequency outside the Moray Firth area, suggesting that the Moray Firth bottlenose dolphin population has expanded outside of the designated Moray Firth bottlenose dolphin

Special Area of Conservation (SAC) under the EU Habitats Directive. The number and proportion of stranded harbour porpoises killed by bottlenose dolphins has increased annually in west Wales between 1999 and 2004. The primary cause(s) of these violent interactions, or why they are becoming more prevalent in West Wales, are not known.

As in the 1990s, bycatch was the most common cause of death in UK-stranded common dolphins accounting for 116 of the 190 (61.1%) examined between 2000 and 2004. The majority of both common dolphin and harbour porpoise bycatches typically stranded in south-west England (Cornwall and Devon) between December and April. The increasing number of harbour porpoise and common dolphin strandings (predominantly bycatches) recorded in south-west England in recent years could reflect a genuine increase in (predominantly bycatch-related) mortality of these two species, although closer analysis of the strandings data suggests that other factors such as changes in abundance and distribution of these species and particularly increased coastal vigilance for stranded carcasses in Cornwall and Devon probably play a considerable role. Studies on harbour porpoises also identified a range of fatal (mainly parasitic and bacterial) infectious diseases that occurred in an increasing proportion in winter strandings. Infectious diseases were much less frequent causes of death in other species.

A novel and potentially fatal pathological entity characterized by acute and chronic systemic gas and fat embolism was first described in UK-stranded cetaceans during this project. These new findings, characterized by acute and chronic cavitary lesions in a range of tissues including liver, kidneys, lymph nodes and spleen, were identified in five common dolphins, four Risso's dolphins (*Grampus griseus*), two harbour porpoises, a Blainville's beaked whale (*Mesoplodon densirostris*) and a Sowerby's beaked whale (*Mesoplodon bidens*) that stranded in the UK between 1992 and 2004. They provide some of the first pathological evidence that cetaceans can suffer *in vivo* gas bubble formation, gas and fat embolism and associated tissue injury and that in some circumstances these bubbles can persist through time. Theoretical mechanisms proposed for bubble formation involve behaviourally altered dive profiles (possibly in response to loud underwater noise) resulting in excessive tissue nitrogen supersaturation (similar to decompression sickness in human divers), or direct physical effects of sound on bubble nucleation and growth in tissues. The history of the UK-stranded gas embolism cases, including whether they were exposed to any kind of acoustic activity, is not known.

Trends in pollutant levels and studies of potential health effects

Data on tissue pollutant levels (organochlorine pesticides and polychlorinated biphenyls, trace elements, butyltins and polybrominated diphenylether flame retardants) and post-mortem findings derived from over 300 UK-stranded harbour porpoises stranded between 1989 and 2002 were analysed during the period of this report. Scottish porpoises had significantly lower levels of PCBs, several organochlorine pesticides, chromium and nickel than porpoises from England and Wales. Levels of some metals and organochlorine pesticides had gradually declined since 1990 in porpoises from all parts of the UK. In contrast, levels of polychlorinated biphenyls (PCBs) were generally higher, more stable over time and many porpoises (particularly in England and Wales) had blubber PCB levels considered toxic in other

aquatic mammals. Levels of butyltins (tributyltin, dibutyltin and monobutyltin) were relatively low and consistent with being phased out as active ingredients in marine antifouling paints. Concentrations of newer pollutants in the marine environment (specifically brominated flame retardants) were detected for the first time in UK-stranded cetacean tissues and have fed into the EU risk assessments of flame retardant compounds and contributed to a ban on the production and use of the penta-mix and octa-mix polybrominated diphenyl ether flame retardant formulations. Monitoring pollutant levels in key species such as harbour porpoises also enables specific reduction targets for these compounds to be set (e.g. specific targets for PCBs and other contaminants in the Species Action Plan for the harbour porpoise under the UK Cetacean and Turtle Biodiversity Action Plan). The data are important to other inter-governmental fora such as OSPAR, as it contributes to the national perspective on the significance of strategies for the control of chemicals.

The pathological and toxicological data collected on UK-stranded harbour porpoises since 1989 enables some robust investigations on the adverse health effects of exposure to marine pollutants. A case-control epidemiological study demonstrated that harbour porpoises dying of infectious disease (n=82) had significantly higher levels of PCBs than healthy porpoises that died of traumatic deaths such as bycatch (n=175). This statistical association occurred independently of a range of potentially confounding variables including age and two quantitative indices of nutritional status, suggesting that PCB exposure may impair immune function in harbour porpoises causing increased susceptibility to potentially fatal viral, parasitic, bacterial and fungal infections. The population-level effects of PCB-related toxicity in harbour porpoises are not known. Porpoises that died of infectious disease also had significantly elevated levels of some metals including mercury (Hg), selenium (Se) and zinc (Zn) in their livers (compared to those that died of physical trauma) but these associations may in part be linked to physiological redistribution caused by loss of nutritional condition in diseased animals. The correlation found between Hg and Se levels is also consistent with the ability of Se to accumulate in marine mammals to bind and detoxify methylmercury. No associations were found between infectious disease and exposure to butyltins (tributyltin, dibutyltin and monobutyltin) or polybrominated flame retardants, although sample sizes were smaller.

Conclusions and recommendations

Bycatch has been the major cause of death in UK-stranded harbour porpoises and common dolphins since systematic post-mortem examinations were first conducted in 1990, and the UK Government has already prioritised research to assess the population-level impacts of bycatch and to improve its mitigation. The potential effect on cetacean populations of changes in prey availability due to commercial fishing activity is another potential area of conflict that is very difficult to study but may be monitored to some degree by continuing longitudinal cetacean stranding studies to assess trends over time in composition of cetacean diet, nutritional condition, life history parameters (such as age of sexual maturity), and prevalence of starvation as a cause of mortality.

The ability of some cetaceans (especially deep-diving species) to suffer potentially fatal gas embolism has been demonstrated, but there is much uncertainty in the mechanisms that produce these intravascular bubbles and in the factors that

influence their initiation and persistence. Retrospective and prospective pathological studies of cetacean tissues from the UK and other countries are recommended to investigate the prevalence of gas and fat emboli in tissues, which may have been historically under-reported. In terms of pollutants and their potential effects on cetacean health, future research activity should continue to monitor levels of contaminants (such as PCBs) that still have been linked to adverse health effects in harbour porpoises and for which future targets have been set for their reduction. Risk assessment-type studies could be conducted in the future to predict the population-level impacts of PCBs on harbour porpoise (and other cetacean) populations of known size and PCB exposure. Further research is also needed to investigate the complex interactions between Hg, Se, health status and nutritional status before the direction of causality can be conclusively ascribed to these relationships. Finally, continued monitoring of the newer brominated flame retardants currently emerging and bioaccumulating in marine food chains are warranted.

1. 0 BACKGROUND

1.1 Project Background

In 1990 the UK Department of the Environment (later DETR, now Defra) initiated a long-term monitoring programme involving the systematic post-mortem examination of UK-stranded marine mammals. The research conducted under this programme continues to provide insight into the diseases, health status and threats to cetaceans and turtles in UK waters. Such a systematic and long-term monitoring programme of stranded animals facilitates the investigation of spatio-temporal trends in disease, causes of mortality and exposure to environmental pollutants largely inaccessible by other methods. It also enables both ongoing assessment of the dynamics of particular threats and their response to specific conservation measures. A range of publications in peer-reviewed scientific journals has been generated directly or indirectly from this research (see **Annex D**).

December 2004 marked the end of the third 5-year period during which research into cetacean strandings has been supported financially by contracts awarded by Defra and its predecessors. Post-mortem investigations of UK marine mammal strandings (funded by Defra, formerly DETR and DoE) have been co-ordinated in England and Wales by the Institute of Zoology (IoZ) at the Zoological Society of London since 1990 and in Scotland by the Scottish Agricultural College in Inverness (SAC) since 1992. More recently, marine turtle examinations have been included within the remit of both organisations. During the period of this report, pathological investigations were conducted under subcontract to the Natural History Museum by IoZ (subcontract ref: ZRT C S047 516 COa) and SAC (subcontract ref: ZRT C S047 516 COb).

1.2 Aims and Policy Background

There are a number of drivers for the research ranging from meeting international legislative requirements, informing UK policy and meeting national and regional reporting requirements for a range of customers.

1. Habitats Directive
All cetacean species are listed on Annex IV (Animal and Plant Species of Community Interest in Need of Strict Protection) of the EC Habitats Directive. Under Annex IV the keeping, sale or exchange of such species is banned, as well as deliberate capture, killing or disturbance. The Directive also requires Member States to monitor the incidental capture and killing of all cetaceans. The bottlenose dolphin and the harbour porpoise are also listed on Annex II of the EC Habitats Directive. Under Annex II marine SACs (Special Areas of Conservation) for bottlenose dolphins have been designated in the Moray Firth, (north-east Scotland) and in Cardigan Bay (west Wales).

2. ASCOBANS

An Agreement on the Conservation of Small Cetaceans in the Baltic and North Seas (ASCOBANS), formulated in 1992, has been signed by ten European countries, including the UK. Under the Agreement, provision is made for protection of specific areas, monitoring, research, information exchange, pollution control and heightening public awareness. These measures were until recently aimed specially at protecting dolphins and porpoises in the North and Baltic Seas but the UK Government has applied the spirit of the Agreement to all UK waters and following the extension of the Agreement area will apply it to formally to all UK waters. This Agreement requires signatory bodies to monitor the causes of death of stranded cetaceans and retrieve and document cases of bycatch. ASCOBANS conservation and management plan states that: *'each party shall endeavour to establish efficient system for reporting and retrieving bycatches and stranding specimens and to carry out ….., full autopsies in order to collect tissues for further studies and reveal possible causes of death and to document food composition. The information shall be made available in an international database'*.

3. UK BAP
The <u>Convention of Biological Diversity</u>, signed in June 1992 by 159 governments at the Earth Summit (Rio Summit), was the first treaty to provide a legal framework for biodiversity conservation and called for the creation and enforcement of national strategies and action plans to conserve, protect and enhance biological diversity. The UK Government response to the Convention was set out in <u>Biodiversity: the UK Action Plan</u>, launched in 1994, including the preparation of costed action plans with quantifiable targets for our most threatened species and wildlife habitats (<u>www.ukbap.org.uk</u>).

Cetaceans and turtles are covered by the following plans:
- Group plan for baleen whales
- Group plan for small dolphins
- Group plan for beaked whales
- Group plan for marine turtles
- Species Action Plan for Harbour porpoise

Each of these plans has detailed actions including research and monitoring requirements which are considered necessary for the conservation of the species.

4. UK Small Cetacean Bycatch Response Strategy
Bycatch, the incidental capture of cetaceans during fishing activities, is thought to be the major threat to the conservation of small cetaceans, not only in Europe, but also throughout the world's oceans. This strategy, published by Defra in 2003, provides a summary of the abundance and distribution of small cetaceans in UK waters, and outlines our international obligations to conserve these species. The strategy reviews fishery interactions with small cetaceans and recommends measures to alleviate bycatch, sets out areas for further research and provides indicative costs for the recommendations.

The <u>principal requirements of the contract</u> are to monitor trends in UK cetacean strandings and to conduct standardised post-mortem investigations on stranded and by-caught cetacean carcasses in order to monitor the incidence of disease; determine causes of mortality; investigate potential relationships between health

status and pollutant exposure; maintain a database of pooled data derived from these investigations; and maintain a national tissue archive for current and future scientific research purposes. The post-mortem investigations of marine turtles were added to the remit in 2000. The complete specification of requirements is detailed in the DETR contract (Contract No: CR0238).

1.3 Project outputs

This report represents the final output of the Cetacean Strandings and Investigations in the UK 2000-2004 (Contract ref: CR0238). Annual reports for 2000, 2001, 2002, 2003 and 2004 were also compiled and submitted to DETR/Defra by contracting organisations. In addition, short quarterly progress reports were submitted to DETR/Defra reporting on UK cetacean strandings recorded and the number and causes of death (by species) of all cetaceans examined at *post mortem* during that period. Data on location, species and date of all UK cetacean strandings is held at NHM where unique reference numbers are allocated. Pathological, toxicological and other data derived from post-mortem investigations on UK-stranded cetaceans are held at IoZ and SAC.

1.4 Key events and milestones in the 5-year period 2000-2004

- In the last 5 years around 3200 strandings were reported to the NHM with database records exceeding 10,000 by November 2003

- Since 2000, quarterly updates on strandings have been provided to Defra and numerous reports given in response to specific direct requests for ministerial purposes/parliamentary questions/discussion. Annual reports have been submitted to Defra for the 2000 to 2004 period.

- 18 species of cetacean were recorded in 2000-2004 period and there were no new additions to the fauna list for this region

- Since 1999 the total number of strandings reported has more than doubled; figures for porpoises show a similar increase while, for short-beaked common dolphins, the increase is more than 3-fold. Although a range of factors may be involved, an increase in reporting effort in south-west England is thought to play a major role.

- During the period of this report, 927 post-mortem examinations of 17 cetacean species (mainly harbour porpoises and common dolphins) were conducted. The principal causes of death in 563 UK-stranded harbour porpoises examined at *post mortem* between 2000 and 2004 were attack from bottlenose dolphins (*Tursiops truncatus*) (n=128), pneumonias due to combinations of parasitic, bacterial and fungal infections (n=102), entanglement in fishing gear (bycatch) (n=93) and starvation (n=74). The principal cause of death in 116 of the 190 (61.1%) common dolphins was bycatch (especially in South-west England).

- The first pathological evidence of acute and chronic gas and fat embolism were described and published in common dolphins (n=5), Risso's dolphins (n=4), harbour porpoises (n=2), a Blainville's beaked whale (n=1) and a Sowerby's beaked whale (n=1) during the period. The cause of, and contributing factors to, this novel disease entity are not known, although a mechanism similar to decompression sickness is suspected.

- Investigations of trends in pollutants in UK-stranded harbour porpoises showed that levels of organochlorine pesticides and butyltins were relatively low, but levels of PCBs and mercury were higher. While levels of both PCBs and mercury in harbour porpoises often occurred at levels toxic to other mammals, levels of selenium were positively correlated with mercury (suggesting some selenium-dependent detoxification of mercury). Levels of the newer brominated flame retardant compounds were also detected in UK harbour porpoises for the first time during this study.

- Epidemiological (case-control) studies demonstrated a statistically significant association between higher PCB levels in harbour porpoises dying from infectious diseases (compared to those dying due to acute physical trauma) that was not confounded by a range of additional factors including age, sex, stranding date and location, and two quantitative indices of nutritional status (mean blubber thickness and ratio of body-weight/body-length). These associations were consistent with PCB-induced immunosupression predisposing highly exposed harbour porpoises to infectious disease mortality (mainly parastic and bacterial infections).

- Since 1990, over 90 peer-reviewed scientific publications have been supported by the data and tissue archive generated by the UK Cetacean Strandings Programme.

2.0 CETACEAN STRANDINGS AND POST-MORTEM FINDINGS

2.1 Collection, analysis, and reporting protocols

MATERIALS AND METHODS

Reporting and collection of marine mammal strandings
The reporting, retrieval and transportation of marine mammal carcasses within England and Wales involves the integration of a number of localised reporting centres with the Institute of Zoology and the Natural History Museum (see below). IoZ works closely with the NHM and other organisations to ensure all cetacean carcasses (and a small number of seal carcasses) that are suitable for post-mortem examinations are promptly and successfully retrieved. The decision about whether to subject a carcass to post-mortem is based on the state of decomposition and whether it can be secured safely prior to collection and transportation to a laboratory for post-mortem examination. The relevant public health considerations of handling stranded marine mammal carcasses are stressed to those individuals and organisations that are involved with the day-to-day reporting and recovery of stranded marine mammal carcasses.

Central reporting centres
A list and full contact details for the central (The Natural History Museum, Institute of Zoology, Scottish Agricultural College, Inverness, Marine Environmental Monitoring) and regional (Cornwall Wildlife Trust; National Marine Aquarium, Plymouth) reporting networks for cetacean and marine turtle strandings is included in **Annex B.** See also:
http://www.nhm.ac.uk/research-curation/projects/strandings/

Reporting of live cetacean strandings

Across the UK, the RSPCA (England and Wales) and the Scottish SPCA (Scotland) are the first point of contact for members of the public reporting live-stranded or distressed marine mammals through their national hotlines. In recent years, the UK-based organisation *British Divers Marine Life Rescue* (BDMLR) has seen a large expansion of its activities including the training of over 2,500 BDMLR-trained Marine Mammal Medics. Further details of the organisation can be found on the BDMLR website at: http://www.bdmlr.org.uk/.

Pathology laboratories

The majority of post-mortem examinations were carried out at one of the following pathology laboratories.

- ***Scottish Agricultural College (Veterinary Science Division), Drummondhill, Stratherrick Road, Inverness, Scotland, IV2 4JZ.***
- Institute of Zoology (Zoological Society of London), Regent's Park, London, NW1 4RY.
- Veterinary Laboratories Agency (Truro), Polwhele, Truro, Cornwall, TR4 9AD.

- Department of Veterinary Pathology, University of Liverpool, Neston, Wirral, L69 3BX.

Standard protocols for post-mortem examination, tissue sampling and data recording

Carcasses were routinely transported to one of the four pathology laboratories listed above. In cases where carcasses were too large or too difficult to retrieve, post-mortem investigations were conducted *in situ* at the stranding site. All cetacean and marine turtle post-mortem investigations (including tissue sampling) are conducted using standard procedures (Law 1994) (**Annex B**). Essentially, organs are systematically examined and routine tissue samples are collected for virological, microbiological, histopathological, toxicological and other studies. Any observed lesions are also sampled for further diagnostic tests, depending on the suspected aetiology. Although it is often not possible to arrive at a definitive cause of death for any individual carcass, a probable cause of death is ascribed wherever possible based on the collective findings from post-mortem and other diagnostic investigations (**Annex B**).

Age and sex class determination
In all species, age was determined mainly by quantification of growth-layer groups from analyses of decalcified tooth sections (Lockyer 1995a). Where data from teeth ageing was not available for sexually immature harbour porpoises, age was estimated from body length (see Lockyer 1995b) as follows: <105cm = 0yo; 105-120cm = 1yo; 121-130cm = 2yo, and >130cm = 3yo. For some data analyses, harbour porpoises were classified as neonates (body length ≤ 90cm), juveniles (sexually immature with a body length > 90cm), and adults (sexually mature). Sexual maturity was established using gonadal appearance and histological evidence of spermatogenesis in testes.

Toxicological analyses.
All tissue samples were stored at –20˚C prior to preparation, and subsequently until analysed. Tissue samples were analysed for a range of trace elements, organochlorine pesticides, chlorobiphenyl and brominated diphenyl ether congeners according to previously established and fully validated protocols (**Annex B**).

For statistical analysis of toxicology data, blubber *p, p'*-DDE/ΣDDTs ratio (DDE/tDDTs) and the hepatic molar Hg:Se ratio were calculated. Two quantitative indices of nutritional status (relative body-weight and mean blubber thickness) were calculated from the residuals from the best-fit regression line of log(body-weight) on log(body-length), and from the mean of the three standard (dorsal, lateral, ventral) blubber thickness measurements, respectively (Law 1994). The stranding locations were divided into the following areas to test for regional variation in all contaminant levels: south-west England (Devon, Cornwall, Somerset and Gloucestershire); east coast of England (Northumberland to Essex); Wales and north-west England; the English Channel (Kent to Dorset); and Scotland.

Analysis of variance and regression models (Sokal & Rohlf 1995, SYSTAT 1998) were used to test for associations between levels of Σ25CBs, ΣDDTs, dieldrin, α-HCH, γ-HCH, HCB, Σ15BDEs (mg kg^{-1} lipid), Ag, As, Cd, Cr, Cu, Fe, Ni, Pb, Se, Zn, ΣBTs (mg kg^{-1} wet wt) and hepatic molar Hg:Se ratio (the dependent variables in the

models) and the following (independent) parameters: age, sex, nutritional status, region of stranding and year of stranding. Intercorrelations between selected individual contaminants or contaminant groups were also investigated using similar statistical methods. For logistical reasons the quantity of data for each contaminant differed, and complete data were not available for all variables (e.g. age). Therefore, sample sizes for the statistical tests differ. Statistical analyses were performed using the natural logarithms (ln) of all continuously distributed data to meet the assumption of normality required by the statistical models (Sokal & Rohlf 1995). Statistical analyses of subsets of these porpoises were also examined where deemed appropriate to investigate further any statistically significant associations.

Tissue archiving. Tissue specimens collected for research and archive are stored at both -20°C and -80°C and in 10% neutral buffered formalin or 70% alcohol at the Institute of Zoology and SAC Inverness or sent to collaborating institutions for research purposes.

Tissue archive and Poseidon central database

The national marine mammal tissue archive and '*Poseidon*' central database were developed under previous contracts to IoZ for marine mammal strandings investigation. The tissue archive provides an extensive range of tissues for retrospective analyses and currently over 50,000 individual samples have been collected during the course of the project, the majority of which are housed at the two main sites (IoZ and SAC). Poseidon is currently run on a Microsoft FoxPro 6.0 platform, although it is in the process of being transferred to a bespoke Access pathological database. Poseidon also holds records on over 13,500 individual strandings of cetacea, pinnipeds and more recently marine turtles, of which nearly 2,800 have been investigated at post mortem.

Reporting of data

Data resulting from this contract has been routinely reported to the Department through annual contract progress reports. Annual data on numbers of strandings of each cetacean species examined, together with the main causes of death, have also regularly been submitted to the Department's Digest of Environmental Statistics (Wildlife Chapter). Occasionally, data have also been submitted to the Department for the purposes of answering Parliamentary Questions. A number of research papers have also been published within the scientific literature (see **Annex D**).

2.2 General observations of trends in UK cetacean strandings

Systematic recording began in England and Wales in 1990 and in Scotland in 1992. Between 2000 and 2004, 18 species of cetacean have been recorded, consistent with diversity figures for the last quarter of a century, but no new additions to the list of cetacean species were recorded during this period. The number of cetacean strandings reported has increased annually since 2000 reaching a high of 799 in 2004 (**Table 1 and Figure 1**). While there have been a relatively consistent number of Scottish cetacean stranding reports in the last 5 years, the main increase in annual

stranding reports has occurred in England and Wales. Most of the increase comes from just one area (south-west England) where similar increases in both harbour porpoises and common dolphins occurred during this period (**Figure 1** and **Table 1**). These trends suggest that the cause of these increasing stranding reports is not biological. The relatively high number of cetacean strandings reported (mainly common dolphin and harbour porpoise bycatches) in south-west England since the early 1990s has undoubtedly lead to a greater awareness of the bycatch issue. It would, therefore, not be surprising if it also precipitated increased coastal vigilance for cetacean strandings. The annual number reported of most other cetacean species have shown relatively small inter-annual fluctuations.
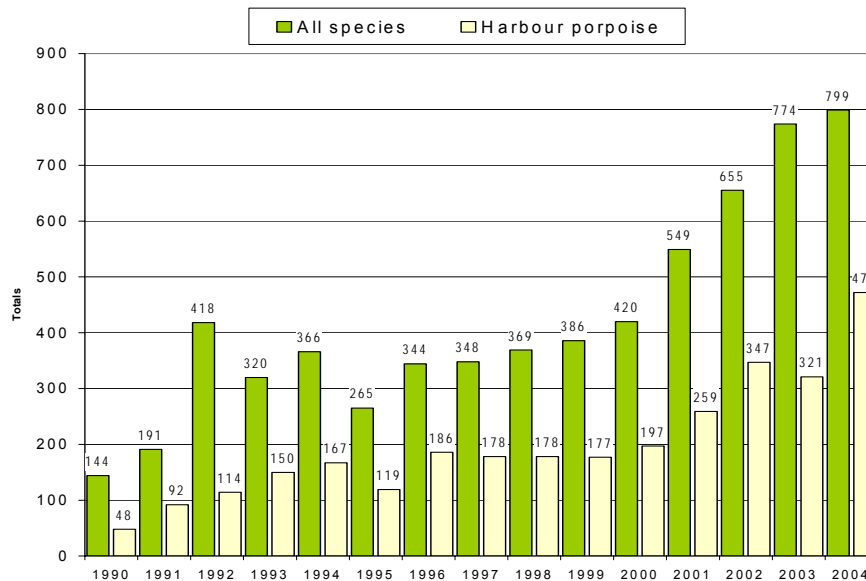
**Figure 1. Total numbers of reported UK cetacean strandings etc.,
1990 – 2004, compared with annual numbers of harbour porpoise.**

**Table 1. A summary of recorded cetacean species stranded around the UK,
1993 - 2004**

	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004
BALAENOPTERIDAE												
Minke whale	15	3	10	7	15	11	16	19	14	18	17	14
Sei whale	-	-	-	-	-	-	-	-	1	-	-	-
Fin whale	-	1	2	-	-	1	-	4	-	-	-	7
Balaenoptera sp.	-	-	-	-	-	-	-	-	-	-	-	1
Humpback whale	-	-	-	-	-	-	-	-	2	-	-	2
DELPHINIDAE												
Common dolphin	47	43	20	42	46	43	44	65	126	119	209	159
Common/striped dolphins	-	1	4	5	4	6	6	1	3	7	3	8
Short-fined pilot whale	10	8	18	12	16	14	16	16	27	21	5	5
Risso's dolphin	9	7	8	9	4	14	12	11	6	11	7	10
Killer whale	-	10	4	-	1	-	-	-	1	1	1	-
Fraser's dolphin	-	-	-	1	-	-	-	-	-	-	-	-
Atlantic white-sided dolphin	7	20	8	10	17	16	5	14	4	4	12	5
White beaked dolphin	11	15	16	11	9	8	8	8	14	7	5	9
Lagenorhynchus sp.	1	5	-	3	-	2	1	3	2	-	1	1
Striped dolphin	4	9	11	9	4	17	17	13	14	9	12	6
Bottlenose dolphin	10	5	6	1	2	6	10	5	9	6	7	8

Unidentified dolphins	19	20	13	15	17	16	13	22	29	51	114	55
PHOCOENIDAE												
Harbour porpoise	150	167	119	186	178	178	177	197	259	347	321	472
PHYSETERIDAE												
Pygmy sperm whale	1	-	-	-	2	-	2	-	-	1	-	-
Sperm whale	5	18	4	8	8	8	4	6	6	4	8	3
ZIPHIIDAE												
Northern bottlenose whale	2	1	2	1	-	1	-	1	3	-	1	2
Sowerby's beaked whale	-	3	-	2	-	1	1	3	3	1	1	3
Blainville's beaked whale	1	-	-	-	-	-	-	-	-	-	-	-
Cuvier's beaked whale	3	1	-	-	-	-	2	-	-	3	1	2
Unidentified odontocetes	13	20	6	5	15	16	17	7	4	18	24	16
Unidentified cetaceans	12	9	14	17	10	11	13	25	22	27	25	11
TOTALS	320	366	265	344	348	369	386	420	549	655	774	799

In the period 1992 to 2003, the relative frequency of strandings of white beaked dolphins in north-west Scotland, a colder water species, has declined while strandings of common dolphin, a warmer water species, have increased. Similarly, sightings surveys conducted in May-September 2002 and 2003 show that the relative occurrence and abundance of white beaked dolphins have declined and common dolphins increased in comparison to previous studies. (MacLeod *et al.* 2005). The striped dolphin (*Stenella coeruleoalba*) has also increased steadily in strandings figures and for the last decade it has ranked as the sixth most frequently found cetacean on our coastline (**Table 1**).

2.3 Trends in cetacean strandings - Species trends

The full details of spatial and temporal trends in reports of cetacean strandings in the UK between 2000 and 2004 are provided in **Annex C**. Only a brief summary of stranding patterns for each species is included here.

BALAENOPTERIDAE

Minke whale (*Balaenoptera acutorostrata*)

The minke whale is the smallest but most commonly encountered baleen whale in UK waters and has on average ranked as the fifth most commonly stranded species from the entire NHM strandings record. It ranked fifth in the 1990-1994 period with 39 reported strandings but both its ranking and absolute numbers have increased (to third and 83 animals, respectively) in the 5-year period under review (**Table 1**). The stranding distribution mainly in the north of the UK with only a few records from the North Sea coast is consistent with recent years (Sheldrick *et al.* 1991). There were 19 reported strandings in 2000, the highest annual figure on record, and 2002 and 2003 each had 18 records.

Sei whale (*Balaenoptera borealis*)

In September 2001 a sei whale stranded alive on the Lancashire coast and later died. It was one of only 13 recorded in the UK since 1913 and was the first to be reported since1990.

Fin whale (*Balaenoptera physalus*)

The fin whale has also shown evidence of a slow increase since the 1980s with the highest number of strandings (n=11) occurring within the five years since 2000. Few had been reported since the 1940s and the current frequency has only appeared in earlier records between 1915 and 1925 (**Figure 2a and 2b**). Historically, only 61 animals were recorded stranded in the UK. Most were found between 1915 and 1925 with another 7 records in the 1940s, but with very low instances at other times until the 1980s when strandings began to reappear.
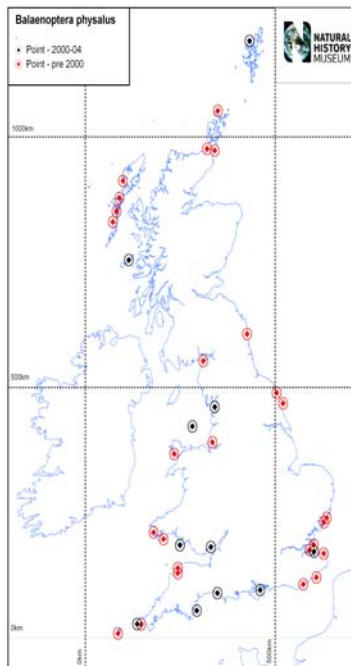
Figure 2a: UK records of stranded fin whales 1913-1999 (blue dots) and 2000-2004 (red dots)
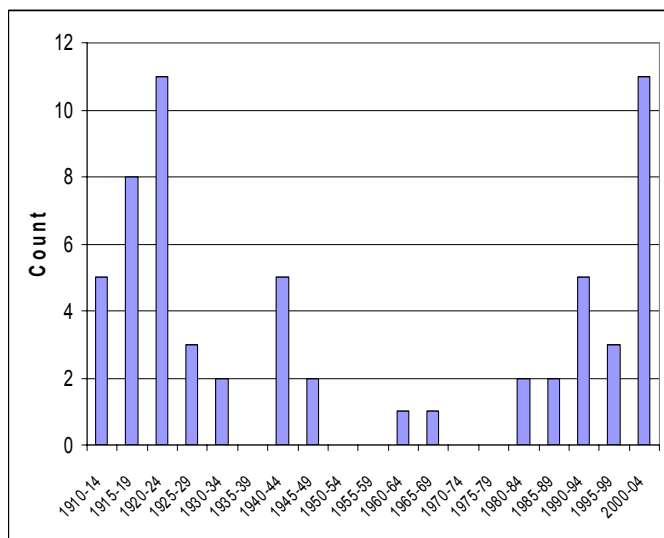
Figure 2b. Frequency of recorded fin whale strandings in the UK 1910-2004

Humpback whale (*Megaptera novaeangliae*)

Among the more unusual species in the British record, the humpback whale (*Megaptera novaeangliae*) was reported on five occasions in the last five years, the only previous records having been two single strandings in the 1980s. Reports suggest that humpback whales tend to occur more frequently in northern British coastal waters, but in March 2001 the first stranding on the English coastline was recorded at Pegwell in Sandwich Bay, Kent. This live-stranded humpback whale was the most southerly record of stranding of this species in the UK. On 13th September 2004 the first record for Northern Ireland was received from the Giant's Causeway, Co. Antrim, as well as another from the Orkney Isles, Scotland in February of that year. Examination of the testes from the Kent stranding revealed that this male was sexually immature and therefore probably between one and three years of age.

DELPHINIDAE

Short-beaked common dolphin (*Delphinus delphis*)

The spatial distribution of the 678 common dolphin strandings reported between January 2000 and December 2004 inclusive are shown in **Figure 3a**. Most common dolphins and indeterminate dolphins were carcases in the south-west of Britain (**Table 1 and Figure 3b**). Common dolphin strandings averaged only two or three per year until the 1970s, before reports rose steadily to around a dozen annually by the end of the 1980s. When reporting effort increased in 1990, annual reported strandings rose to around 40 for that decade. For the last five years, from 2000-2004 that figure has risen to an average of 136. The number of common dolphin strandings reported peaked at 209 in 2003. The majority of strandings occur in the first three or four months of each year.
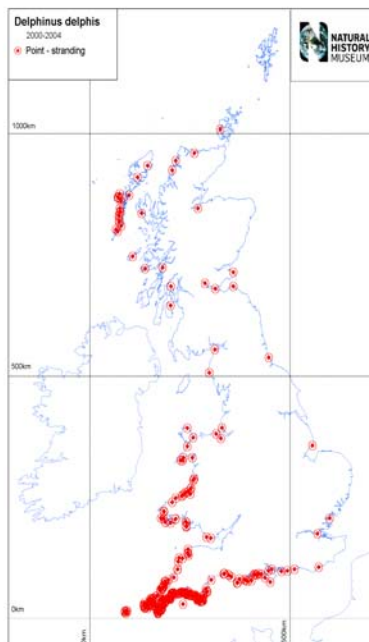
Figure 3a: Distribution of UK-stranded common dolphins reported (2000-2004)
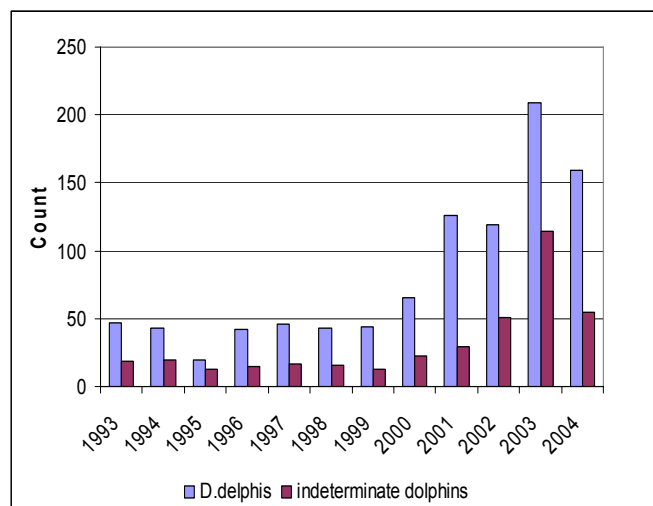
Figure 3b.   Frequency of strandings common dolphin vs indeterminate dolphins 1993-2004

MacLeod *et al*. (2005) have suggested that, for NW Scotland, there has been a relative increase in sightings and strandings of common dolphins (whose range extends south and westwards) since 1981 compared with a decline in reports of white beaked dolphins (*Lagenorhynchus albirostris*), whose range extends north and eastwards in the same region. Monthly strandings figures show that common dolphins historically strand in Scottish waters all year and that most are found in the winter months, peaking around February and March.

Long-finned pilot whale (*Globicephala melas*)

Between 5 and 27 pilot whales per annum (74 in total) were recorded during the period of this report. Pilot whale strandings are usually recorded annually in the UK and this species has occurred in widely fluctuating numbers as a result of its propensity to occasionally strand in massive groups. In 1950 two such events occurred involving 97 and 148 animals respectively. This variability in stranding has led to similar changes in its rank in abundance. It has often been the most frequently stranded species of cetacean on the British coastline, exceeding the number of porpoises in those years. But in recent years there have been no recorded instances of mass stranding involving more than two pilot whales. The most recent event involving more than two animals was in April 1992 when seven whales stranded together, and the last sizeable mass event occurred in January 1985. Only one stranding of this species has been recorded in the North Sea in the last five-year period.

Risso's dolphin (*Grampus griseus*)

There are 112 records of Risso's dolphin strandings in the UK between 1913 and 1989 and 45 recorded between 2000 and 2004 inclusive. Data for the five years from 2000 to 2004 clearly show a west coast distribution with the highest concentrations in Scottish waters (**Annex C**). Since 1992, Scottish strandings have accounted for 76% of all British records. Sporadic strandings of Risso's dolphins have been recorded in the English Channel and in the northern part of the North Sea, but there have been no North Sea records since a stranding at Lossiemouth in 1992. With the exception of a solitary stranding at Cleethorpes in Lincolnshire in 1933, the other North Sea strandings have all been restricted to the east coast of Scotland, mainly on the most northerly shores.

Killer whale (*Orcinus orca*)

Although sightings data show killer whales in UK waters throughout the year (Reid *et al.* 2003), strandings of killer whales are not reported every year on UK shores. Only 85 killer whales have been recorded stranded from 1913 to 2004. In the period under review three strandings were reported; one each in 2001, 2002 and 2003. Most strandings occurred in the north and west of Scotland, but this species has been found around the entire coastline. The three most recent strandings were all recorded in England (Merseyside, Humberside and Cumbria).

Atlantic white-sided dolphin (*Lagenorhynchus acutus*)

A total of 39 Atlantic white-sided dolphins were recorded during the period of the report. The species is gregarious, and mass strandings (an event involving two or more animals other than a mother-calf pair, stranding at the same time and location) is not uncommon. In the 1920's, three separate events, all in the Shetland Isles, each involved over 30 animals. Since then, reports of mass strandings have all been of much smaller groups. In the last five years three mass strandings were recorded, involving six, five and two animals, respectively. Most strandings of this species occur on the Scottish coast with only rare strandings occurring in England and Wales.

There has been a change in the ratio of the two species of *Lagenorhynchus* in British waters in recent years, based on the evidence from strandings data. For most of the period from 1913 to 1966 stranding reports of Atlantic white-sided dolphins were only about 30% of those for white beaked dolphins in the UK. From 1967 to 1989 this increased to around 60% and increased again to 85% in the last 15 years from 1990 to 2004. In the last ten years of this period the numbers of each of the two species were more-or-less similar although in some years, reports of Atlantic white-sided dolphins have outnumbered white beaked dolphins.

White beaked dolphin (*Lagenorhynchus albirostris*)

A total of 43 white beaked dolphins were recorded during the period of this report. Although most frequently sighted at sea in Scottish and North Sea waters, sightings have also been recorded as far south as Brittany (Reid et al. 2003). Most of the British strandings have occurred on the east coasts of England and Scotland, but there are also occasional strandings records in southwest England.

In a status review of marine mammals in the southern North Sea, Klinowska (1987) described the white beaked dolphin as the third most commonly reported species in North Sea sightings and strandings records. Although there have been substantially more records in the last fifteen years, possibly as a result of increased reporting effort and reporting procedures in Scotland since 1992, the number of reports has declined since the start of this period. Since 1991 numbers of reported strandings have fluctuated between 5 and 19 per year with an annual average of 9 over the last 5 years.

Since around 2000, there has been a change in the geographical distribution of strandings (see **Annex C**) with fewer strandings records from the southern North Sea and a greater number from Yorkshire to the extreme north of Scotland. In the 1990s the bulk of strandings still occurred on the east coast from Norfolk to the north-east of England and Scotland. But at the same time more regular strandings began to occur in the northern island groups of Orkney and Shetland. There was an absence of records from the southern North Sea from 1997-1999 when Lincolnshire was the southernmost limit of strandings. The geographical distribution of these records suggests a possible shift in distribution. In 2000 there were further indications that the distribution of this species may have moved northwards with all reports coming from Orkney, Shetland or the Highland region. In 2001, again all occurrences of this species stranding were in Scotland, including a mass stranding event in the Shetland Isles involving eight animals.

A study on sightings and strandings of cetaceans, including white beaked dolphins, in the north-west region of Scotland reports some changes in distribution consistent with possible effects of climate change (MacLeod *et al.* 2005).

Striped dolphin (*Stenella coeruleoalba*)

The similarity in size and shape to common dolphins may have produced some confusion in the identification of these two species on occasion but any early rarity in the record of striped dolphin was not due to lack of awareness of the possibility of its being found amongst British strandings (Harmer, 1927). The striped dolphin has

never been particularly abundant in the strandings records but it has been reported in gradually increasing numbers since the 1970s. For the past ten years it has ranked as the sixth most frequently stranded cetacean on our coast.

The number of striped dolphins ranged between 6 and 14 per year between 2000 and 2004, with a total of 54 for the period. The spatial distribution of striped dolphin strandings between 2000 and 2004 is shown in **Annex C**. Strandings have always been greatest in south-western waters, but Muir *et al.* (2000) reported a spread in distribution in the 1995-1999 quinquennium to include Scotland. MacLeod *et al.* (2005) considered its presence in north western Scottish waters, along with changing patterns for other species, to be possibly indicative of climate change although reports in the most recent five-year period from northern parts of Britain have declined. The only part of the country where the number of striped dolphin stranding reports has continued to rise is in the south-west of England, in common with recent increasing numbers of harbour porpoise and common dolphin in the same region.

Bottlenose dolphin (*Tursiops truncatus*)

Thirty-five bottlenose dolphins were reported stranded in the UK during the period of this report. The spatial distribution of UK-stranding reports of bottlenose dolphin is shown in **Annex C**. This is a species whose abundance in the stranding record has changed significantly with time and one for which the geographical distribution of strandings has also undergone some marked changes. Broadly, bottlenose dolphins were a greater proportion of all cetacean strandings in the earlier part of the record than they have been since the 1970's. Between the 1940's and the 1960's these dolphins were the second or third most commonly reported cetacean species, but their ranking has dropped to tenth or eleventh in the last 15 years. From 1948 to 1966, it was reported in concentrations on the north-western British coast around the Irish Sea and in the south-east along the English Channel to East Anglian shores (Fraser 1974). From 1960 to 2004 there was a decline in stranding reports from the north-west and a more gradual decline in the south-east. Since 1990, most strandings have occurred in West Wales and the Moray Firth reflecting the (semi-)resident bottlenose dolphin populations concentrated in those regions. Stranding reports of bottlenose dolphins have historically been recorded to occur in all months of the year and peaking in summer.

Unidentified dolphins

The number of unidentified dolphins recorded reached a high of 114 in 2003 when the number of common dolphin strandings was also at its peak. These are generally carcases too decomposed, incomplete or inaccessible for examination and retrieval. Most were found in south-west England between January and March.

PHOCOENIDAE

Harbour porpoise (*Phocoena phocoena*)

The spatial and temporal distribution of harbour porpoise stranding reports is shown in **Figure 1**, **Table 1** and **Annex C**. Harbour porpoises represent approximately half of all cetaceans that strand annually in the UK. In the period under review, the

percentage has fluctuated more than usual with 59% in 2004 compared with only 41% in 2003. When systematic recording was introduced in 1990, reports of harbour porpoise strandings increased steadily for a number of years, but from 1996 to 2000 the figure was remarkably stable. Since 2001 the number of reports of porpoise strandings has increased significantly from 259 to 474 in 2004.

The most pronounced rise in harbour porpoise stranding reports has been in the south-west of England where numbers have increased from 353 in 1995-1999 to 897 in 2000-2004. The increase in harbour porpoise strandings in south-west England is mirrored by proportional increases of reports of other species in this region (particularly common dolphins), suggesting that the cause is not biological. It is possible that increasing awareness of bycatch as a common cause of cetacean mortality and strandings in south-west England has driven increasing coastal surveillance in this region during the period of this report.

More detailed examination of the data reveals a seasonal difference in the distribution of strandings between Wales and south-west England. The Welsh reports occur mainly in the summer months while strandings in south-west England predominate in winter (**Annex C**). Although these data suggest a possible north-south seasonal migration between south-west England and Wales, differences in ratios of body length to age have been detected between harbour porpoises from south-west England and other UK regions (including Wales) (**Annex C**) (Jepson 2003).

PHYSETERIDAE

Pygmy sperm whale (*Kogia breviceps*)

Pygmy sperm whales have appeared slightly more frequently in the strandings record since 1980, peaking with four individuals between 1995 and 1999 (Muir *et al.*, 2000), but in the most recent five year period only one report of this whale was received. Since the NHM began keeping records in 1913 there have been nine reports of stranded pygmy sperm whales, the first in 1966. The distribution included strandings on the south-west coast of Scotland, south-west of England and south Wales. The most recent and southernmost record occurred at Thurlestone, Devon in January 2002. These animals are difficult to separate from their congener, dwarf sperm whale, at sea, but the limited sightings data for this species suggest a low level occurrence in the Bay of Biscay, southwestern approaches and Irish Atlantic waters (Reid *et al* 2003).

Sperm whale (*Physeter catodon*)

Twenty-seven individuals were recorded during the period of this report. The sperm whale is distributed widely in the oceans of the world with its main concentrations in warmer waters, but its range extends into colder waters with indications that old males have a preference for these higher latitudes. In the last ten years the number of reported strandings has been stable with expected annual fluctuation, although the total of only three in 2004 was unusually low. The longer term trend (**Annex C**) shows that there has been a steady rise in the number of UK stranding reports since

the 1960's.

ZIPHIIDAE

Relatively few strandings of beaked whales occurred during the period of this report comprising 11 Sowerby's beaked whales (*Mesoplodon bidens*), seven northern bottlenose whales (*Hyperoodon ampullatus*) and six Cuvier's beaked whales (*Ziphius cavirostris*) reported between 2000 and 2004 (**Annex C**). All strandings were single.

Unidentified toothed whales and other cetaceans

It is often not possible to identify carcases which are very inaccessible or in an advanced state of decomposition. The numbers of unidentified odontocetes and cetaceans that could not be identified as either toothed or baleen whales are given in **Table 1**.

2.4 Causes and results of cetacean and marine turtle mortality (2000-2004)

Between 1st January 2000 and 31st December 2004, 892 stranded and 35 bycaught cetacean carcasses were examined at *post mortem* in the UK (**Table 2**). Of the stranded cetaceans, 320 carcasses were examined in Scotland (**Table 3**) and 572 were examined in England and Wales (**Table 4**). The annual numbers of cetacean post-mortem examinations conducted in Scotland were relatively constant during the period of the report (n=51-73), but the annual number of UK-stranded cetacean post-mortem examinations increased annually from 2000 (n=67) to 2004 (n=159) in England and Wales, predominantly owing to increased short-beaked common dolphin (*Delphinus delphis*) and harbour porpoise (*Phocoena phocoena*) strandings reported in south-west England during the winter. Five UK-stranded marine turtles were examined during the period of the report (**Tables 2-4**). **Table 5** summarises the causes of death of 927 cetaceans and five marine turtles examined *post mortem* in the UK between 1st January 2000 and 31st December 2004.

Table 2: Total UK Cetacean and Turtle Examinations 1st Jan 2000-31st Dec 2004

Species	Number
Harbour porpoise (*Phocoena phocoena*)	563
Short-beaked common dolphin (*Delphinus delphis*)	190
Striped dolphin (*Stenella coeruleoalba*)	33
Atlantic white-sided dolphin (*Lagenorhynchus acutus*)	27
White beaked dolphin (*Lagenorhynchus albirostris*)	16
Bottlenose dolphin (*Tursiops truncatus*)	14
Minke whale (*Balaenoptera acutorostrata*)	12
Risso's dolphin (*Grampus griseus*)	12
Sowerby's beaked whale (*Mesoplodon bidens*)	7
Long-finned pilot whale (*Globicephala melas*)	7
Sperm whale (*Physeter catodon*)	3

Fin whale (*Balaenoptera physalus*)	2
Northern bottlenose whale (*Hyperoodon ampullatus*)	1
Pygmy sperm whale (*Kogia breviceps*)	1
Humpback whale (*Megaptera novaengliae*)	1
Killer whale (*Orcinus orca*)	1
Cuvier's beaked whale (*Ziphius cavirostris*)	1
Unknown delphinid	1
Loggerhead turtle (*Caretta caretta*)	3
Green turtle (*Chelonia mydas*)	1
Leatherback turtle (*Dermochelys coriacea*)	1
Total	**897**

Table 3: Cetaceans and Turtles Examinations in Scotland 1st Jan 2000-31st Dec 2004

Species	Number
Harbour porpoise (*Phocoena phocoena*)	209
Common dolphin (*Delphinus delphis*)	20
Atlantic white-sided dolphin (*Lagenorhynchus acutus*)	24
White beaked dolphin (*Lagenorhynchus albirostris*)	15
Striped dolphin (*Stenella coeruleoalba*)	11
Minke whale (*Balaenoptera acutorostrata*)	11
Bottlenose dolphin (*Tursiops truncatus*)	9
Risso's dolphin (*Grampus griseus*)	9
Sowerby's beaked whale (*Mesoplodon bidens*)	5
Pilot whale (*Globicephala melas*)	5
Sperm whale (*Physeter catodon*)	1
Northern bottlenose whale (*Hyperoodon ampullatus*)	1
Loggerhead turtle (*Caretta caretta*)	2
Total	**322**

Table 4: Cetaceans and Turtles Examinations in England and Wales 1st Jan 2000-31st Dec 2004

Species	Number
Harbour porpoise (*Phocoena phocoena*)	354
Common dolphin (*Delphinus delphis*)	170
Striped dolphin (*Stenella coeruleoalba*)	22
Bottlenose dolphin (*Tursiops truncatus*)	5
Atlantic white-sided dolphin (*Lagenorhynchus acutus*)	3
Risso's dolphin (*Grampus griseus*)	3
Sperm whale (*Physeter catodon*)	2
Sowerby's beaked whale (*Mesoplodon bidens*)	2

Pilot whale (*Globicephala melas*)	2
Fin whale (*Balaenoptera physalus*)	2
White beaked dolphin (*Lagenorhynchus albirostris*)	1
Minke whale (*Balaenoptera acutorostrata*)	1
Pygmy sperm whale (*Kogia breviceps*)	1
Humpback whale (*Megaptera novaengliae*)	1
Killer whale (*Orcinus orca*)	1
Cuvier's beaked whale (*Ziphius cavirostris*)	1
Unknown delphinid	1
Loggerhead turtle (*Caretta caretta*)	1
Green turtle (*Chelonia mydas*)	1
Leatherback turtle (*Dermochelys coriacea*)	1
Total	**575**

Entanglement in fishing gear (Bycatch)

Diagnosis of bycatch-related deaths followed the established criteria for bycatch diagnosis established by Kuiken *et al.* (1994a, 1994b). Within the UK 116/190 (61.1%) stranded common dolphins were diagnosed as bycatches between 1st January 2000 and 31st December 2004 inclusive, of which 113 were found in the south-west of England (Cornwall, Devon and Dorset). For UK-stranded harbour porpoises, 93/563 (16.5%) were diagnosed during the same period of which 67 stranded in England, 15 in Wales, and 11 in Scotland. Another 35 by-caught harbour porpoise carcasses were retrieved for post-mortem examination directly from fishing vessels as part of observer-based research co-ordinated by the Sea Mammal Research Unit (University of St. Andrews). Bycatch was also diagnosed as the cause of death of 5/33 (15.2%) striped dolphins, 1/27 (3.7%) Atlantic white-sided dolphins and 1/12 (8.3%) Risso's dolphins examined between January 2000 and December 2004 inclusive. Six minke whales were diagnosed to have been entangled (probably in rope), so it was not clear if these were additional bycatches in commercial fisheries or other types of entanglement in rope (e.g. mooring buoy). A leatherback turtle was also diagnosed as bycatch. The spatial distribution of all stranded cetaceans diagnosed as by-caught (by species) between 2000 and 2004 is shown in **Figure 5 a-c**.

By-caught harbour porpoises frequently presented with thin, linear cutaneous cuts or depressions on the leading edges of the pectoral fins, tail flukes and dorsal fin, or had similar lesions that sometimes partly or completely circumscribed the head, consistent with the use of large-meshed monofilament set nets. It should be noted, however, that many of the numerous bycatches in South-west England strand in a state of "moderate decomposition", in which the skin on the beak, fins and flukes is often partly abraded or peeled-off, making detection of subtle net-induced lesions difficult.
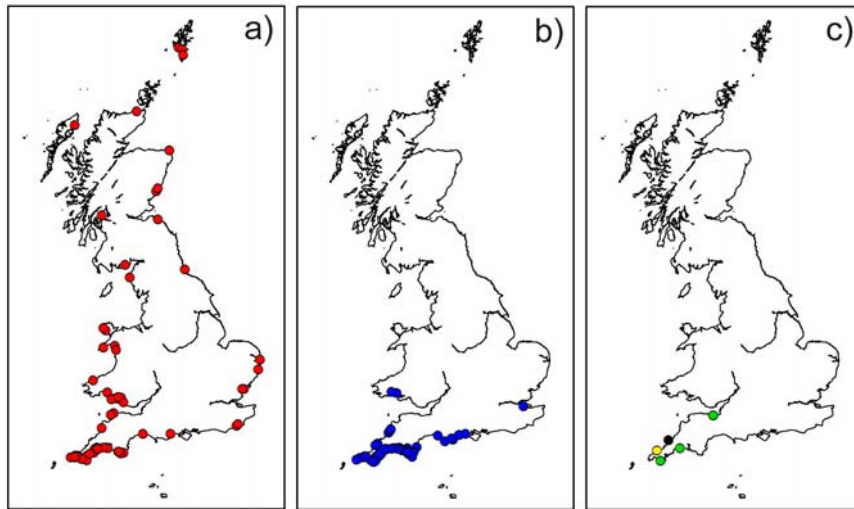
Figure 5a-c Spatial stranding distribution of; a) 93 harbour porpoises (in red); b) 116 common dolphins (blue): c) five striped dolphins (green), one white sided dolphin (yellow) and one Risso's dolphin (black) stranded and diagnosed bycaught between Jan 2000 and Dec 2004

Physical trauma (including fatal attack from bottlenose dolphins)

Between January 2000 and December 2004 inclusive, 155/563 (27.5%) UK-stranded harbour porpoises, 4/190 (2.1%) common dolphins, 1/16 (6.3%) white beaked dolphins and 1/2 (50%) fin whales died from acute physical trauma (excluding bycatch). Using established pathological criteria (Jepson & Baker 1998), fatal attack by bottlenose dolphins was the single most common cause of death diagnosed in UK-stranded harbour porpoises during the period of this report (2000-2004) accounting for 128/563 (22.7%) of all UK-stranded harbour porpoises examined. These individuals comprised 59 porpoises from Scotland (generally between the Moray Firth and Firth of Forth areas), 64 from West Wales, and five from South-west England (Cornwall, Devon and Dorset) (**Figure 6**). As in previous years, most porpoises killed by bottlenose dolphins in all regions were juveniles (76%) compared to adults (22%) and neonates (body length ≤90cm) (2%).
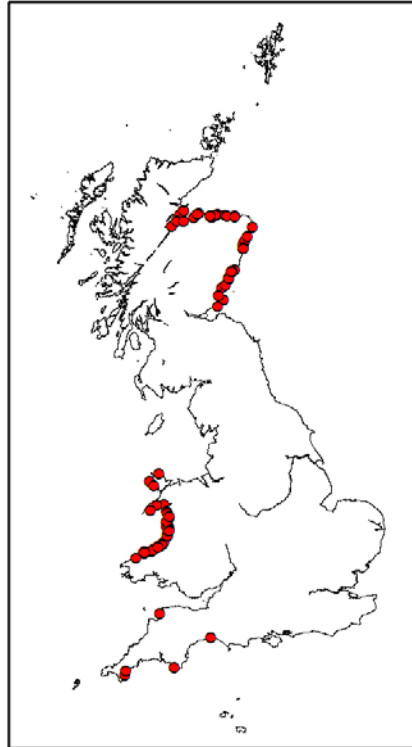
Figure 6 Spatial distribution of 128 harbour porpoise strandings killed by bottlenose dolphins between Jan 2000 and Dec 2004

Apart from bycatch and bottlenose dolphin attacks, other fatal cases of acute physical trauma during the period of this report (2000-2004) included one harbour porpoise, one common dolphin and one fin whale with injuries consistent with a fatal impact from a boat strike. There were 26 harbour porpoises, three common dolphins and one white beaked dolphin that died of acute physical trauma of unspecified origin. A high proportion of these unidentified trauma cases were probably fatal impact(s) from watercraft though some could also be undiagnosed bycatches or bottlenose dolphin attacks.

Infectious disease mortality
Between January 2000 and December 2004, 180 UK-stranded cetaceans comprising 155/563 (27.5%) harbour porpoises, 7/190 (3.7%) common dolphins, 7/33 (21.2%) striped dolphin, 4/16 (25%) white beaked dolphins, 4/12 (33.3%) Risso's dolphins and 3/27 (11.1%) Atlantic white-sided dolphins died from infectious diseases. The spatial distribution of all stranded cetacea that died as a result of infectious disease (by species) between 2000 and 2004 is shown in **Figure 7**.
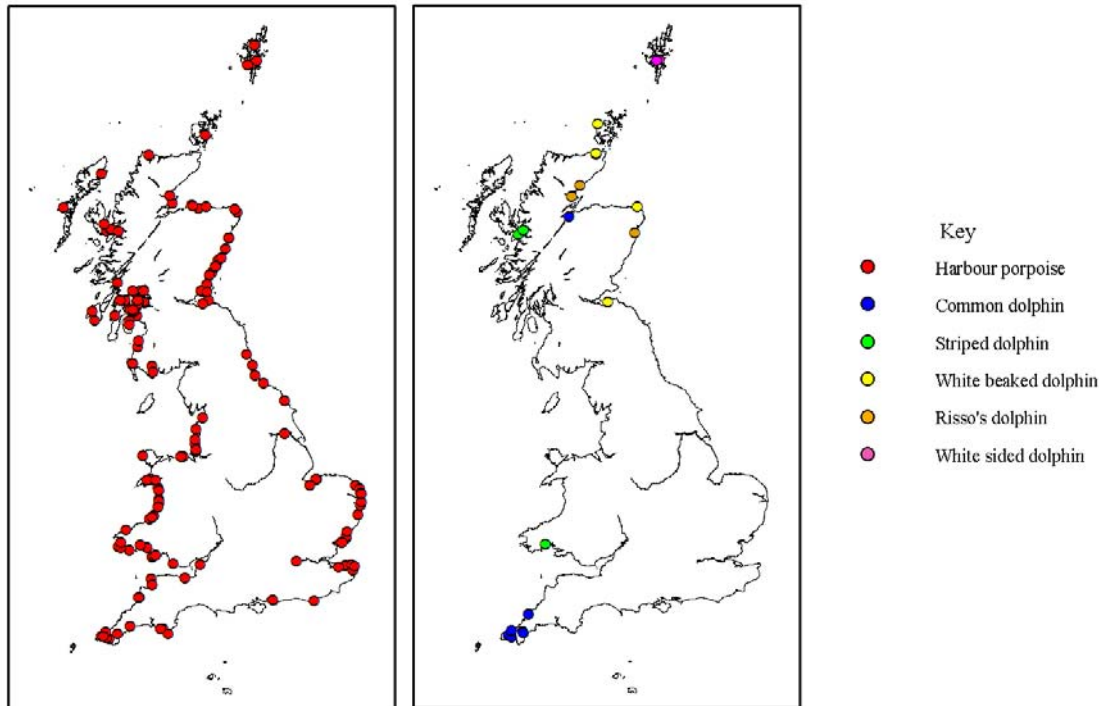
Figure 7 Spatial distribution of stranded cetaceans diagnosed to have died as a result of infectious disease between Jan 2000 and Dec 2004

In the porpoises, parasitic infections of the lungs resulting in severe airway obstruction, acute pulmonary haemorrhage, parasitic pneumonia or secondary bacterial or fungal infections caused the death of 101 individuals and generalised bacterial infections accounted for another 29 individuals. Other infectious disease causes of death included gastric or enteric parasitic or bacterial infections (n=12), meningoencephalitis (n=3), bacterial or mycotic otitis media (n=3), peritonitis (n=2) and single cases of bacterial pericarditis, pyelonephritis, mandibular osteomyelitis, pulmonary parasitism/oesophageal impaction and pneumonia of unknown origin.

Four common dolphins died of gastric parasitism, two died of infectious pneumonias, and one died of peritonitis. Five striped dolphins that stranded alive had meningitis/meningoencephalitis (three of which were associated with *Brucella* sp. infections), one died of parasitic gastritis and one died of peritonitis. Single cases of parasitic and bacterial pneumonia, heavy pulmonary parasite infestation, meningoencephalitis and a heavy gastrointestinal parasitism with secondary bacterial enteritis were diagnosed in the four white beaked dolphins. Three Risso's dolphins died of meningoencephalitis and one died as a result of peritonitis. Single cases of *Brucella* meningoencephalitis, bacterial pneumonia and a generalised bacterial infection with *Brucella* were diagnosed in Atlantic white-sided dolphins. There were no cases of distemper due to morbillivirus infection in any UK-stranded cetacean carcasses examined between 2000 and 2004.

Starvation

Starvation was attributed as the cause of death in animals that were severely emaciated in the absence of any other underlying disease processes that could explain the poor nutritional status. The death of 73 harbour porpoises (including 26 neonates), seven common dolphins (including one neonate), three striped dolphins, two white beaked dolphins, two Atlantic white-sided dolphins, two minke whale calves, two Risso's dolphin, one bottlenose dolphin, one killer whale, one pilot whale, one Cuvier's beaked whale and one humpback whale were attributed to starvation. The spatial distribution of these cases is shown (**Figure 8**). The deaths of three stranded loggerhead turtles were also attributed to starvation during the period of this report.
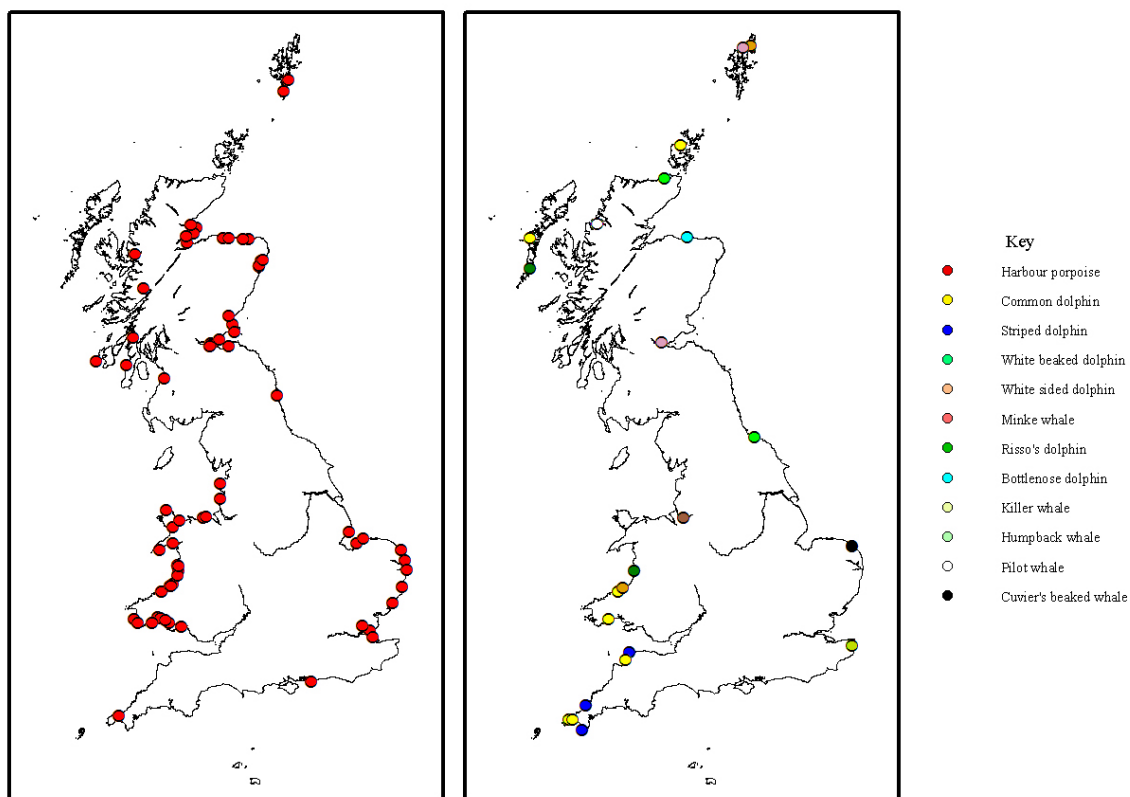

Figure 8 Spatial distribution of stranded cetaceans diagnosed to have died as a result of starvation between Jan 2000 and Dec 2004

Live stranding

Live stranding was attributed as the cause of death in animals that were known or suspected (from post-mortem examination) to have live-stranded while in apparent good health and nutritional status. This category excluded severely diseased or emaciated animals that were suspected to have stranded alive *in extremis* and in which the cause of death was attributed to the disease process rather than the live-stranding event.

Live stranding was attributed as the cause of death of 29 common dolphins, 21 Atlantic white-sided dolphins, 19 harbour porpoises, 9 striped dolphins, seven white beaked dolphins, six Sowerby's beaked whales, five pilot whales, three sperm whales, two bottlenose dolphins, two minke whales, one Risso's dolphins, one northern bottlenose whale and a fin whale. The spatial distribution of these cases is shown (**Figure 9**).
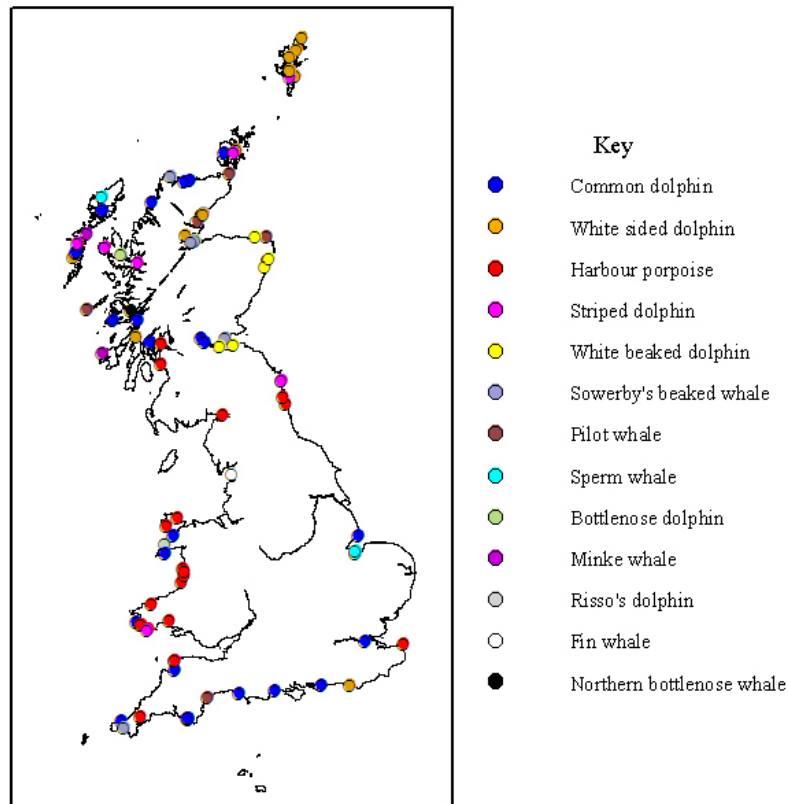
Figure 9 Spatial distribution of stranded cetaceans diagnosed to have died as a result of live stranding between Jan 2000 and Dec 2004

Neoplasia (Tumours)

A harbour porpoise that stranded in Scotland in 2000 had a myeloid leukaemia. A harbour porpoise that stranded in Scotland in 2001 had uterine tumours (fibroleiomyomas) (Patterson *et al.* 2002) associated with dystocia and foetal maceration. A harbour porpoise that stranded in England in 2003 had an adrenocortical carcinoma. A fourth harbour porpoise that stranded in Wales had a suspected pulmonary neoplasm but the tissues were too autolysed for a more specific characterisation of the lesion.

Gas embolism (see separate section)

During the period of this report, 13 UK-stranded cetaceans consisting of five common dolphins, four Risso's dolphins, two harbour porpoises, a Blainville's beaked whale and a Sowerby's beaked whale died of a novel disease process involving the development of acute and chronic macroscopic and microscopic cavitary lesions associated with *in vivo* gas and fat embolism within the liver and other tissues (*see* Gas Embolism

section). A detailed description of the pathology of 10 of these UK-stranded cases was published in the scientific literature (Jepson *et al*. 2005b).

Other causes of mortality

The remaining causes of death of stranded cetaceans and marine turtles examined between 1st January 2000 and 31st December 2004 are detailed in **Table 5**.

Table 5: Cause of death categories of marine mammals and marine turtles stranded in the UK (January 2000-December 2004)

Species	Cause of death Category	No.
Harbour porpoise	Physical Trauma (bottlenose dolphin attack)	128
	Bycatch*	128
	Starvation (including 26 neonates)	73
	Pneumonia, Parasitic	65
	Generalised Bacterial Infection	29
	Pneumonia, Parasitic and Bacterial	29
	Physical Trauma (including one boat/propeller strikes)	27
	Live Stranding	19
	Others	16
	Gastropathy &/or Enteropathy	12
	Dystocia & Stillborn	5
	Pneumonia, Parasitic and Mycotic	5
	Neoplasia	4
	(Meningo) Encephalitis	3
	Pneumonia, Bacterial	1
	Pneumonia, Bacterial and Mycotic	1
	Pneumonia, Unknown Aetiology	1
	Gas Embolism	1
	Not Established	51
Common dolphin	Bycatch	116
	Live Stranding	29
	Starvation (including 1 neonate)	7
	Others	5
	Physical Trauma (includes one propeller strike)	4
	Gas Embolism	3
	Pneumonia, Bacterial	2
	Gastropathy &/or Enteropathy	2
	Not Established	22
Striped dolphin	Live Stranding	9
	Bycatch	5
	(Meningo) Encephalitis	5
	Others	5
	Gastropathy &/or Enteropathy	2
	Starvation	3
	Not Established	4
Atlantic white-sided dolphin	Live Stranding	21
	Starvation	2
	(Meningo) Encephalitis	1
	Bycatch	1
	Generalised Bacterial Infection	1
	Pneumonia, Bacterial	1
White beaked dolphin	Live Stranding	7
	Starvation	2
	Dystocia & Stillborn	2
	(Meningo) Encephalitis	1
	Gastropathy &/or Enteropathy	1
	Physical Trauma	1
	Pneumonia, Parasitic	1
	Pneumonia, Parasitic and Bacterial	1

Species	Cause of death Category	No.
Bottlenose dolphin	Live Stranding	2
	Others	1
	Starvation	1
	Not Established	10
Minke whale	Entanglement	6
	Live Stranding	2
	Starvation	2
	Dystocia & Stillborn	1
	Not Established	1
Risso's dolphin	(Meningo) Encephalitis	3
	Starvation	2
	Dystocia & Stillborn	2
	Live Stranding	1
	Bycatch	1
	Gas Embolism	1
	Others	1
	Not Established	1
Sowerby's beaked whale	Live Stranding	6
	Gas Embolism	1
Pilot whale	Live Stranding	5
	Starvation	1
	Not Established	1
Sperm whale	Live Stranding	3
Fin whale	Live Stranding	1
	Physical Trauma (boat strike)	1
Northern bottlenose whale	Live Stranding	1
Pygmy sperm whale	Not Established	1
Humpback whale	Starvation	1
Killer whale	Starvation	1
Cuvier's beaked whale	Starvation	1
Unknown delphinid	Not Established	1
Loggerhead turtle	Starvation	3
Green turtle	Others	1
Leatherback turtle	Bycatch	1

* Of the 128 harbour porpoise bycatches reported here, 35 were carcasses retrieved directly from fishing vessels as part of observer-based research conducted by the Sea Mammal Research Unit. The post-mortem investigations of these carcasses were conducted in collaboration with the Sea Mammal Research Unit

2.5 Trends in selected causes of cetacean mortality (1990-2004)

Since the initiation of the UK Marine Mammal Strandings Project in 1990, over 2150 cetacean and 600 pinniped necropsies have been conducted on stranded carcasses in the UK. In this section trends in some of the main causes of death (bycatch in all species; harbour porpoises killed by bottlenose dolphins: infectious disease and starvation in harbour porpoises) are reported in spatial, temporal and biological contexts.

### Trends in bycatch in UK-stranded cetaceans (1990-2004)

Between September 1990 and December 2004 inclusive, entanglement in fishing gear (bycatch) was the most common cause of death in UK-stranded cetaceans subjected to detailed post-mortem examination accounting for 234/420 (55.7%) common dolphins, 225/1235 (18.2%) harbour porpoises, 6/84 (7.1%) striped dolphins, 5/27 (18.5%) Risso's dolphins, 5/60 (8.3%) white beaked dolphins, 3/58 (5.2%) Atlantic white-sided dolphins, 2/44 (4.6%) bottlenose dolphins, and 1/28 (3.6%) pilot whales. In addition, 75 by-caught harbour porpoises, one common dolphin, one bottlenose dolphin and one white beaked dolphin were retrieved directly from fishing vessels for post-mortem examination during the same period.

By-caught harbour porpoises frequently presented with thin, linear cutaneous cuts or depressions on the leading edges of the pectoral fins, tail flukes and dorsal fin, or had similar lesions that sometimes partly or completely circumscribed the head, consistent with the use of large-meshed monofilament gillnet-type fishing gear. In contrast, stranded common dolphins diagnosed as bycatches often lacked these gillnet-type external injuries. It should be noted, however, that many of the numerous common dolphin and harbour porpoise bycatches in south-west England (mainly Cornwall and Devon) strand in a state of "moderate decomposition", in which the skin on the beak, fins and flukes is often partly abraded or peeled-off, making detection of subtle net-induced lesions difficult.

The spatial distribution of UK-stranded harbour porpoise and common dolphin bycatches between 1990 and 2004 is shown in **Figure 10a-b**. The vast majority of common dolphin bycatches (n=222 or 94.9% of the total number of by-caught common dolphins diagnosed) occurred in south-west England (Cornwall, Devon and Dorset). Stranded harbour porpoise bycatches were also most frequently recorded in south-west England (n=90) comprising 40% of the total number of UK-stranded harbour porpoises diagnosed as bycatch. Harbour porpoise bycatches were also recorded in Wales (n=62), and along the North Sea coastline (Northumberland to Humberside) (n=45), comprising 27.6% and 20% of the total number of by-caught harbour porpoises diagnosed in the UK, respectively. In contrast, only 32 stranded cetaceans were diagnosed as bycatches in Scotland comprising 25 harbour porpoises, three Risso's dolphins, two Atlantic white-sided dolphins, one common dolphin and one bottlenose dolphin diagnosed between 1990 and 2004.

The annual number of common dolphin and harbour porpoise strandings diagnosed by-caught in Southwest England (Cornwall, Devon and Dorset) since 1990 are

shown in **Figure 11**. Although the number of stranded common dolphin (and harbour porpoise strandings) has increased annually since 1999, the proportion of by-caught common dolphins and harbour porpoises diagnosed in this region has considerable inter-annual variation since 1990 (**Figure 12**).   The seasonal distribution of stranded harbour porpoise and common dolphin bycatches in south-west England show a strong winter (December-March) bias (**Figure 13**) that is not reflected in the rest of the UK where common dolphin bycatches are rare and harbour porpoise bycatches have a more year-round distribution (data not shown). The age and sex structure of the harbour porpoise and common dolphin bycatches examined in the UK between 1990 and 2004 (**Figures 14-15**) demonstrates a fairly even sex distribution of both species, but a greater proportion of juvenile harbour porpoises and adult common dolphins were diagnosed as bycatches compared to other age classes.
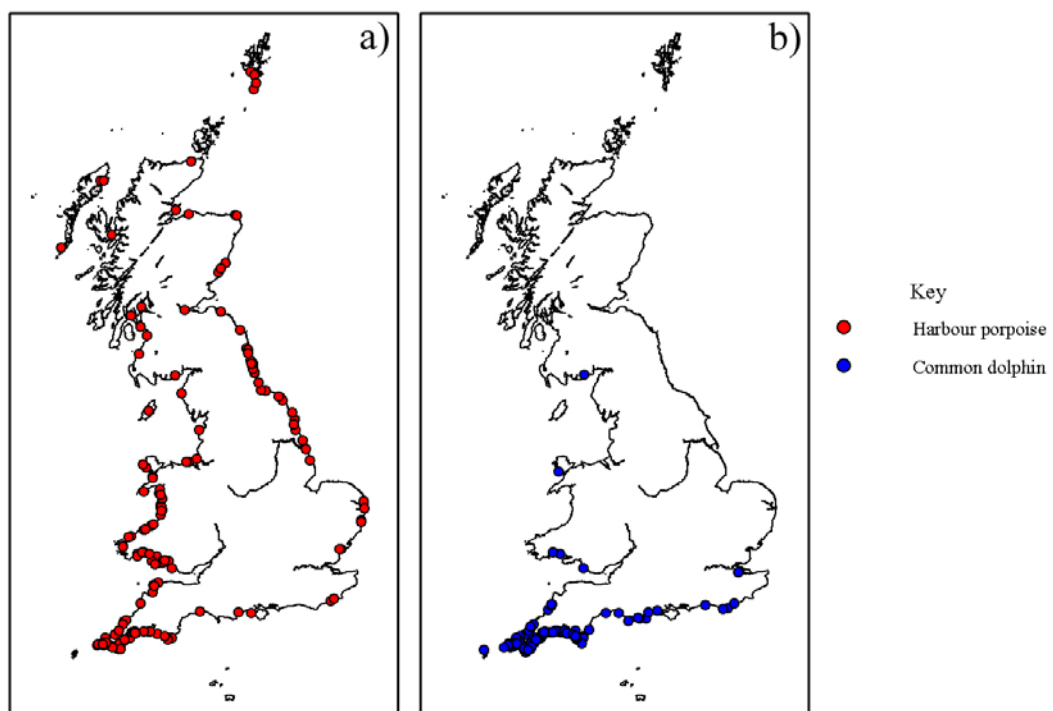
Figure 10 Spatial distribution of a) 225 harbour porpoises and b) 234 common dolphins by-caught between Sep. 1990 and Dec 2004
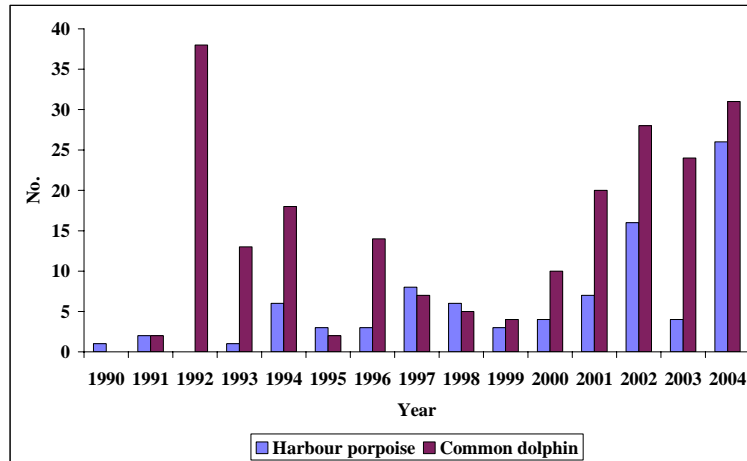
Figure 11 Annual numbers of stranded harbour porpoise and common dolphin diagnosed as bycatches in the SW of England from Sep 1990 to Dec 2004
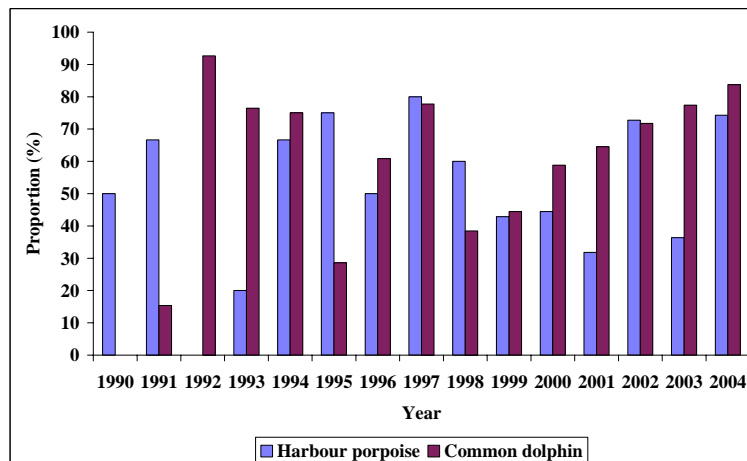
Figure 12 Annual proportion (number by-caught/ number examined at post-mortem) of stranded harbour porpoise and common dolphin bycatches in the SW of England from September 1990 to December 2004
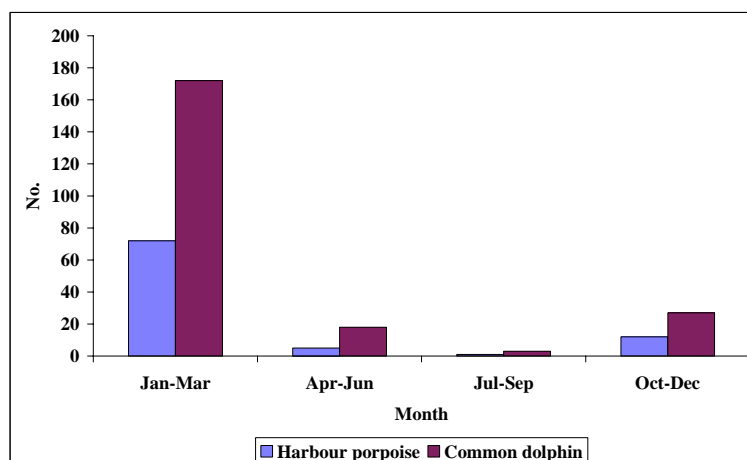
Figure 13 Seasonal distribution of stranded harbour porpoise and common dolphin bycatches in the SW of England from September 1990 to December 2004
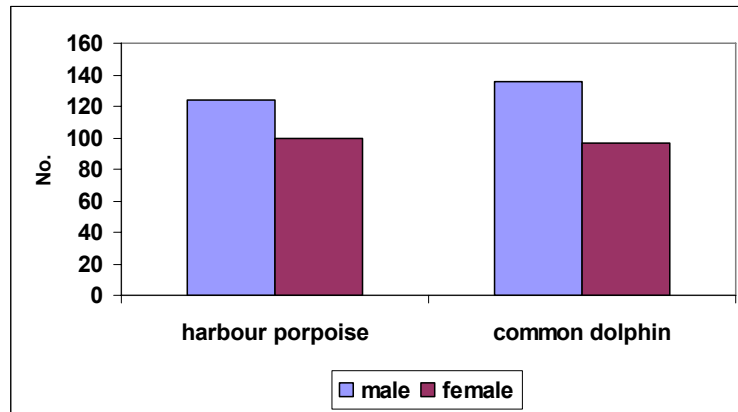
Figure 14 Frequency of male and female harbour porpoise and common dolphin bycatches stranded in the UK (September 1990 to December 2004)
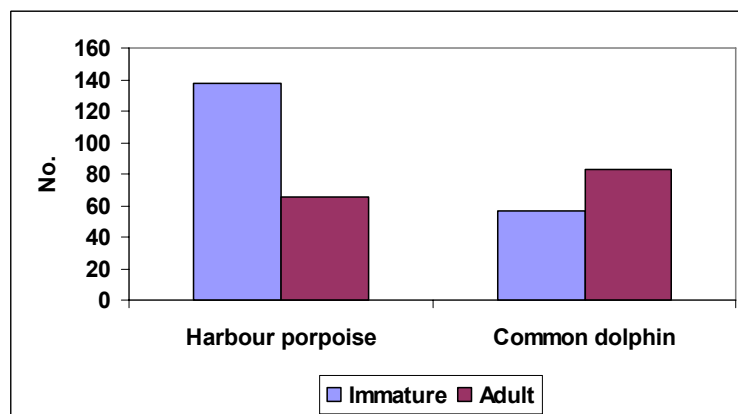
Figure 15 Sexual maturity of 203 harbour porpoise and 167 common dolphin bycatches stranded in the UK from September 1990 to December 2004

Trends in UK-stranded harbour porpoises fatally attacked by bottlenose dolphins (1990-2004)

Between 1990 and 2004 inclusive, 238 UK-stranded harbour porpoises were diagnosed as having probably been killed by bottlenose dolphins based on characteristic external and internal traumatic injuries (Ross & Wilson 1996, Jepson & Baker 1998). All cases of fatal attack on harbour porpoises by bottlenose dolphins occurred in north-east Scotland, west Wales or south-west England where porpoises share sympatric distributions with bottlenose dolphin populations (**Fig. 16a**). In Scotland the spatial distribution of stranded harbour porpoises fatally attacked by bottlenose dolphins has occurred with increasing frequency outside the Moray Firth area in recent years, consistent with a change in distribution of the resident Moray Firth bottlenose dolphin population (Wilson *et al.* 2004). The annual numbers of harbour porpoises diagnosed to have been killed by bottlenose dolphins in Wales has increased significantly since 1999 (**Figure 16c**), despite knowledge of pathological

criteria to diagnose these interactions since 1992/1993. In contrast, the annual numbers of harbour porpoises killed by bottlenose dolphins in Scotland has varied between years but does not show any particular trend (**Figure 16b**).

Most porpoises killed by bottlenose dolphins were juveniles (76.2%) compared to adults (20.8%) and neonates (3%) and this age structure was consistent between all regions where these fatal interactions occurred. There were 130 male and 107 female harbour porpoises killed by bottlenose dolphins during this period in the UK.

In Scotland, the number of harbour porpoises diagnosed to have died as bycatch has been relatively low compared to the number diagnosed to have died from fatal attack from bottlenose dolphins (**Figure 17a**). In contrast, the increasing number and proportion of harbour porpoises stranded in west Wales that were killed by bottlenose dolphins since 1999 has been inversely proportional to those diagnosed to have died of bycatch in the same period (**Figure 17b**).

A

B

C
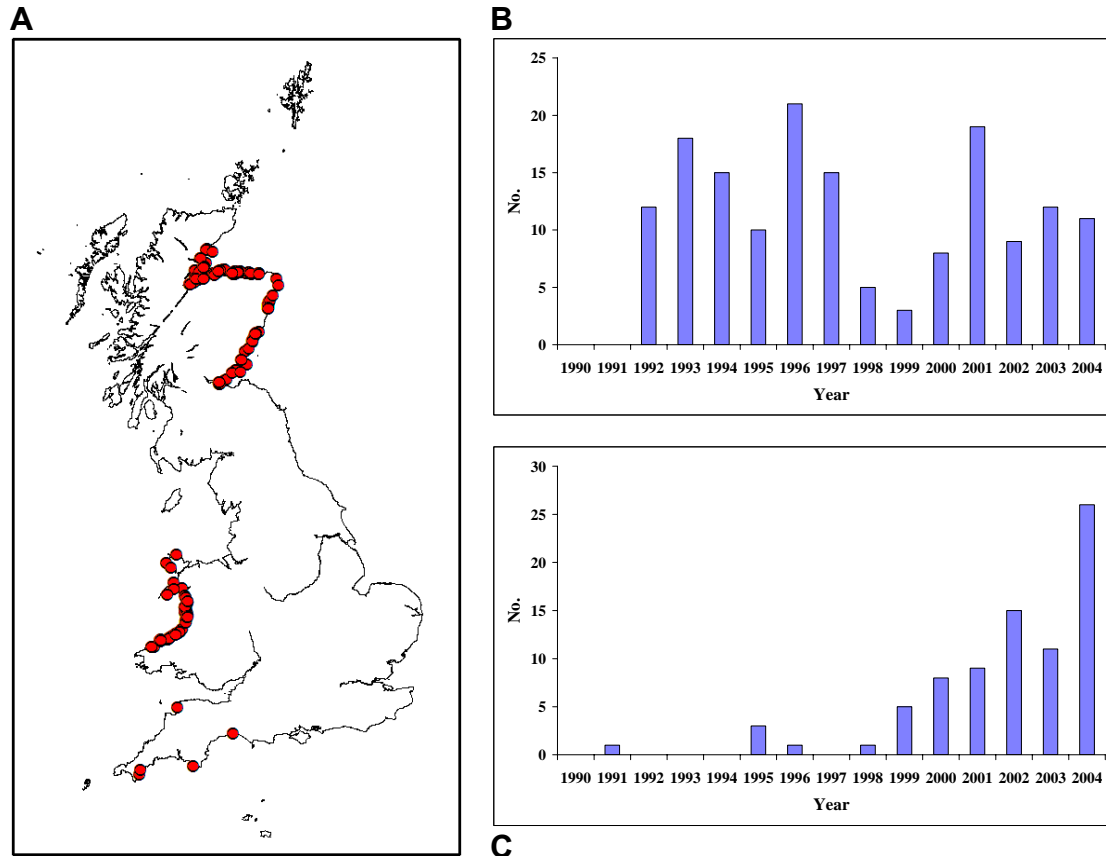
Figure 16 (A) Spatial distribution of 238 harbour porpoises killed by bottlenose dolphins between Sep 1990 and Dec 2004. **(B)** Annual numbers of stranded harbour porpoises killed by bottlenose dolphins in Scotland (Sept 1990 - Dec 2004). **(C)** Annual numbers of stranded harbour porpoise killed by bottlenose dolphins in England and Wales from Sep 1990 to Dec 2004
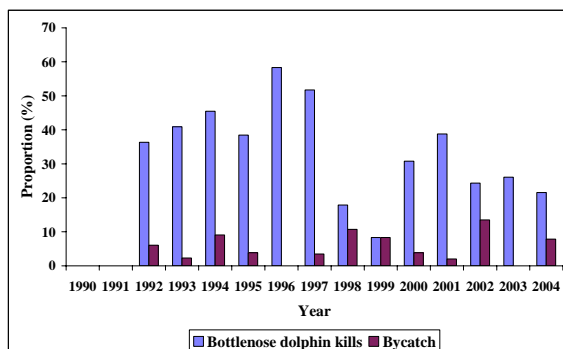
Figure 17a Annual proportion of stranded harbour porpoises diagnosed to be killed by bottlenose dolphins and by-caught in Scotland (as % of all post-mortem
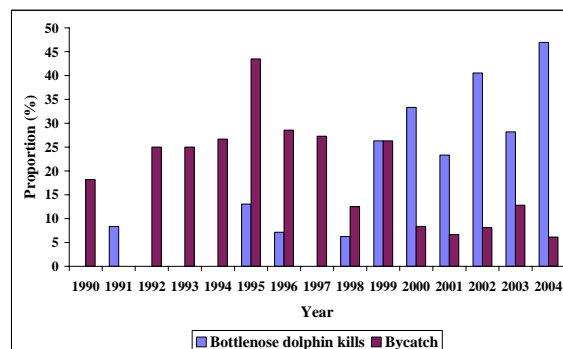
Figure 17b Annual proportion of stranded harbour porpoises diagnosed to be killed by bottlenose dolphins and by-caught in Wales (as % of all post-mortem

Trends in infectious disease mortality in UK-stranded harbour porpoises 1990-2004

A range of infectious diseases caused the death of 283 stranded harbour porpoises comprising 121 adults (68 male, 53 female), 116 juveniles (49 male, 67 female), 4 neonates (2 male, 2 female) and 42 (19 male, 23 female) unclassified. The age distribution of all fatal infectious diseases, showing a predominance of cases between 1 and 3 years old, is shown in **Figure 18**. Of the 283 fatal infectious disease cases, 127 stranded in Scotland, 59 stranded in the east coast of England, 70 stranded in Wales/north-west England, 21 in south-west England and 6 in the Channel (**Figure 19**). The annual numbers of fatal infectious disease cases are shown in **Figure 20**. A higher proportion of porpoises examined died of infectious diseases in the winter (October-March) (168/590 = 29%) than in the summer (April-September) (115/720 = 16%), with the highest monthly total of infectious disease cases occurring in January (n= 37) and February (n= 32).

The most common causes of infectious disease mortality were heavy lungworm infections and pneumonias due to combinations of parasitic, bacterial and/or mycotic infections (n=176) and generalised bacterial infections (n=54) (Kirkwood *et al.* 1997; Jepson *et al.* 2000; Jepson *et al.* 2003). Bacterial organisms were often microscopically visualised within lesions and were usually isolated in pure culture from a range of tissues and lesions. A range of agents were responsible including *Salmonella* sp., *Streptococcus canis*, *Photobacterium* (*Listonella*) *damsela*, *Escherichia coli*, *Erysipelothrix rhusiopathiae*, *Morganella morganii*, *Mycobacterium fortuitum*, *Cetobacterium ceti*, *Streptococcus phocae*, *Streptococcus* spp., *Vibrio parahaemolyticus* and coryneforms (Jepson *et al.* 2003). The *Salmonella* spp. were not typed in all cases, but monophasic group B *Salmonella* having the antigenic structure 4, 12: a: - (Foster *et al.* 1999) were identified in all cases where typing was conducted. Only one case of generalised cetacean morbillivirus infection was diagnosed in a UK-stranded harbour porpoise during this period (Kennedy *et al.* 1992).

Trends in mortality due to starvation in UK-stranded cetaceans (1990-2004)

Between September 1990 and December 2004, 194 UK-stranded cetaceans were ascribed starvation as the most probable cause of death. These comprised harbour porpoises (n=146, including 72 neonates), common dolphins (n=10), Atlantic white-sided dolphins (n=10), striped dolphins (n=7), white beaked dolphins (n=7), minke whales (n=4), killer whales (n=2), Risso's dolphins (n=2), bottlenose dolphin (n=1), Cuvier's beaked whale (n=1), pilot whale (n=1) and humpback whale (n=1). This particular cause of death was given to animals that were in poor/emaciated condition but that did not have any other disease conditions that would explain the loss of nutritional status. The annual percentage of all harbour porpoises examined at post-mortem that died of starvation is shown in **Figure 21.**

Figure 18 The age distribution of 241 harbour porpoises that died as a result of infectious disease between Sep 1990 and Dec 2004
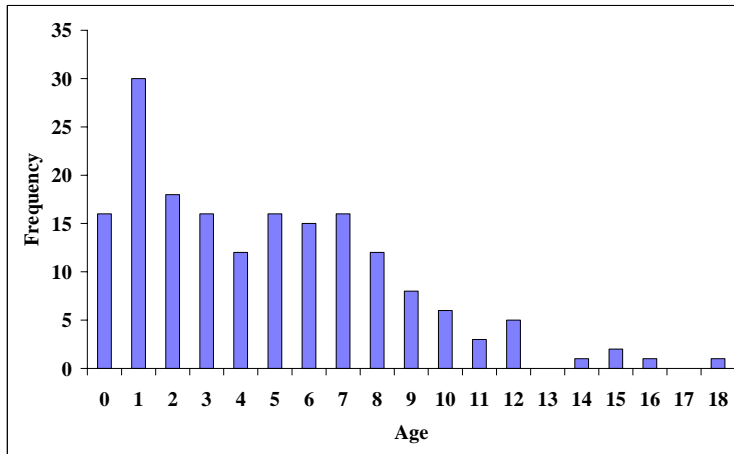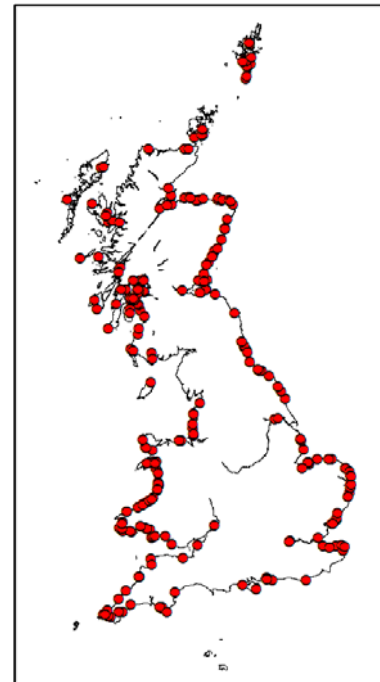

Figure 19 Spatial distribution of 283 harbour porpoises that died as a result of infectious disease between Sep 1990 and Dec
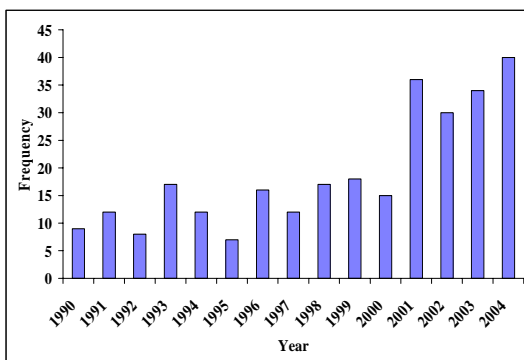
Figure 20 Annual number of UK-stranded harbour porpoises that died as a result of infectious disease between Sep 1990 and Dec 2004
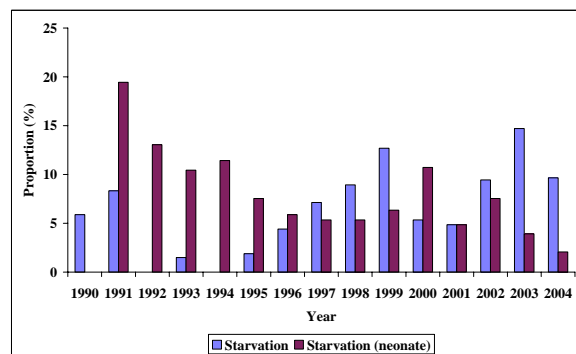
Figure 21 Annual proportion of juvenile/adult (blue) and neonate (purple) harbour porpoises that died of starvation (expressed as a percentage of all causes of death).

3.0 SPECIFIC RESEARCH ACTIVITY

3.1 Gas Embolism

During the period of this report, summaries of novel pathological findings from eight individually UK-stranded cetaceans between 1992 and 2003 that had acute and chronic gas embolic lesions, were published in conjunction with findings in ten beaked whales that mass-stranded in the Canary Islands in 2002 (Jepson *et al.* 2003; Fernandez *et al.* 2004). Detailed pathological descriptions of the eight UK-stranded cases, plus two additional UK-stranded cetacean cases (a Risso's dolphin and a common dolphin) were subsequently published in detail (Jepson *et al.* 2005b).   The detailed pathological findings from the ten beaked whales in the 2002 mass stranding in the Canary Islands have also been published (Fernandez *et al.* 2005). In the UK, a further three cases were identified in the UK in 2004 comprising a common dolphin in England, a harbour porpoise in Scotland, and a Sowerby's beaked whale in Wales. Therefore, a total of 13 confirmed UK-stranded cetaceans with acute or chronic gas embolism have been identified between 1992 and 2004 inclusive, comprising five common dolphins (*Delphinus delphis*), four Risso's dolphins (*Grampus griseus*), two harbour porpoises (*Phocoena phocoena*), a Blainville's beaked whale (*Mesoplodon densirostris*) and a Sowerby's beaked whale (*Mesoplodon bidens*). The spatial distribution of these UK-stranded cases is shown in **Figure 22**.   The number of marine mammal necropsies and relative frequency of cases of acute or chronic gas embolism (by taxonomic family) between September 1990 and December 2004 are shown in **Table 6**.

In addition to these confirmed gas embolism cases, three Sowerby's beaked whales that mass stranded alive at Eye, Western Isles on 15th June 1992 [SW1992/141A(1); SW1992/141A(2); SW1992/141A(3)] and a Sowerby's beaked whale [SW2001/201] that live-stranded at Praa Sands, Cornwall on 7th June 2002 had some lesions consistent with those described in three beaked whale species that mass-stranded in the Canary Islands, Spain in September 2002 (Fernandez *et al.* 2005). These four cases could be additional cases of gas embolism in cetaceans, although a more conservative diagnosis of "live stranding" was ascribed to them based on the histopathological appearance of the lesions.

A range of lesions were described in the 13 confirmed UK-stranded cetacean gas embolism cases. The most striking lesions were hepatic gas-filled cavitary lesions (0.2-6.0 cm diameter) involving approximately 5-90% of the liver volume in four (three adult, one juvenile) common dolphins (*Delphinus delphis*), four (two juvenile, two adult) Risso's dolphins (*Grampus griseus*), two (adult) harbour porpoises (*Phocoena phocoena*) and an adult Blainville's beaked whale (*Mesoplodon densirostris*) (**Figures 23(1-2 and 4)**). Two common dolphins also had multiple and bilateral gross renal cavities (2.0-9.0mm diameter) (**Figure 23(3)**) that, microscopically, were consistent with acute (n=2) and chronic (n=1) arterial gas emboli-induced renal infarcts. The Sowerby's beaked whale had extensive macroscopic bubbles in vessels within the intestinal serosa, the intestinal mesentery, the peri-renal venous sinuses and other vessels including the coronary vessels of the heart and meningeal vessels of the brain.

Histopathological examination of eight dolphin and one harbour porpoise cases with gross liver cavities revealed variable degrees of peri-cavitary fibrosis (**Figure 23(5)**),
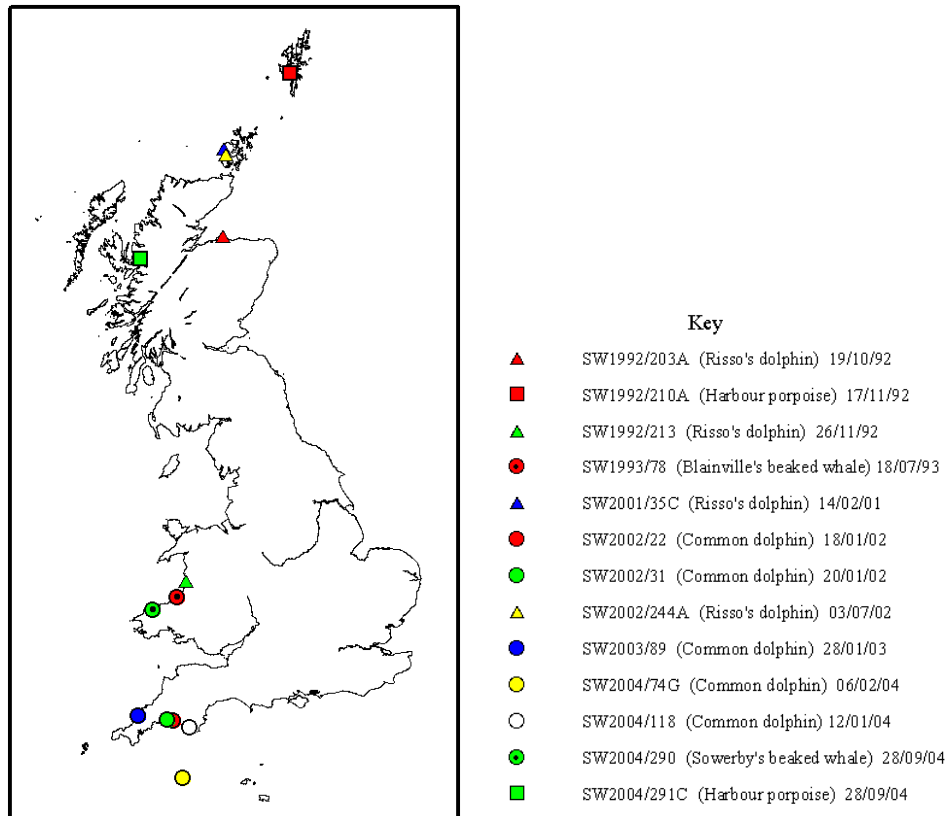
microscopic intra-hepatic spherical non-staining cavities (typically 50-750 μm in diameter) consistent with gas emboli within distended portal vessels and sinusoids and associated with hepatic tissue compression, haemorrhages, fibrin/organising thrombi and foci of acute hepatocellular necrosis (**Figure 23(6)**). Microscopically, the abnormal kidneys in two common dolphins had acutely infarcted and haemorrhagic lobules and one also had multiple clear cavities of variable size surrounded by mature pericavitary and interstitial fibrosis (confirmed by Gomori's trichrome stain) consistent with chronic gas embolic lesions (**Figure 23(7)**). Elastin stains suggested that, in contrast to the hepatic lesions, at least some of these renal gas emboli were intra-arterial (**Figure 23(8)**).

Microscopic bubble-like cavities were also found in other tissues including mesenteric lymph node (n=5), adrenal (n=2), spleen (n=2), pulmonary associated lymph node (n=1), posterior cervical lymph node (n=1) and thyroid (n=1). No bacterial organisms were isolated from 7/9 cavitated liver samples and the single cavitated kidney samples tested. Oil-Red-O stained frozen sections of formalin-fixed lung tissue were considered positive for intravascular fat emboli in a Risso's dolphin and a common dolphin and inconclusive in two other common dolphins tested. In two common dolphins with hepatic cavities, anti-cytokeratin antibody markers bound specifically to the biliary epithelium but did not stain the wall or lining of hepatic cavities in liver sections. Antibody markers for smooth muscle actin (SMA) demonstrated that a small proportion of intrahepatic gas bubbles resided within muscular blood vessels (possibly portal sphincters). In a common dolphin with intra-renal cavitation, some intrarenal non-staining bubble cavities resided within SMA-positive muscular blood vessels.

Although an increasing range of causes of gas emboli have been identified in human medicine (Peloponissios *et al.* 2003), the most likely cause in a cetacean may be that nitrogen bubbles and emboli (within venous, arterial, portal and lymphoid vessels) can develop *in vivo* due to expansion of pre-existing gas nuclei submitted to rapid decompression (Francis & Mitchell 2003). If bubble formation is extensive enough, the process can produce decompression sickness (DCS), even in humans undertaking repetitive breath-hold diving (Paulev 1965, Ferrigno & Lundgren 2003). The predominantly venous (including portal) distribution of the gas emboli in these cases is consistent with findings of DCS in humans (Francis & Mitchell 2003). In addition, fat emboli, found in the lungs in two of the four cases that had sections of lung stained for fat, have been associated with DCS including dysbaric osteonecrosis (Jones & Neuman 2003) and were reported in beaked whales that mass stranded with acute DCS-like lesions in the Canary Islands in 2002 (Jepson *et al.* 2003, Fernandez *et al.* 2004; Fernandez *et al.* 2005). It is therefore plausible that the pulmonary fat emboli, lymphoid gas bubbles, and (portal, sinusoidal and arterial) gas emboli and associated lesions in these cetacean cases could have a pathogenesis similar to DCS. Potential sources of bubbles include decompression-related embolism of intestinal gas and nitrogen bubble evolution from gas supersaturated tissues (see Jepson *et al.* 2005b). The latter process may occur via static means, as in typical human DCS, but alternative acoustically mediated mechanisms of bubble formation have also been proposed (e.g. acoustic activation) (Crum and Mao 1996, Crum *et al.* 2005).

Of the cetaceans examined in this study, there was a higher prevalence of bubble lesions in deep-diving species, such as Risso's dolphins and beaked whales (**Table**

6), which is consistent with modelled predictions of nitrogen tissue saturation and risk of nitrogen bubble formation by static or rectified diffusion (Houser *et al.* 2001). No evidence of gas bubble lesions has ever been described in pinnipeds and none were found in over 400 phocid seal necropsies in this study. Unlike cetaceans, most phocid seals studied are thought to dive on expiration (Falke *et al.* 1985; Ponganis *et al.* 2003) and complete lung collapse occurs at relatively shallower depths, such as 25-50 m in Weddell seals (Falke *et al.* 1985), resulting in reduced tissue nitrogen uptake compared to cetaceans. Further research is needed to elucidate the aetiology and pathogenesis of these novel gas and fat embolic lesions.

Key

▲	SW1992/203A (Risso's dolphin) 19/10/92
■	SW1992/210A (Harbour porpoise) 17/11/92
▲	SW1992/213 (Risso's dolphin) 26/11/92
◉	SW1993/78 (Blainville's beaked whale) 18/07/93
▲	SW2001/35C (Risso's dolphin) 14/02/01
●	SW2002/22 (Common dolphin) 18/01/02
●	SW2002/31 (Common dolphin) 20/01/02
△	SW2002/244A (Risso's dolphin) 03/07/02
●	SW2003/89 (Common dolphin) 28/01/03
○	SW2004/74G (Common dolphin) 06/02/04
○	SW2004/118 (Common dolphin) 12/01/04
◉	SW2004/290 (Sowerby's beaked whale) 28/09/04
■	SW2004/291C (Harbour porpoise) 28/09/04

Figure 22 Spatial distribution of 13 stranded cetaceans diagnosed with acute or chronic gas embolism between Sep 1990 and Dec 2004

Table 6: Number of UK-stranded marine mammal necropsies and number and relative frequency of cases with *in vivo* bubble disease (by taxonomic family) (Sept 1990 - December 2004)

Family	Number of animals necropsied	Number and relative frequency of cases with *in vivo* bubble disease
Phocoenidae	1310	2 (0.2%)
Delphinidae	701	9 (1.3%)*
Ziphidae	21	2 (9.5%)**
Balaenopteridae	37	0
Physeteridae	26	0
Kogiidae	4	0
Phocidae	685	0
TOTAL	**2784**	**13 (0.5%)**

* These comprise (cases/necropsies) 5/421 (1.2%) common dolphins (*Delphinus delphis*); 4/27 (15.0%) Risso's dolphins (*Grampus griseus*); 0/84 striped dolphins (*Stenella coeruleoalba*); 0/61 white beaked dolphins (*Lagenorhynchus albirostris*); 0/58 Atlantic white-sided dolphins (*Lagenorhynchus acutus*); 0/45 bottlenose dolphins (*Tursiops truncatus*); 0/28 pilot whales (*Globicephala melas*); 0/5 killer whales (*Orcinus orca*) and 0/1 unidentified dolphin.

** These comprise 1/1 Blainville's beaked whale (*Mesoplodon densirostris*), 1/16 (6.3%) Sowerby's beaked whales (*Mesoplodon bidens*), 0/3 northern bottlenose whales (*Hyperoodon ampullatus*), and 0/1Cuvier's beaked whales (*Ziphius cavirostris*).
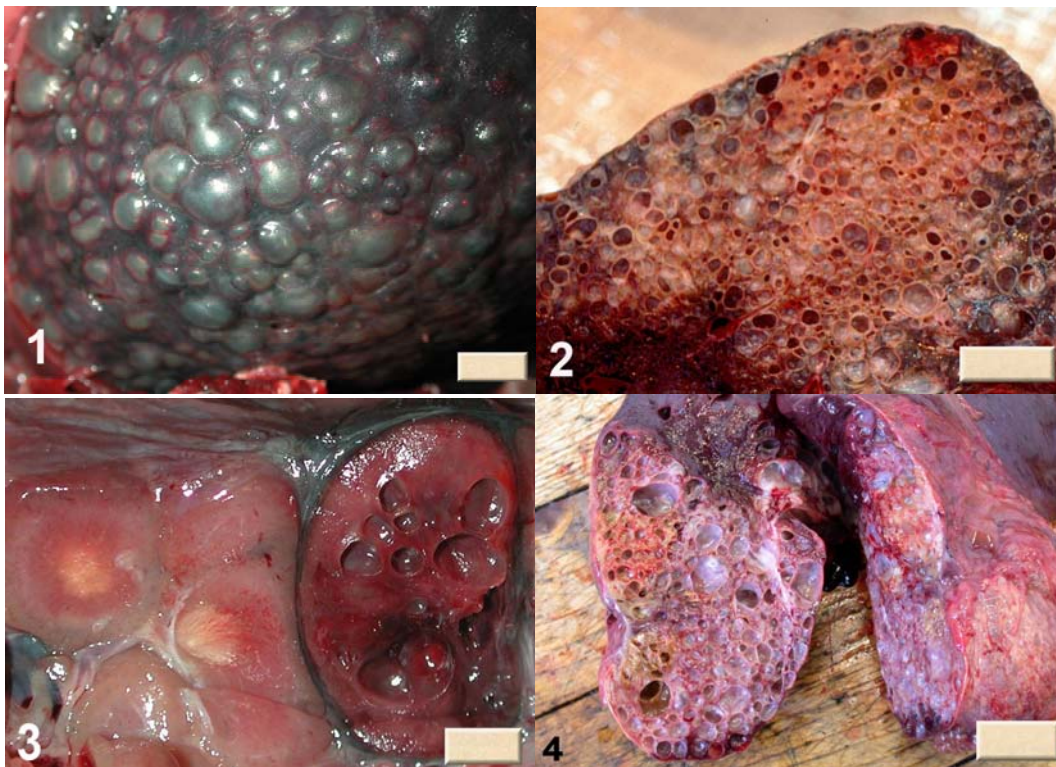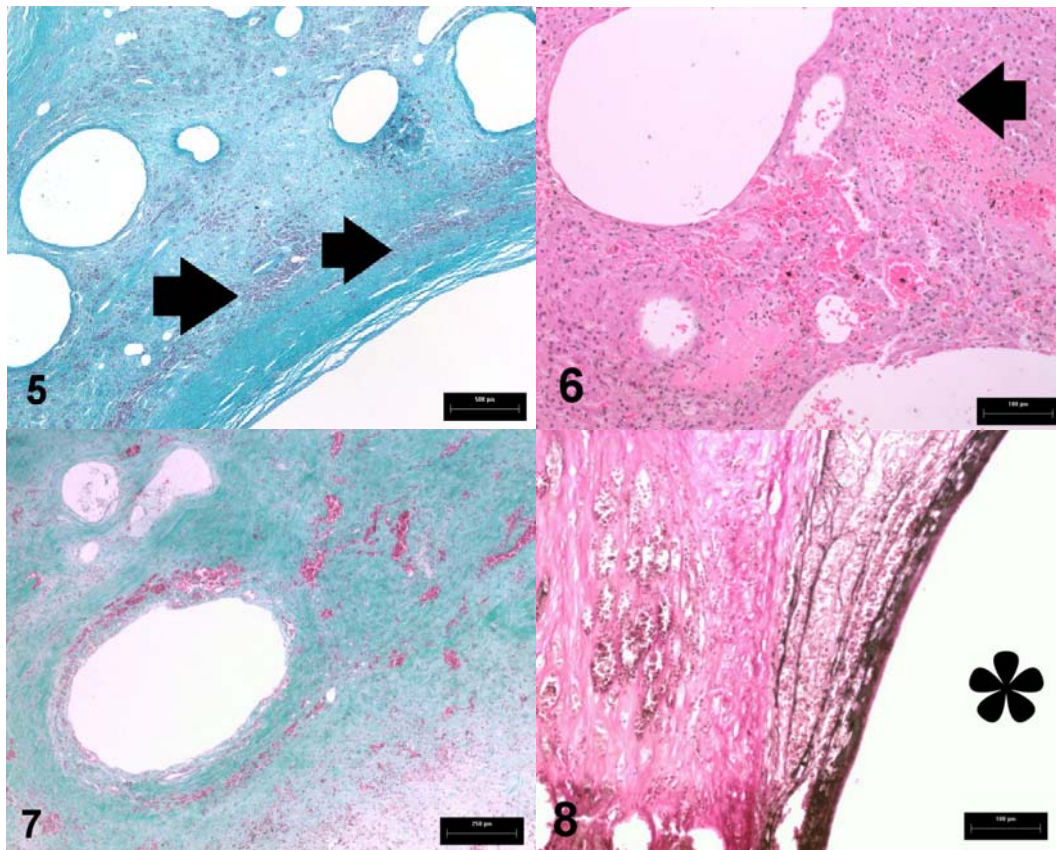
Figure 23 (1-8) Cetacean gas embolic lesions
Fig. 36(1) Liver (*D. delphis*): Surface of liver with multiple sub-capsular gas-filled cavitary lesions. Bar = 2cm. **Fig. 36(2)** Liver (*G. griseus*): Extensive intra-hepatic cavitary lesions. Bar = 4cm. **Fig. 36(3)** Kidney (*D. delphis*): Intra-renal cavitary lesions within dark red (infarcted) reniculus (right) adjacent to grossly normal reniculus (left). Bar = 6mm. **Fig. 36(4)** Liver (*D. delphis*): Extensive intra-hepatic cavitary lesions. Bar = 4cm. **Fig. 36(5)** Liver (*G. griseus*): Cavitary lesions and extensive hepatic/peri-cavitary fibrosis (blue-green stain). Note thin zones of preserved hepatocytes (arrows). Gomori's trichrome. Bar = 500μm. **Fig. 36(6)** Liver (*G. griseus*): Intra-hepatic bubble-like cavities associated with haemorrhage and hepatocellular necrosis (arrow). HE. Bar = 100μm. **Fig. 36(7)** Kidney (*D. delphis*). Renal peri-cavitary fibrosis. Fibrous tissue stained blue-green. Gomori's trichrome. Bar = 250μm. **Fig. 36(8)** Kidney (*D. delphis*): Positive elastin-staining (brown) of fragmented internal elastic lamina of gas-distended arterial blood vessel. The vessel lumen is marked (*). Elastic van Gieson. Bar = 100μm.

3.2 Relationships between environmental contaminant exposure and health status in UK-stranded harbour porpoises

a) Trends in Contaminant Exposure in UK-stranded Harbour Porpoises (1989-2001)

An analysis of biological, pathological and toxicological data on a number of UK-stranded harbour porpoises between 1989 and 2001 was completed in 2003 (Jepson 2003). The analyses included blubber levels of the sum of 25 chlorobiphenyl (CB) congeners (Σ25CBs) (n=340 porpoises), sum of *p, p'*-DDE, *p, p'*-TDE, *p, p'*-DDT (ΣDDTs) (n=169), dieldrin (n=169), α-hexachlorocyclohexane (α-HCH) (n=168), γ-

hexachlorocyclohexane (γ-HCH; Lindane) (n=169) and hexachlorobenzene (HCB) (n=222) and the sum of 15 brominated diphenyl ether congeners (Σ15BDEs) (n=62) were obtained together with hepatic levels of 11 trace elements (n=95-308) and the sum of three butyltin compounds (ΣBTs; tributyltin and its degradation products dibutyltin and monobutyltin)(n=132). The spatial and temporal distribution of the available toxicological data is summarised in **Table 7**. The regions were defined as: south-west England (Devon, Cornwall, Somerset and Gloucestershire); east coast of England (Northumberland to Essex); Wales and north-west England; the English Channel (Kent to Dorset); and Scotland.

RESULTS

The mean, median, minimum and maximum concentrations and sample size for each contaminant (mg kg^{-1} lipid wt) are given in **Table 8**. In single-variate statistical analyses, Scottish porpoises had significantly lower Σ25CBs (**Figure 24**) and ΣDDTs levels (**Figure 25**) than animals stranding in other regions. There was also statistically significant temporal variation (slight decline) in Σ25CBs levels between 1989 and 2001 (**Figure 26**) that persisted when Scottish porpoises were excluded (**Figure 27**). Blubber levels of ΣDDTs (**Figure 28**), dieldrin (**Figure 29**), α-HCH and HCB declined significantly between 1989 and 2001. Levels of Σ25CBs levels were also negatively correlated with nutritional status, although no significant correlations were found between nutritional status and dieldrin, α-HCH, γ-HCH and HCB levels.

Levels of Σ15BDEs were positively correlated with age and higher in porpoises from the East Coast of England and higher in porpoises stranding between July and September. Hepatic levels of ΣBTs were positively correlated with age and were highest in porpoises from the Channel and the North Sea and the lowest in porpoises from Wales/northwest England (**Figure 30**).

Levels of silver (Ag), arsenic (As), cadmium (Cd), iron (Fe), mercury (Hg), lead (Pb), selenium (Se) and molar Hg:Se ratio accumulated with age. Levels of copper (Cu) were negatively correlated with age and significantly higher in neonates compared to juveniles/adults (**Figure 31**). Levels of As were positively correlated with nutritional status and levels of Fe, nickel (Ni) and zinc (Zn) were negatively correlated with nutritional status. Levels of Ag, cromium (Cr), Ni and Pb were significantly higher in females compared to males of all ages (**Figure 32-35**). Porpoises in Scotland had the highest Cd levels and porpoises in Wales the lowest, although porpoises on the East coast of England/Channel also had higher levels than porpoises from the south-west England (**Figure 36**). Levels of Cr and Ni were significantly lower in Scotland than in other regions (**Figures 37-38**). Levels of Pb were significantly higher in Wales than all other regions (**Figure 39**). Levels of Ag increased significantly between 1991 and 1997 and then levelled-off between 1998 and 2001 (**Figure 40**). Levels of Cr, Ni and Pb declined significantly between 1989 and 2001 (**Figures 41-43**). No significant annual temporal variation was found for Cd, Cu, Hg, Se and Zn between 1989 and 2001, for As between 1991 and 2001 and for Fe between 1995 and 2001.

Levels of Σ25CBs were strongly and positively correlated with all organochlorine pesticides analysed: ΣDDTs, dieldrin, α-HCH, γ-HCH and HCB. Levels of Σ25CBs were also positively correlated with Hg and Σ15BDEs, but not with ΣBTs or Se.

Levels of Hg were strongly and positively correlated with Se (**Figure 44**), molar Hg:Se ratio (**Figure 45**) and ΣBTs. Levels of Σ15BDEs were positively correlated with all organochlorine pesticides analysed: ΣDDTs, dieldrin, α-HCH, γ-HCH and HCB. There was insufficient data to investigate potential correlations between Σ15BDEs and ΣBTs levels.

Table 7: Summary of aggregated spatial and temporal distribution of contaminant data in UK-stranded and by-caught harbour porpoises between 1989 and 2001[a]

Contaminant		SW Eng	E coast Eng	Wales/NW Eng	Channel	Scotland	Irish Sea
Σ25CBs	No.	34	134	109	7	53	3
	Period	1990-2001	1989-2001	1990-2001	1989-1998	1989-2000	1993
ΣDDTs	No.	19	19	75	60	8	none
	Period	1990-2000	1990-2000	1990-2000	1989-1998	1990-2000	
Dieldrin	No.	19	75	60	7	8	none
	Period	1990-2000	1990-2000	1990-2000	1989-1998	1990-2000	
α-HCH	No.	19	75	59	7	8	none
	Period	1990-2000	1990-2000	1990-2000	1989-1998	1990-2000	
γ-HCH	No.	19	75	60	7	8	none
	Period	1990-2000	1990-2000	1990-2000	1989-1998	1990-2000	
HCB	No.	19	85	83	7	28	none
	Period	1990-2000	1989-2000	1990-2000	1989-1998	1989-2000	
Σ15BDEs	No.	6	35	20	1	none	none
	Period	1996-1999	1996-1999	1996-1999	1998		
ΣBTs	No.	18	49	39	1	25	none
	Period	1994-2001	1992-2001	1994-2001	1994	1996-2000	
Ag	No.	29	89	76	3	27	3
	Period	1992-2001	1991-2001	1992-2001	1993-1998	1993-2000	1993
As	No.	29	89	75	3	27	3
	Period	1992-2001	1991-2001	1992-2001	1993-1998	1993-2000	1993
Cd	No.	34	120	112	7	32	3
	Period	1990-2001	1989-2001	1989-2001	1989-1998	1990-2000	1993
Cr	No.	34	120	112	7	32	3
	Period	1990-2001	1989-2001	1989-2001	1989-1998	1990-2000	1993
Cu	No.	34	119	112	7	32	3
	Period	1990-2001	1989-2001	1989-2001	1989-1998	1990-2000	1993
Fe	No.	21	66	51	1	26	none
	Period	1996-2001	1995-2001	1996-2001	1998	1995-2000	
Hg	No.	34	120	112	7	32	3
	Period	1990-2001	1989-2001	1989-2001	1989-1998	1990-2000	1993
Ni	No.	34	120	112	7	32	3
	Period	1990-2001	1989-2001	1989-2001	1989-1998	1990-2000	1993
Pb	No.	34	120	112	7	32	3
	Period	1990-2001	1989-2001	1989-2001	1989-1998	1990-2000	1993
Se	No.	33	119	90	7	32	3
	Period	1990-2001	1989-2001	1990-2001	1989-1998	1990-2000	1993
Zn	No.	34	120	112	7	32	3
	Period	1990-2001	1989-2001	1989-2001	1989-1998	1990-2000	1993

[a] Blubber concentrations of the sum of 25 individual CB congeners (Σ25CBs), total DDTs (ΣDDTs), dieldrin, α-HCH, γ-HCH, HCB and 15 BDE congeners (Σ15BDEs) were expressed as mg kg^{-1} lipid weight. Hepatic concentrations of total butyltins (ΣBTs) and Ag, As, Cd, Cr, Cu, Fe, Hg, Ni, Pb, Se, Zn were expressed as mg kg^{-1} wet weight. No. = number of individuals. Period = period for which data is available.

Table 8: Mean, median, minimum and maximum levels of different contaminants in UK-stranded harbour porpoises[a].

Contaminant	Mean		Median	Min	Max	N
Σ25CBs	19.9	11.9	0.13	159.7	340	
ΣDDTs	5.61	3.39	0.36	34.0	169	
Dieldrin	2.54	1.41	<0.01	16.9	169	

α-HCH	0.04	0.03	<0.01	0.35	168
γ-HCH	0.20	0.15	<0.01	1.67	169
HCB	0.34	0.30	<0.01	2.02	222
Σ15BDEs	2.39	1.71	0.08	7.70	62
ΣBTs	0.18	0.13	<0.01	1.20	132
Ag	1.46	0.75	0.01	23.0	227
As	0.60	0.49	0.06	6.80	226
Cd	0.21	0.13	0.01	2.80	308
Cr	0.48	0.41	0.04	4.40	308
Cu	15.9	9.40	2.50	138.0	307
Fe	331.8	326.0	65.0	1080.0	165
Hg	13.6	4.10	0.01	190.0	308
Ni	0.25	0.13	0.02	1.10	308
Pb	0.10	0.05	0.01	1.10	308
Se	7.06	3.00	0.39	106.0	284
Zn	56.9	43.0	5.50	217.0	308

[a] Blubber concentrations of the sum of 25 individual CB congeners (Σ25CBs), total DDTs (ΣDDTs), dieldrin, α-HCH, γ-HCH, HCB and 15 BDE congeners (Σ15BDEs) were expressed as mg kg^{-1} lipid weight. Hepatic concentrations of total butyltins (ΣBTs) and Ag, As, Cd, Cr, Cu, Fe, Hg, Ni, Pb, Se, Zn were expressed as mg kg^{-1} wet weight. N = number of individuals.

b) Relationships between PCB exposure and health status in UK-stranded porpoises

The main objective of this study was to use a case-controlled statistical analysis to test the hypothesis that chronic exposure to potentially immunosuppressive polychlorinated biphenyls (PCBs) may predispose harbour porpoises to mortality associated with infectious disease utilising a considerably larger dataset than has been examined previously (Jepson *et al*. 1999). Experimental data on aquatic mammals has recently been collated to derive dose-response relationships for the adverse health effects of PCB exposure (Kannan *et al*. 2000, Schwacke *et al*. 2002). These resulting dose-response relationships, based on experimental PCB-induced immunological and reproductive effects in mink, seals and otters have led to a proposed total PCBs threshold concentration for adverse health effects (including immunosuppression) in all marine mammals of 17 mg/kg PCBs lipid weight in blubber (Kannan *et al*. 2000). A standard conversion factor was therefore used to calculate total PCBs levels (based on the *Aroclor 1254* envelope) in porpoises in this study so that levels could be compared directly with this proposed threshold for adverse biological effects. This approach is designed to help determine whether any statistical associations between disease and PCB exposure identified occur at biologically relevant concentrations that are potentially indicative of adverse toxic effects. The detailed methods, results and conclusions of this study have been reported in the scientific literature (Jepson *et al*. 2005a) and are summarised here.

For figures 24 to 43, the bars indicate 1 S. E.
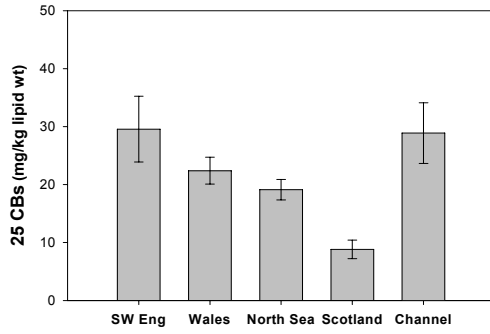
Figure 24 Spatial distribution of Σ25CBs levels.
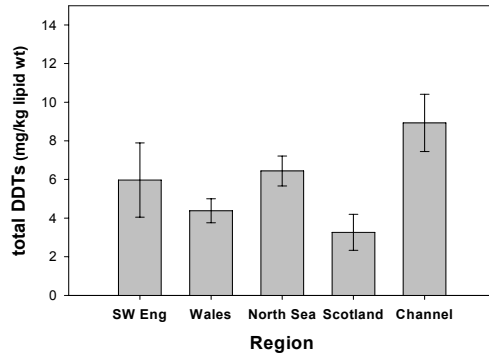
Figure 25 Spatial distribution of ΣDDTs levels.
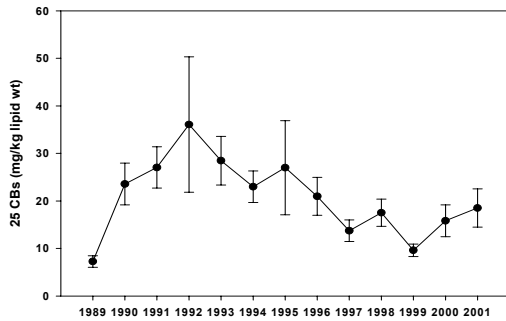
Figure 26 Temporal distribution of Σ25CBs levels 1989-2001 (all UK).
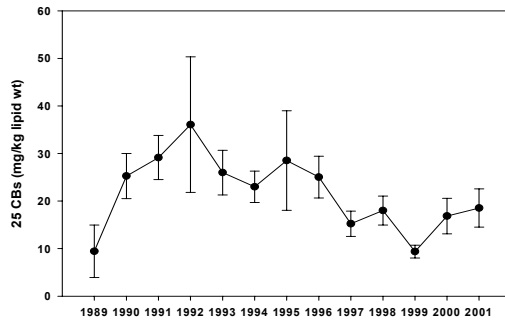
Figure 27 Temporal distribution of Σ25CBs levels in porpoises from England and Wales 1989-2001.
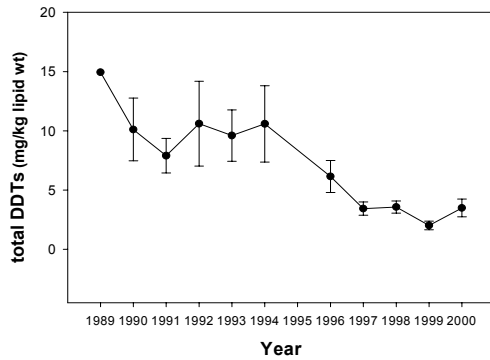
Figure 28 Temporal distribution of ΣDDTs levels in UK-stranded porpoises 1989-2000.
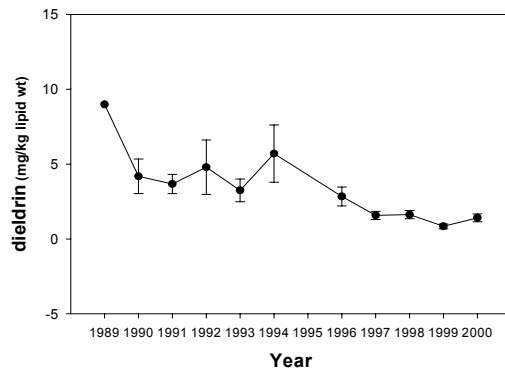
Figure 29 Temporal distribution of dieldrin levels in UK-stranded porpoises 1989-2000.

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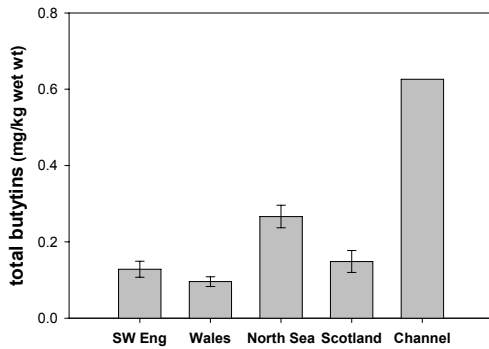
Figure 30 Spatial distribution of total butyltins.
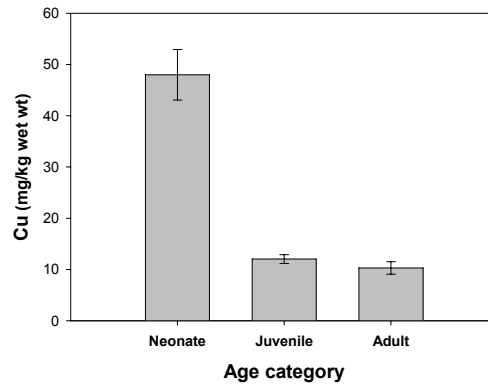
Figure 31 Hepatic concentration of Cu within different age categories.
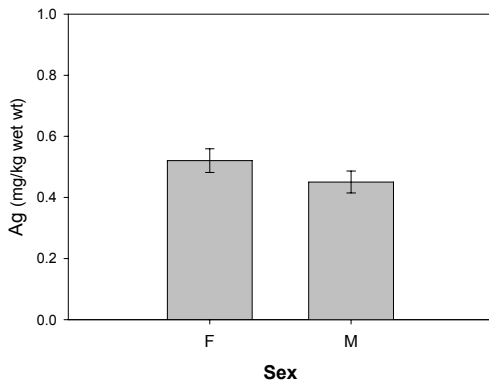
Figure 32 Hepatic concentration of Ag within female (n=87) and male (n=140) porpoises.
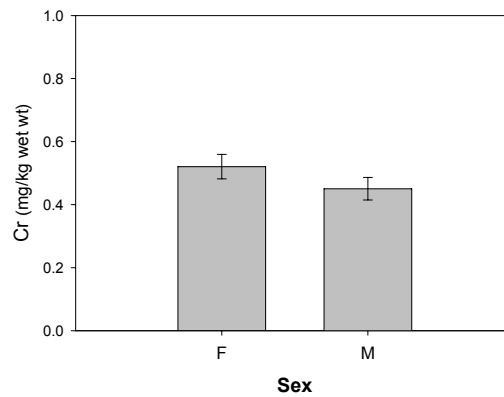
Figure 33 Hepatic concentration of Cr within female (n=129) and male (n=179) porpoises.
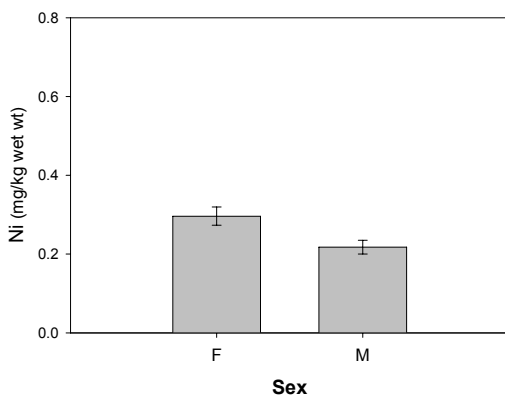
Figure 34 Hepatic concentration of Ni within female (n=129) and male (n=179) porpoises.
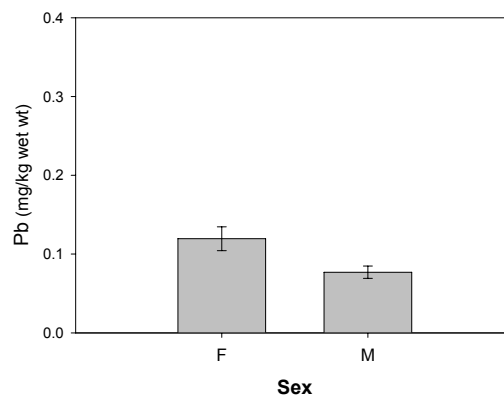
Figure 35 Hepatic concentration of Pb within female (n=129) and male (n=179) porpoises.
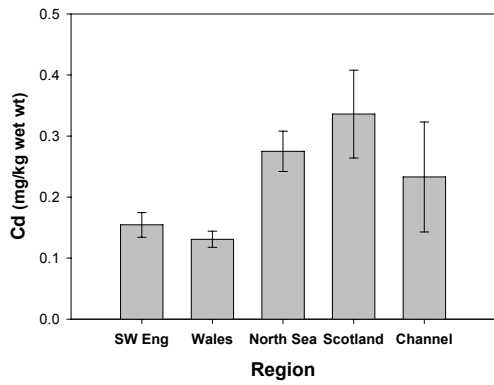
Figure 36 Regional variation in
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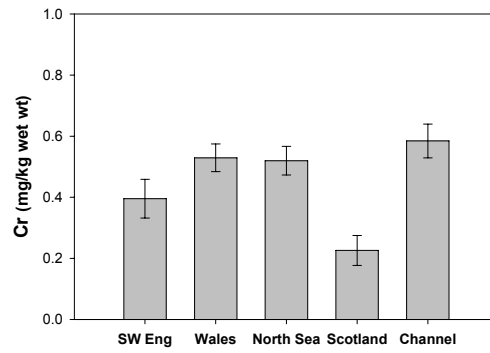
Figure 37 Regional variation in
hepatic Cr levels.
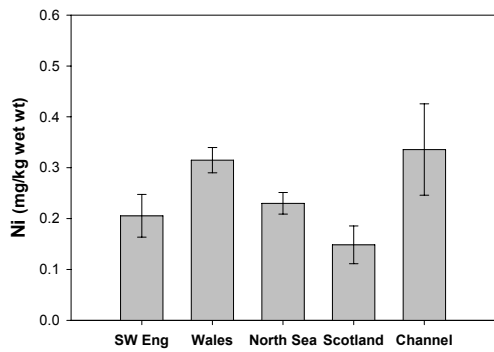
Figure 38 Regional variation in
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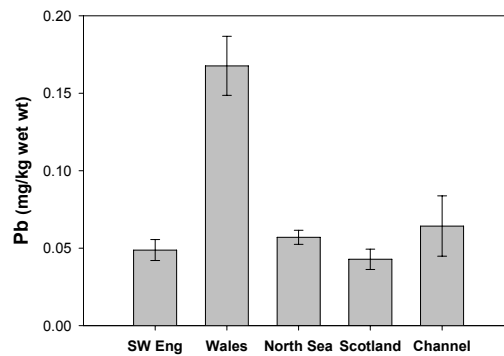
Figure 39 Regional variation in
hepatic Pb concentrations.
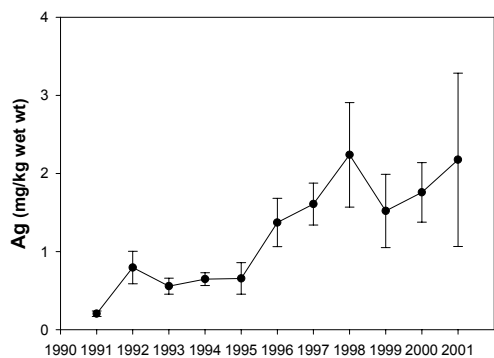
Figure 40 Annual variation in hepatic
Ag concentrations.
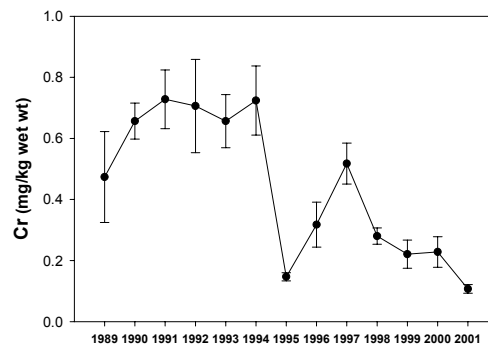
Figure 41 Annual variation in hepatic
Cr concentrations 1989-2001.
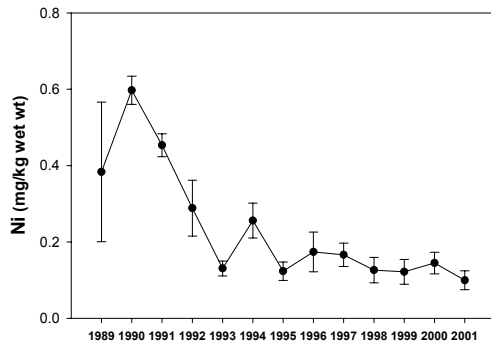
Figure 42 Annual variation in hepatic Ni concentrations 1989-2001.
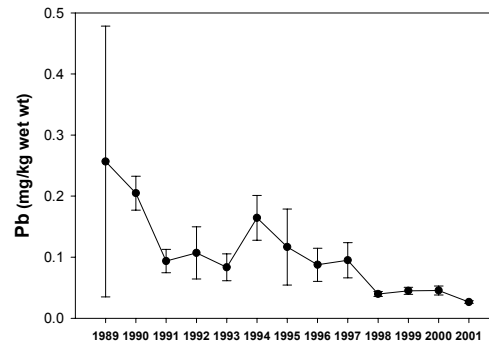
Figure 43 Annual variation in hepatic Pb concentrations 1989-2001.
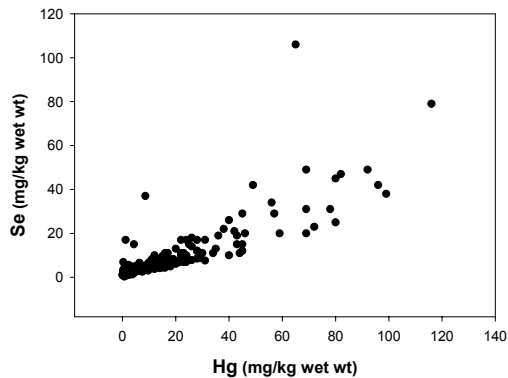
Figure 44 Plot of liver Se against Hg concentrations (n=284)
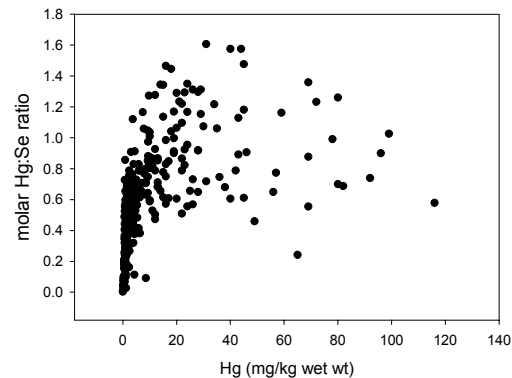
Figure 45 Plot of liver molar Hg:Se ratio against Hg concentrations (n=284)

METHODOLOGICAL APPROACH

Harbour porpoises for which Σ25CBs data were available (n=340) were divided into two groups based on the cause of death as determined by detailed post-mortem examination. Individuals diagnosed to have died from physical trauma, bycatch or dystocia were included in the physical trauma category (n=175). Animals diagnosed to have died from disease processes caused by one or more infectious agents (parasitic, bacterial, mycotic or viral) were included in the infectious disease category (n=82). Animals were excluded from the analyses if the cause of death could not be established or was attributed to other causes (n=84).

Analysis of variance and regression models (Sokal & Rohlf 1995, SYSTAT 1998) were used to test for associations between Σ25CBs (mg kg^{-1} lipid) and the following parameters: cause of death category (acute physical trauma v infectious disease), age, sex, nutritional status, region of stranding, year of stranding, state of decomposition and method of carcass storage (whether frozen or not). The regions

were defined as: south-west England (Devon, Cornwall, Somerset and Gloucestershire); east coast of England (Northumberland to Essex); Wales and north-west England; the English Channel (Kent to Dorset); and Scotland. Associations between nutritional status and cause of death category were also tested statistically.

RESULTS

There were no significant differences in age ($F_{1,240}$= 3.57, p>0.05), age categories (neonates, juveniles, adults) (X^2_2 = 4.40, p> 0.10, n= 247) or percentage of hexane extractable lipid in the blubber samples for Σ25CBs analyses ($F_{1,255}$= 0.45, p>0.50) between the acute physical trauma and infectious disease groups. The infectious disease group, however, had significantly poorer nutritional status (relative body-weight) ($F_{1,253}$ = 74.1, p<0.001) and mean blubber thickness ($F_{1,251}$ = 43.7, p<0.001) than the physical trauma group.

The infectious disease group had significantly greater Σ25CBs concentrations (mean = 27.6 mg kg^{-1} lipid) than the physical trauma group (mean = 13.6 mg kg^{-1} lipid) (p<0.001) (**Table 9a**) (**Figure 46**). Females had significantly lower Σ25CBs levels than males (**Table 9a**) and, within both the physical trauma and infectious disease groups, adult females had lower Σ25CBs levels than adult males or juveniles (**Figures 47a and 47b**). There was significant spatial variation in Σ25CBs levels, with Scottish porpoises having significantly lower levels than animals stranding in other regions (**Table 9a**). There was also statistically significant temporal variation in Σ25CBs levels between 1989 and 2001 demonstrating a gradual decline from the early 1990s to 2001 (**Table 9a**). Finally, Σ25CBs levels were negatively correlated with nutritional status (relative body-weight) and mean blubber thickness (**Table 9b**). No associations were found between Σ25CBs concentrations and the following variables: age, Season 2, state of decomposition and method of carcass storage (**Table 9a**).

A multivariate analysis of variance using all available data, with Σ25CBs as the dependent variable and cause of death category and sex as independent variables, demonstrated that both cause of death category ($F_{1,255}$ = 34.9, p<0.001) and sex ($F_{1,255}$ = 16.3, p< 0.01) remained significantly and independently associated with Σ25CBs (overall model: Multiple R = 0.39, n=257, p<0.001). A multivariate analysis of variance using all available data, with Σ25CBs as the dependent variable and cause of death category and region found as independent variables, demonstrated that both cause of death category ($F_{1,252}$ = 4.20, p<0.05) and region found ($F_{4,249}$ = 3.48, p< 0.01) remained significantly and independently associated with Σ25CBs (overall model: Multiple R = 0.46, n=254, p<0.001).

In contrast, analysis of covariance with Σ25CBs as the dependent variable, cause of death category as an independent factor and nutritional status (relative body-weight) as an independent covariate, demonstrated that only the cause of death category remained significantly associated with variation in Σ25CBs (higher Σ25CBs in the infectious disease group) (**Table 10a**). A similar analysis of covariance with Σ25CBs as the dependent variable, cause of death category as an independent factor and nutritional status (mean blubber thickness) as an independent covariate, also

demonstrated that only the cause of death category remained significantly associated with variation in Σ25CBs (higher Σ25CBs in the infectious disease group) (**Table 10b**). Collectively, these multivariate analyses demonstrate a significant independent association between elevated Σ25CBs levels and infectious disease mortality that is not confounded by sex, season, region or either index of nutritional status (relative bodyweight or mean blubber thickness).

Total blubber PCBs levels (expressed on a PCB formulation basis as Aroclor 1254[®]) were calculated [total PCB concentration = 3.0 x sum of ICES7 congeners (mg/kg lipid)] enabling direct comparison with a proposed threshold concentration for adverse health effects (including immunosuppression) in marine mammals of 17mg/kg lipid weight (Kannan *et al.* 2000). Within animals with total PCBs levels above 17 mg/kg lipid (*n*=154), the associations between total PCBs levels and cause of death category (**Figure 48a**) (**Table 11a**), and between total PCBs levels and nutritional status (relative body-weight) and mean blubber thickness were highly significant (**Table 11b**), but there was no significant relationship between total PCBs levels and either sex (**Table 11a**) or age (**Table 11b**) and no significant mean age difference between the physical trauma and infectious disease groups ($F_{1,142}$= 0.01, *p*>0.95). Among porpoises with total PCBs above 17 mg/kg lipid, 67 males and 25 females died of physical trauma and 30 males and 32 females died of infectious disease.

In the subset of animals with total PCBs below the proposed threshold level of 17 mg/kg lipid for adverse health effects in marine mammals (Kannan *et al.* 2000) (*n*=103), no significant difference in mean total PCBs levels was found either between the physical trauma and infectious disease groups (**Figure 48b**) (**Table 12a**), or with either index of nutritional status (relative body-weight or mean blubber thickness) (**Table 12b**). Within the group of porpoises with total PCBs below 17 mg/kg lipid, there was no significant relationship between total PCBs levels and either age or sex (**Table 12a and 12b**) but porpoises dying of infectious disease had a significantly greater mean age than those dying of physical trauma ($F_{1,96}$= 3.99, *p*=0.05). Among porpoises with total PCBs below 17 mg/kg lipid, 41 males and 42 females died of physical trauma and 8 males and 12 females died of infectious disease.

Nutritional status (relative body-weight) was significantly lower in the infectious disease group (compared to the physical trauma group) within the subgroups of animals with total PCBs below 17 mg/kg lipid ($F_{1,101}$= 42.6, *p*<0.001) and above 17 mg/kg lipid ($F_{1,150}$= 30.3, *p*<0.001). Mean blubber thickness was also significantly lower in the infectious disease group (compared to the physical trauma group) within the subgroups of animals with total PCBs below 17 mg/kg lipid ($F_{1,100}$= 41.5, *p*<0.001) and above 17 mg/kg lipid ($F_{1,149}$= 12.6, *p*=0.001). Within all porpoises that died of physical trauma, nutritional status (relative body-weight) ($F_{1,171}$= 4.89, *p*<0.05) and mean blubber thickness ($F_{1,171}$= 7.50, *p*<0.01) were significantly greater in porpoises with total PCBs below 17 mg/kg lipid than those with total PCBs above this threshold. In contrast, there was no significant difference in either nutritional status (relative body-weight) ($F_{1,80}$= 0.87, *p*>0.35) or mean blubber thickness ($F_{1,78}$= 1.11, *p*>0.25) between porpoises with total PCBs above and below 17 mg/kg lipid that died of infectious disease.
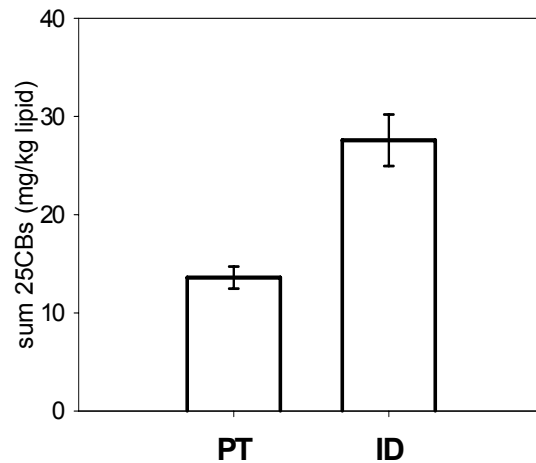
Figure 46 The 25 summed CB congeners are significantly higher in the infectious disease group (ID) (n=82) than the physical trauma group (PT) (n=175).The bars represent 1 S E
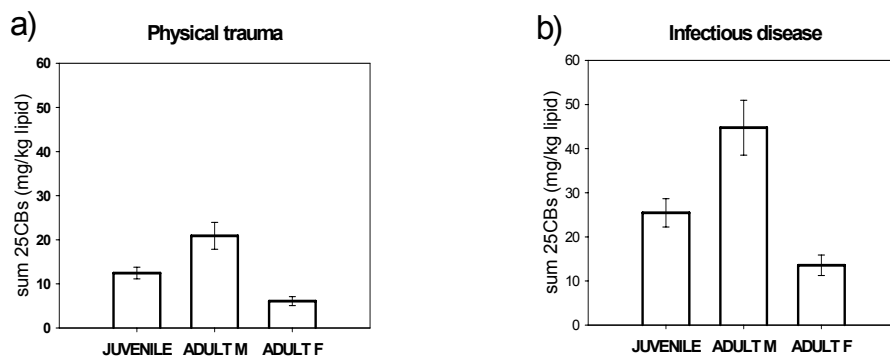
Figure 47 (a) Σ25CBs levels in sexually immature (juvenile) (n=107), adult males (n=40) and adult females (n=19) that died of physical trauma. **(b)** Σ25CBs levels in sexually immature (juveniles) (n=43), adult males (n=18) and adult females (n=20) that died of infectious disease. The bars represent 1 SE
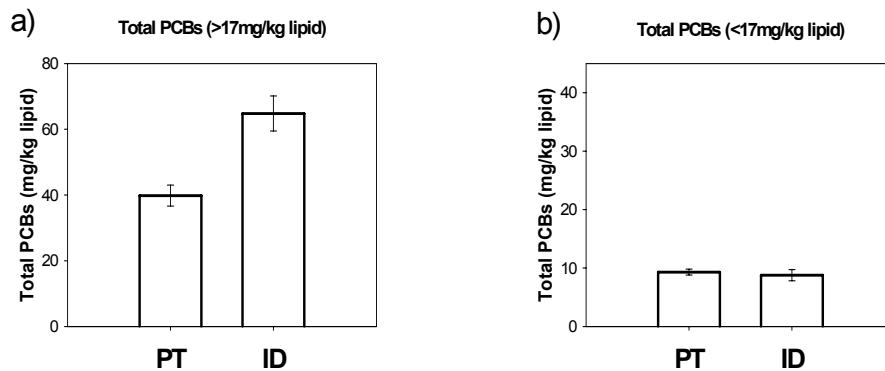
Figure 48 (a) Total PCB levels are significantly greater in the infectious disease group (ID) (n=62) compared to the physical trauma group (PT) (n=92) within individuals with total PCB levels >17mg/kg lipid wt. **(b)** Total PCB levels do not significantly differ between the physical trauma (PT) (n=83) and infectious disease groups (ID) (n=20) in individuals with total PCB levels <17mg/kg lipid wt. The bars represent 1 SE

Table 9a: Associations between the sum of 25 chlorobiphenyl congener concentrations and the factors shown[a].

Independent variable	n	F-ratio	Probability (p)
Cause of death	257	$F_{1, 255} =$ 27.5	*<0.001*
Sex	257	$F_{1, 255} =$ 9.41	<0.01
Year of death	255	$F_{12, 242} =$ 3.31	*<0.001*
Region	257	$F_{5, 251} =$ 9.78	*<0.001*
Season	252	$F_{1, 250} =$ 0.22	>0.60
Method of carcass storage	257	$F_{1, 255} =$ 2.72	>0.05
State of decomposition	238	$F_{3, 234} =$ 1.16	>0.30

[a] The natural logarithm of Σ25CBs is the dependent variable in each model. The associations between the categorical variables - cause of death, sex, season, region, year of death and method of storage - and the ln of Σ25CBs were tested using one-way analysis of variance (SYSTAT 1998).

Table 9b: Results of linear regression analysis between the sum of 25 chlorobiphenyl congener concentrations and the independent variables shown[a].

Independent variable Probability (p)	n	b (SE)	F-ratio
Age >0.75	242	0.03 (0.10)	$F_{1, 240} =$ 0.10
Nutritional status			
relative body-weight *<0.001*	255	-1.66 (0.42)	$F_{1, 253} =$ 15.4
mean blubber thickness *<0.001*	253	-0.64 (0.18)	$F_{1, 251} =$ 13.2

[a] The natural logarithm of Σ25CBs is the dependent variable in each model. The continuously distributed variables, Σ25CBs, age and mean blubber thickness, were ln transformed prior to analysis. The associations between these variables were tested using least-squares regression, as implemented by SYSTAT (1998), but only the sample size, regression coefficient (standard error), F-ratio statistic and its probability are shown. *, **, *** denote statistical significance at the p = 0.05, 0.01 and 0.001 levels, respectively.

Table 10 Results of analyses of covariance, each model with one factor and one covariate, to test whether variation in the sum of 25 chlorobiphenyl congener concentrations, the dependent variable in the model, is associated with the cause of death category and two indices of nutritional status: (a) relative body-weight, and (b) mean blubber thickness.

Independent variable	F-ratio	p
a)		
Cause of death	$F_{1,253}$= 14.9	*<0.001*
Nutritional status (*relative body-weight*)	$F_{1,253}$= 2.91	>0.05
Overall model: Multiple R: 0.33	*n*=255	*p*<0.001
b)		
Cause of death	$F_{1,251}$= 18.6	*<0.001*
Mean blubber thickness	$F_{1,251}$= 3.27	>0.05
Overall model: Multiple R: 0.34	*n*=253	*p*<0.001

Table 11

(a) Associations between blubber total PCBs concentration (as *Aroclor 1254*) and the factors shown[a] for porpoises with total PCBs levels > 17 mg/kg lipid.

Independent variable Probability (*p*)	*n*	F-ratio	
Cause of death	154	$F_{1, 152}$ = 24.8	*<0.001*
Sex	154	$F_{1, 152}$ = 3.11	>0.05

(b) Results of linear regression analysis between total PCBs concentration (as *Aroclor 1254*) and the independent variables shown[a] for porpoises with total PCBs levels > 17 mg/kg lipid.

Independent variable Probability (*p*)	*n*	b (SE)	F-ratio	
Age	144	0.12 (0.07)	$F_{1, 142}$ = 3.59	>0.05
Nutritional status				
relative body-weight	152	-1.24 (0.30)	$F_{1, 150}$ = 16.5	*<0.001*
mean blubber thickness	151	-0.46 (0.12)	$F_{1, 149}$ = 14.7	*<0.001*

[a] The natural logarithm of Σ25CBs is the dependent variable in each model. The associations between the categorical variables - cause of death and sex - and ln[Σ25CBs] were tested using one-way analysis of variance (SYSTAT 1998). The continuously distributed variables, Σ25CBs, age and mean blubber thickness were ln transformed prior to analysis. The associations between these variables were tested using least-squares regression, as implemented by SYSTAT (1998), but only the sample size, regression coefficient (standard error), F-ratio statistic and its probability are shown. *, **, *** denote statistical significance at the *p* = 0.05, 0.01 and 0.001 levels, respectively.

Table 12

(a) Associations between blubber total PCBs concentration (as *Aroclor 1254*) and the factors shown[a] for porpoises with total PCBs levels < 17 mg/kg lipid.

Independent variable	n	F-ratio	Probability (p)
Cause of death	103	$F_{1, 101} = 0.12$	>0.70
Sex	103	$F_{1, 101} = 1.60$	>0.20

(b) Results of linear regression analysis between total PCBs concentration and the independent variables shown[a] for porpoises with total PCBs levels < 17 mg/kg lipid.

Independent variable	n	b (SE)	F-ratio	Probability (p)
Age	98	-0.08 (0.13)	$F_{1, 96} = 0.38$	>0.50
Nutritional status				
relative body-weight	103	0.25 (0.56)	$F_{1, 101} = 0.20$	>0.65
mean blubber thickness	102	0.12 (0.26)	$F_{1, 100} = 0.22$	>0.60

[a] The natural logarithm of [total PCBs] is the dependent variable in each model. The associations between the categorical variables - cause of death and sex - and ln[total PCBs] were tested using one-way analysis of variance (SYSTAT 1998). The continuously distributed variables, total PCBs, age and mean blubber thickness were ln transformed prior to analysis. The associations between these variables were tested using least-squares regression, as implemented by SYSTAT (1998), but only the sample size, regression coefficient (standard error), F-ratio statistic and its probability are shown.

c) Relationships between trace metals and health status in UK-stranded porpoises

The objective of this study was to use case-control statistical analyses to test the hypothesis that chronic exposure to heavy metals may predispose harbour porpoises to mortality associated with infectious disease utilising a larger dataset than has been examined previously (Bennett *et al.* 2001).

METHODOLOGICAL APPROACH

Metals data were available for 165-308 individual harbour porpoises depending on the specific metal investigated (**Tables 7 and 8**). Porpoises diagnosed to have died from acute physical trauma, bycatch or dystocia were included in the physical trauma category (n=95 to 151) and those that died from disease processes caused by one or more infectious agents (parasitic, bacterial, mycotic or viral) were included in the infectious disease category (n=39 to 79). Animals were excluded from the analyses if the cause of death could not be established or was attributed to other causes (n=78).

One-way analysis of variance was used to test whether individual liver metal concentrations (mg kg^{-1} wet weight) differed between the physical trauma and infectious disease categories. To investigate the potential relationship between Hg and Se (Se-dependent detoxification), the molar Hg:Se ratio was calculated and tested for association with disease status. For those metals (and molar Hg:Se) that were associated with disease status, analysis of variance and regression models

(Sokal & Rohlf 1995, SYSTAT 1998) were used to test for associations between the hepatic metal concentrations (or molar Hg:Se ratio) and a range of potentially confounding variables including age, sex, nutritional status, region of stranding, year of stranding, state of decomposition and method of carcass storage (whether frozen or not). The regions were defined as: south-west England (Devon, Cornwall, Somerset and Gloucestershire); east coast of England (Northumberland to Essex); Wales and north-west England; the English Channel (Kent to Dorset); and Scotland.

If multiple associations occurred in single variate analyses, multivariate analysis of variance or forward stepwise multiple regression models were calculated separately for the metals associated with disease status (Sokal & Rohlf 1995, SYSTAT 1998) in order to attempt to identify the independent effects of each predictor. A forward stepwise procedure was used to help prevent the introduction of uninformative variables into each regression model. The criteria used for a variable to enter the model was that the inclusion of the variable in question would improve the overall variance ratio of the model (i.e. explained variance/residual variance) by a factor significant at the 10% level.

Where metal levels and the molar Hg:Se ratio were significantly higher in the infectious disease group, associations between nutritional status and cause of death category were tested using these statistical methods. For logistical reasons complete data were not available for all variables (e.g. age) and thus sample sizes for the statistical tests differ. Analyses of subsets of these porpoises were also examined where deemed appropriate to further investigate any statistically significant associations. The residuals of ln(Hg) v ln(age), and of molar Hg:Se ratio v ln(age), were calculated from the deviations from the least squares regression line between the two variables in each regression.

RESULTS

Liver concentrations of Ag, Hg, Se, Zn, Cr and Pb, and the molar Hg:Se ratio, were significantly greater in the infectious disease category than the physical trauma (**Figures 49 and 50**) (**Tables 13, 14 and 15**). Hepatic concentration of Cd, Fe and Ni did not differ significantly between the diseased and healthy groups (**Table 13**) and concentrations of Cu and As were significantly greater in the physical trauma group (**Table 13**). Therefore, levels of As, Cd, Cu, Fe and Ni were not examined further. The results of the statistical analyses to establish whether variation in hepatic concentrations of Ag, Hg, Se, Zn, Cr, Pb and the molar Hg:Se ratio were significantly associated with variables other than disease status are shown in **Tables 14 and 15**.

Hepatic levels of Hg, Se and molar Hg:Se ratio were significantly and positively correlated with age (**Figures 51 and 52**) (**Tables 13, 14 and 15**) and levels of Hg and Se were also significantly and positively inter-correlated (**Figure 53**). Levels of Hg and molar Hg:Se ratio were negatively correlated with nutritional status (**Tables 14 and 15**) although the relationship between Hg level and nutritional status was no longer significant when the effect of a single outlier (hepatic Hg concentration = 190mg/kg) was excluded ($F_{1,225} = 3.59$, $p > 0.05$). The relationship between elevated hepatic Hg levels in porpoises dying of infectious disease (compared to those dying

of physical trauma) remained highly significant when the same outlier was excluded ($F_{1,227} = 17.1$, $p < 0.001$).

A one-way analysis of variance using the residuals of the best fit regression line between ln[Hg] and ln[age] (for all physical trauma and infectious disease cases) as the dependent variable, demonstrated that the relationship between Hg and disease status was statistically significant once the effect of age-related bioaccumulation of Hg was controlled for. Similarly, a one-way analysis of variance using the residuals of the best fit regression between molar Hg:Se ratio and ln[age] (for all physical trauma and infectious disease cases) as the dependent variables, demonstrated that the relationship between molar Hg:Se ratio and disease status remained statistically significant once the effects of age-related variation in the molar Hg:Se ratio were controlled for.

Forward stepwise multiple regression analyses demonstrated that hepatic levels of Hg and Se, and hepatic the molar Hg:Se ratio, were significantly and independently correlated with age and disease status (**Table 16**). Forward stepwise multiple regression analyses demonstrated that hepatic levels of Se were significantly and independently correlated with disease status and age (**Table 16**).

Zinc levels were significantly higher in the infectious disease group (**Figure 54**) and were negatively correlated with nutritional status (**Tables 13, 14 and 15**). Forward stepwise multiple regression analyses demonstrated significant independent correlations between hepatic Zn concentrations and both disease status and nutritional status (**Table 16**). For the associations between elevated levels of Hg, Se, Zn, Pb and molar Hg:Se ratio in porpoises dying of infectious diseases, one-way analysis of variance with nutritional status as the dependent variable demonstrated that nutritional status ($F_{1,226} = 66.1$, $p < 0.001$) was significantly lower within the infectious disease group compared to the physical trauma group.

Table 13:   Mean tissue concentrations of metals in porpoise livers grouped according to whether the animals died from infectious disease or acute physical trauma [a]

	Infectious disease			Physical trauma				
Metal	Mean	± S.E.	n	Mean	± S.E.	n	F-ratio	P
Hg	18.8	3.00	79	11.3	1.50	151	$F_{1,228}=$ 18.5	p<0.001***
Se	9.55	1.75	70	6.29	0.87	142	$F_{1,210}=$ 10.0	p<0.01**
Zn	79.3	4.84	79	44.3	1.75	151	$F_{1,228}=$ 71.6	p<0.001***
Cr	0.51	0.06	79	0.41	0.04	151	$F_{1,228}=$ 4.61	p<0.05*
Fe	321.7	23.1	39	338.1	16.1	95	$F_{1,132}=$ 0.44	p>0.50
Cu	10.2	0.71	79	14.4	1.36	151	$F_{1,228}=$ 4.04	p<0.05*
Ni	0.27	0.03	79	0.23	0.02	151	$F_{1,228}=$ 1.27	p>0.25
As	0.51	0.05	53	0.69	0.06	122	$F_{1,173}=$ 7.56	p<0.01**
Ag	2.55	0.57	53	1.21	0.15	122	$F_{1,173}=$ 4.12	p<0.05*
Cd	0.22	0.03	79	0.21	0.02	151	$F_{1,228}=$ 0.31	p>0.55
Pb	0.11	0.02	79	0.09	0.01	151	$F_{1,228}=$ 4.72	p<0.05*

[a] The results of one-way analysis of variance (ANOVA) to test for differences between the two disease groups are shown (SYSTAT 1998). The ANOVAs were performed on logarithmically

transformed metal concentrations. All concentrations are expressed in mg/kg wet weight and are from liver tissue. S.E. refers to the standard errors. n is the number of individuals. *F*-ratio is the value of the *F* statistic resulting from the ANOVA to test for a significant difference between the two disease groups.   P is the probability associated with *F* test. *, **, *** denote statistical significance at the *P* = 0.05, 0.01 and 0.001 levels, respectively.

There were strong spatio-temporal dynamics to hepatic Pb and Cr levels that confounded their associations with disease status and other independent variables (**Tables 13, 14 and 15**). Across all data, multivariate analysis of variance using Pb as the dependent variable and combinations of disease status, region, year found, age, sex, state of decomposition and nutritional status (relative body-weight; mean blubber thickness) as independent variables, demonstrated that only the year of stranding ($p < 0.001$) and region of stranding ($p < 0.001$) remained significantly associated with variation in Pb levels when the other variables were held constant. Similarly, a multivariate analysis of variance conducted using Cr as the dependent variable and combinations of disease status, region, year found, sex, state of decomposition as independent variables demonstrated that only year of stranding ($p < 0.001$) and region of stranding ($p < 0.001$) remained significantly associated with variation in Cr levels when the other variables were held constant.

Hepatic Ag levels were significantly elevated in infectious disease group and were also positively correlated with age, higher in females compared to males and demonstrated significant spatial and temporal (inter-annual) variation. Across all data, a multivariate analysis of variance was conducted using Ag as the dependent variable in the model and combinations of disease status, region, year found, age and sex as independent variables. These analyses demonstrated that only age ($p < 0.001$) and the year found ($p < 0.01$) remained significantly associated with hepatic Ag levels when the effects of the other independent variables were controlled for.

Table 14: Results of the single-variate analyses for associations between variation in liver concentrations of Hg, Se, Zn and molar Hg:Se ratio and the factors shown[a]

Factor	Log(Hg)		Log(Se)		Log(Zn)		Hg:Se molar ratio	
	F-ratio	p	F-ratio	p	F-ratio	p	F-ratio	p
Cause of death	$F_{1, 228}=$ 18.5	<0.001***	$F_{1, 210}=$ 10.0	<0.01**	$F_{1, 228}=$ 71.6	<0.001***	$F_{1, 210}=$ 14.2	<0.001***
Age	$F_{1, 202}=$128.1	<0.001***	$F_{1, 187}=$150.1	<0.001***	$F_{1, 202}=$ 2.27	>0.10	$F_{1, 187}=$ 48.1	<0.001***
Nutritional status	$F_{1, 226}=$ 4.77	<0.05*	$F_{1, 209}=$ 1.24	>0.25	$F_{1, 226}=$ 36.9	<0.001***	$F_{1, 209}=$ 4.02	<0.05*
Sex	$F_{1, 228}=$ 2.60	>0.10	$F_{1, 210}=$ 0.94	>0.30	$F_{1, 228}=$ 0.48	>0.45	$F_{1, 210}=$ 0.02	>0.90
Region	$F_{5, 224}=$ 1.08	>0.35	$F_{5, 206}=$ 1.09	>0.35	$F_{5, 224}=$ 1.06	>0.35	$F_{5, 206}=$ 0.52	>0.75
Year of death	$F_{12, 215}=$ 0.79	>0.65	$F_{11, 198}=$ 0.74	>0.70	$F_{12, 215}=$ 0.70	>0.75	$F_{11, 198}=$1.80	>0.05*
State of decomposition	$F_{3, 225}=$ 0.16	>0.90	$F_{3, 207}=$ 0.36	>0.75	$F_{3, 225}=$ 0.10	>0.95	$F_{4, 207}=$ 0.24	>0.85
Method of storage	$F_{1, 228}=$ 2.33	>0.10	$F_{1, 210}=$ 1.36	>0.20	$F_{1, 228}=$ 0.10	>0.75	$F_{1, 210}=$ 2.79	>0.05*

[a] Hg, Se, Zn and molar Hg:Se are the dependent variables in the model. Factor refers to the independent variable in each test. The associations between the continuously distributed variables: log(age) and nutritional status were tested using least-squares regression, as implemented by SYSTAT (1998), but only the *F*-ratio statistic and its probability are shown. The associations between the categorical variables – cause of death category, sex, season 2, season 4, season 6, region, year of death, state of decomposition and method of storage - and Hg, Se, Zn (each log transformed) and the Hg:Se molar ratio were tested using one-way analysis of variance (SYSTAT 1998) *, **, *** denote statistical significance at the *P* = 0.05, 0.01 and 0.001 levels, respectively.

Table 15: Results of the single-variate analyses for associations between variation in liver concentrations of Ag, Cr and Pb and the factors shown[a]

Factor	Log(Ag) F-ratio	p	Log(Cr) F-ratio	p	Log(Pb) F-ratio	p
Cause of death	$F_{1, 173}$= 4.12	<0.05*	$F_{1, 228}$= 4.61	<0.05*	$F_{1, 228}$= 4.72	<0.05*
Age	$F_{1, 150}$= 14.8	<0.001***	$F_{1, 202}$= 0.16	>0.65	$F_{1, 202}$= 3.70	>0.05*
Nutritional status	$F_{1, 173}$= 0.78	>0.35	$F_{1, 226}$= 2.88	>0.05	$F_{1, 226}$= 3.77	>0.05*
Sex	$F_{1, 173}$= 4.68	<0.05*	$F_{1, 228}$= 6.72	=0.01**	$F_{1, 228}$= 4.73	<0.05*
Region	$F_{5, 169}$= 3.10	<0.05*	$F_{5, 224}$= 4.92	<0.001***	$F_{5, 224}$= 8.91	<0.001***
Year of death	$F_{9, 164}$= 2.81	<0.01**	$F_{12, 215}$=15.7	<0.001***	$F_{12, 215}$=8.51	<0.001***
State of decomposition	$F_{3, 171}$= 1.88	>0.10	$F_{3, 225}$= 9.52	<0.001***	$F_{3, 225}$= 4.00	<0.01**
Method of storage	$F_{1, 173}$= 1.56	>0.20	$F_{1, 228}$= 4.27	<0.05*	$F_{1, 228}$= 3.64	>0.05*

[a] Ag, Cr and Pb are the dependent variables in the model. Factor refers to the independent variable in each test. The associations between the continuously distributed variables: log(age) and nutritional status were tested using least-squares regression, as implemented by SYSTAT (1998), but only the *F*-ratio statistic and its probability are shown. The associations between the categorical variables – cause of death category, sex, season 2, season 4, season 6, region, year of death, state of decomposition and method of storage - and Ag, Cr and Pb (each log transformed) were tested using one-way analysis of variance (SYSTAT 1998) *, **, *** denote statistical significance at the *P* = 0.05, 0.01 and 0.001 levels, respectively.

Table 16 Results of the forward stepwise multiple regression analyses[a]

Dependent variable	Independent predictors	Partial R	*F*	*P*
Hg	Cause of death	0.63	$F_{1, 200}$= 12.1	=0.001***
	Age	1.24	$F_{1, 200}$=117.0	<0.001***
Se	Cause of death	0.25	$F_{1, 186}$= 4.13	<0.05*
	Age	0.91	$F_{1, 186}$=140.5	<0.001***
Hg:Se ratio	Cause of death	0.13	$F_{1, 186}$= 7.34	<0.01**
	Age	0.21	$F_{1, 186}$= 44.1	<0.001***
Zn	Cause of death	0.40	$F_{1, 200}$= 26.9	<0.001***
	Nutritional status	-0.59	$F_{1, 200}$= 7.92	<0.01**

[a] The multiple regression model implemented in SYSTAT (1998) was used. Cause of death category, age and nutritional status were the candidate independent variables in each model. Only the significant independent predictors are shown. Hg, Se, Zn and age were log transformed. F-ratio is the value of the F statistic when the other tested independent variables were held constant, *P* is the associated probability. *, **, *** denote statistical significance at the *P* = 0.05, 0.01 and 0.001 levels, respectively.
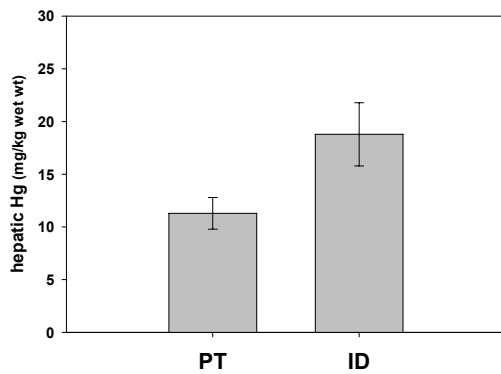
Figure 49 The mean liver Hg concentration (mg/kg wet wt) was significantly greater in the infectious disease (ID) group (n=79) compared to the physical trauma (PT) group (n=151). Bars = 1 S.E.
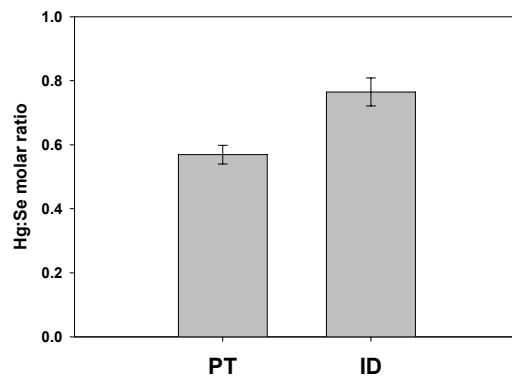
Figure 50 The mean hepatic Hg:Se molar ratio was significantly greater in the infectious disease (ID) group (n=70) compared to the physical trauma (PT) group (n=142). Bars = 1 S.E.

$$Hg = 2.90(Age) + 2.91$$
$$(r = 0.49, n = 204, F_{1,202} = 62.9, p < 0.001)$$
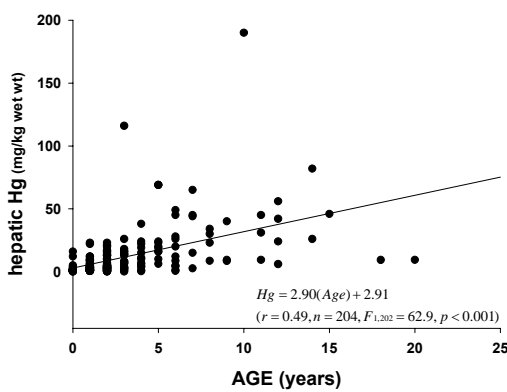
Figure 51 Hepatic Hg concentrations (mg/kg wet wt) plotted against age.
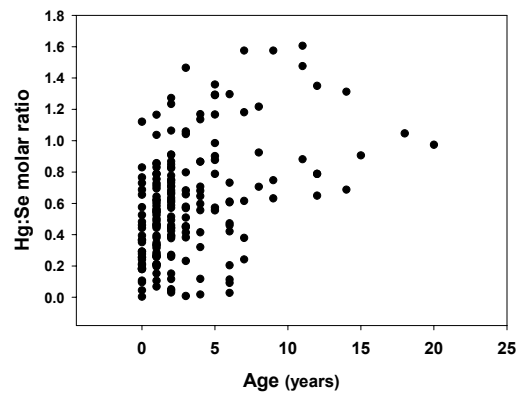
Figure 52 The hepatic Hg:Se molar ratio plotted against age.

$$Se = 0.53(Hg) + 0.51$$
$$(r = 0.83, n = 212, F_{1,210} = 451.8, p < 0.001)$$
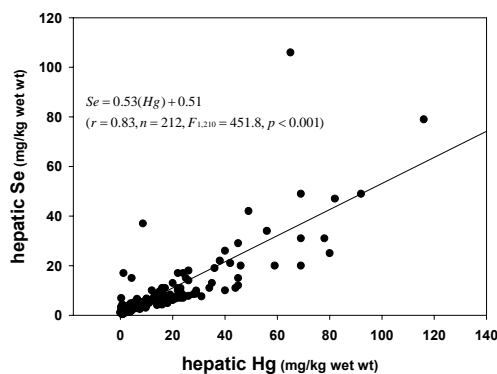
Figure 53 Hepatic Se concentrations (mg/kg wet wt) plotted against hepatic Hg concentrations (mg/kg wet wt).
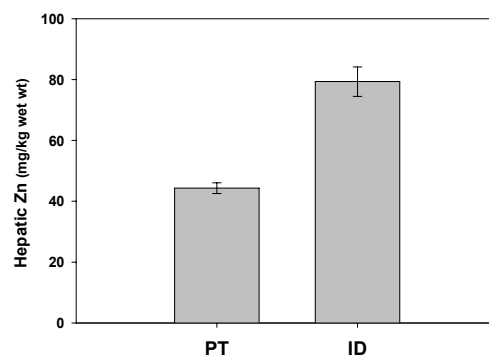
Figure 54 The mean liver Zn concentration (mg/kg wet wt) was significantly greater in the infectious disease (ID) group (n=79) compared to the physical trauma (PT) group (n=151). Bars = 1 S.E.

d)   Relationships between polybrominated flame retardants (PBDEs), butyltins (BTs) and health status in UK-stranded porpoises

The objective of these analyses was to test the hypothesis that chronic exposure to polybrominated flame retardants (PBDEs) and butyltins (BTs) may predispose harbour porpoises to mortality associated with infectious disease. Although PBDEs are considered less immunotoxic than PCBs (de Boer *et al.* 2000, Brouwer *et al.* 2001, Darnerud *et al.* 2001), butyltins have been shown to be immunotoxic in experimental studies (see Kannan & Tanabe 1997). In these analyses, summed liver concentrations of three butyltin compounds (tributyltin, dibutyltin and monobutyltin) (ΣBTs) and blubber levels of the sum of 15 individual BDE congeners (Σ15BDEs) in apparently healthy animals that died from acute physical trauma (e.g. bycatch) were compared statistically with ΣBTs and Σ15BDEs concentrations in animals that died due to a variety of infectious diseases.

METHODOLOGICAL APPROACH

Porpoises for which ΣBTs or Σ15BDEs data were available were divided into two groups based on the cause of death as determined by detailed post-mortem examination. Individuals with ΣBTs levels that died from acute physical trauma, bycatch or dystocia were included in the physical trauma category (n=86) and porpoises diagnosed to have died from disease processes caused by one or more infectious agents (parasitic, bacterial, mycotic or viral) were included in the infectious disease category (n=25). For porpoises with available Σ15BDEs data, 24 were included within the physical trauma group and 20 were included in the infectious disease group. Animals were excluded from the analyses if the cause of death could not be established or was attributed to other causes and no individuals were included or omitted on the basis of their contaminant levels (e.g. relatively high or low). The stranding regions were defined as: south-west England (Devon, Cornwall, Somerset and Gloucestershire); east coast of England (Northumberland to Essex); Wales and north-west England; the English Channel (Kent to Dorset); and Scotland.

One-way analysis of variance (Sokal & Rohlf 1995, SYSTAT 1998) was used to test whether ΣBTs or Σ15BDEs concentrations differed between the physical trauma and infectious disease categories. If any statistically significant associations were found, analysis of variance and regression models (Sokal & Rohlf 1995, SYSTAT 1998) would be used to test for additional relationships between ΣBTs and Σ15BDEs concentrations and the following independent variables: age, sex, nutritional status and region of stranding. For logistical reasons complete data were not available for all variables (e.g. age) and thus sample sizes for the statistical tests differ. Statistical analyses were performed using the natural logarithms (ln) of all continuously distributed data to meet the assumption of normality required by the statistical models.

RESULTS

There were no significant difference in mean ΣBTs levels ($F_{1,109}$ = 1.84, p > 0.15) between the physical trauma and infectious disease groups. Mean ΣBTs levels in the physical trauma group (n=86) were 0.17mg kg^{-1} wet weight and 0.26mg kg^{-1} wet weight in the infectious disease group (n=25). The mean age ($F_{1,90}$ = 0.08, p > 0.75) and sex (X^2_1 = 1.97, p > 0.15, n=111) did not differ significantly between the physical

trauma and infectious disease groups, although nutritional status ($F_{1,109}$ = 14.9, p < 0.001) was significantly thinner in the infectious disease group.

There was no significant difference in mean Σ15BDEs levels ($F_{1,42}$ = 0.27, p > 0.60) between the physical trauma and infectious disease groups. Mean Σ15BDEs levels were 1.84mg kg^{-1} lipid in the physical trauma group (n=24) and 1.87mg kg^{-1} lipid in the infectious disease group (n=20). The mean age ($F_{1,39}$ = 0.87, p > 0.35) and sex (X^2_1 = 0.36, p > 0.50, n=44) did not differ significantly between the physical trauma and infectious disease groups, although nutritional status ($F_{1,42}$ = 21.0, p < 0.001) was significantly poorer in porpoises dying of infectious disease than acute physical trauma.

INTERPRETATION OF TOXICOLOGICAL ANALYSES

a) Trends in contaminant levels

The mean, median and maximum blubber levels of Σ25CBs (mg/kg lipid) were around one order of magnitude higher than for the organochlorine pesticides and Σ15BDEs. Hepatic levels of Hg were also around one order of magnitude higher than other non-essential and potentially toxic metals such as Cd and Pb, although Cd and Pb are known to accumulate in higher concentrations in the kidney and skin/bone respectively (see Aguilar *et al*. 1999). Hepatic levels of ΣBTs (between <0.01 to 1.2 mg/kg wet weight) were comparable with numerous other marine mammal studies (Tanabe 1999).

Statistical analyses demonstrated significant independent associations between Σ25CBs levels and sex (lower levels in females), region (lower levels in Scotland) and a strong negative correlation with nutritional status. The effect of sex on Σ25CBs levels was lost when adults were excluded from the analysis and is undoubtedly due to the maternal offloading of PCBs and other lipophilic compounds during gestation and lactation (e.g. Aguilar *et al*. 1999, O'Hara & O'Shea 2001). Depending on species and parity, up to 90% of the maternal body burden of organochlorines can be offloaded via gestation and lactation to the calf, with primiparous females offloading the greatest quantity (e.g. Schwacke *et al*. 2002). Lower levels of organochlorine contaminants in harbour porpoises in Scotland (Kuiken *et al*. 1994b, Wells *et al*. 1994) may reflect lower inputs of contaminants into Scottish waters compared to other UK regions.

Most of the organochlorine pesticides analysed in this study demonstrated significant temporal declines between 1989 and 2000, even when the effect of lower organochlorine levels in Scottish porpoises was controlled for. In contrast, levels of Σ25CBs were both higher and more temporally stable with only a slight (but statistically significant) decline detected between 1991 and 2001. These differences reflect the greater persistence, and possibly the rate of input, of PCBs compared to the organochlorine pesticides within the marine environment (Borrell & Reijnders 1999). Blubber Σ25CBs levels were strongly and negatively correlated with nutritional status, but this correlation was only weakly held for ΣDDTs and was not significant for any of the other organochlorine pesticides or Σ15BDEs measured in the blubber. Although these negative correlations could be caused by the partial concentration of PCBs and DDTs in blubber due to loss of lipid volume during disease/starvation (Aguilar *et al*. 1999), they are also consistent with a potential causal relationship

between PCB and DDT exposure and loss of nutritional status mediated by PCB/DDT-induced (immuno)toxicity resulting in increased susceptibility to infectious disease (see Jepson *et al* 2005a).

Levels of Ag, As, Fe, Hg, Pb, Se and molar Hg:Se ratio were positively correlated with age in this study, with Hg, Se and Cd demonstrating the strongest positive correlation with age and hence rate of bioaccumulation. The greater degree of bioaccumulation of Hg, Cd and Se compared to other metals is consistent with a number of other studies (Aguilar *et al*. 1999). Hepatic levels of Ag, Cr, Ni and Pb were significantly higher in female porpoises and could be due to differences in spatial distribution of male and female porpoises leading to differences in dietary exposure. Differences in metabolic pathways of males and females linked to hormonal cycles have also been suggested to explain sex differences in accumulation of specific metals in other studies (Caurant *et al*. 1994).

Levels of Se were also correlated with levels of Hg, a phenomenon that has been shown in many other marine mammal studies including harbour porpoises (Koeman & van den Ven 1975, Palludan-Muller *et al*. 1993, Law 1996, Bennett *et al*. 2001) and is widely attributed to a Se-dependent biochemical detoxification pathway for Hg eventually involving the formation of biological inert mercury selenide (HgSe) (Cuvin-Aralar & Furness 1991, Augier *et al*. 1993, Palmisano *et al*. 1995). This mechanism would explain how some marine mammals are able to acquire levels of Hg that would be toxic in terrestrial mammals (O'Hara & O'Shea 2001). A trend towards a 1:1 molar ratio of Hg:Se in the liver has been shown in a number of cetacean and pinniped species as Hg is progressively bioaccumulated and detoxified by Se (Koeman & van den Ven 1975, Nigro & Leonzio 1996). In this study the hepatic molar Hg:Se ratio was also strongly correlated with age and was undoubtedly due to the age-related bioaccumulation of Hg and parallel bioaccumulation of Se for detoxification.

Hepatic Cu levels were significantly higher in neonates and may reflect a higher biological requirement in tissues undergoing rapid differentiation and development (Law 1996). Both Cr and Ni levels declined significantly between 1989 and 2001 and were significantly lower in porpoises in Scotland, possibly due to lower levels of industrial activity in Scotland reducing riverine inputs into the marine environment. Levels of Pb were positively correlated with age (bioaccumulation), higher in porpoises stranding in Wales than from other regions, and declined significantly between 1989 and 2001. These patterns are probably due to greater localised inputs of Pb into the marine environment from historical mining activity in Wales (Evans 1991, Pyatt & Collin 1999) along with decreased atmospheric inputs of Pb due to the reduction in use of alkyl lead derivatives in petrol (OSPAR 2000a) and reduced direct inputs (from sewage and industrial effluent) (OSPAR 2000b).

The strong regional variation in Cd levels, with porpoises in Scotland having the highest levels and porpoises in Wales having the lowest, may reflect regional variation in diet (Palludin-Muller *et al*. 1993). Cephalopods are known to concentrate Cd even in unpolluted waters (Bustamante *et al*. 1998a), and are an important link for Cd transfer in marine food chains (Bustamante *et al*. 1998b). High hepatic and renal levels of Cd in by-caught common and striped dolphins from the North-east Atlantic have been attributed to the high proportion of cephalopods in their diet (Das *et al*. 2000).

Levels of Zn were independently and negatively correlated with nutritional status; these associations are discussed in more detail below. Arsenic (As) levels were

positively correlated with both age (bioaccumulation) and nutritional status. The positive correlation with nutritional status may be related to recent dietary input in healthier animals. Levels of silver (Ag) increased significantly (and independently of any correlation with age) in UK-stranded porpoises between 1991 and 1997 and then levelled-off between 1998 and 2001. According to figures published by the Silver Institute (2002), global demand for Ag rose from 22.6 million kg in 1992 to 29.0 million kg in 2000 then fell to 22.3 million kg in 2001. Its use for photographic purposes rose steadily from 6.2 to 7.0 million kg over the period 1992 to 1999, and has since fallen to 6.4 million kg in 2001 and may continue to fall in the future as the impact of digital photographic technology reduces demands for silver halide (Silver Institute 2002).

From the overall levels of exposure of different contaminants in this study, combined with what is known about these contaminants from experimental and non-experimental studies, it is possible to make some general comments about the relative risks posed to UK harbour porpoises from these disparate groups of contaminants. In this study CB levels were higher and more temporally stable than DDT and the other organochlorine pesticides. Furthermore, some specific (coplanar) CB congeners are known to have significantly greater immunotoxicity than pesticides such as DDT and its metabolites (Saboori & Newcombe 1992, Safe 1994), the effect mediated principally through their greater affinity for the cytosolic Aryl-hydrocarbon (*Ah*)-receptor (Safe 1994, De Waal *et al.* 1997). The *in vitro* expression and demonstration of high affinity TCDD-binding to the beluga *Ah*-receptor further suggests that cetaceans as a taxonomic group may be particularly vulnerable to *Ah*-receptor-mediated immunotoxicity (Jensen & Hahn 2001). Furthermore, since only 30% of all PCBs produced have been dispersed into the environment and only 1% has reached the ocean, slow dispersal will continue globally and no apparent reduction in PCB levels will occur until sometime into the 21st century (see Borrell & Reijnders 1999). Therefore, of all the persistent organochlorine compounds analysed in this study, and many that were not, the PCBs undoubtedly pose the greatest threat to the health status of harbour porpoises in UK waters owing to their relative toxicity and greater biomagnification, abundance and persistence within marine food chains.

Of all the heavy metals examined in this study, Hg is likely to pose the greatest toxic threat to porpoises due to its greater biomagnification potential and relative toxicity. However, unlike PCBs and other organochlorines, marine mammals have evolved mechanisms for the detoxification of Hg utilising its high binding affinity for Se (Cuvin-Aralar & Furness 1991, Augier *et al.* 1993, Palmisano *et al.* 1995). A strong correlation between hepatic Hg and Se levels was also recorded in this study providing further evidence for this Se-dependent Hg detoxification process. Levels of other potentially toxic metals such as Cd, Cr, Pb & Ni occurred at relatively low levels compared to Hg in this study, and levels of Cr, Pb & Ni demonstrated significant temporal declines thus presenting a diminishing threat.

Hepatic ΣBTs levels in this study were positively correlated with age and were higher in females than males. Butyltin levels in cetacean livers have been shown in some other studies to increase with age until sexual maturity and then level off (Kim *et al.* 1996, Kannan & Tanabe 1997). Butyltin levels in harbour porpoises in this study were higher than in grey seals (*Halichoerus grypus*) and pelagic cetaceans found in UK waters (Law *et al.* 1998, Law *et al.* 1999) but were still comparable with butyltin levels in numerous other marine mammal studies (Tanabe 1999) and are low in comparison to PCBs and Hg levels. Since TBT has been used extensively in antifouling paints in

shipping, it is perhaps not surprising to find the highest levels in animals from the North Sea and Channel, both areas with a very high volume of shipping activity compared to other regions. The potential (immuno)toxicity of butyltins (Kannan & Tanabe 1997, Kannan *et al*. 1998, Nakata *et al*. 2002) appears to justify the ban imposed in 1987 on the use of TBT in antifouling paints in all marine vessels less than 25m in length and the total phasing-out of TBTs in antifouling paints on all maritime vessels by 2008.

Blubber Σ15BDEs levels were positively correlated with age (bioaccumulation), and were higher in porpoises from the East Coast of England. There were insufficient data to test rigorously for spatial or temporal trends since Σ15BDEs data were only available for 1996-1999 and none were available from Scotland. Current levels of PBDEs in UK harbour porpoises appeared considerably lower than for PCBs (Law *et al*. 2002) (Table 19). These levels are unlikely to pose a significant toxic threat to harbour porpoises in UK waters, but the ongoing use of these (or similar brominated flame retardant) compounds may pose an escalating threat in future due to their potential for endocrine disruption and other toxic effects (de Boer *et al*. 2000, Brouwer *et al*. 2001, Darnerud *et al*. 2001).

b) Relationships between PCB exposure and health status in UK-stranded porpoises

An association between elevated Σ25CBs and infectious disease mortality (when compared to a control group of individuals that died of acute physical trauma) has been shown previously in harbour porpoises stranded in England and Wales, albeit with a much smaller subset of this dataset (67 animals) (Jepson *et al*. 1999). In multivariate analyses, only the cause of death category remained significantly associated with variation in Σ25CBs, suggesting that the significant association between elevated Σ25CBs levels and infectious disease mortality is not solely explained by a concentrating effect on Σ25CBs levels in animals losing nutritional condition/blubber thickness. This is further supported by the lack of a significant difference in the percentage of hexane extractable lipid in blubber samples between the physical trauma and infectious disease groups.

To investigate this association between elevated Σ25CBs and infectious disease mortality in relation to the proposed blubber total PCB threshold (17 mg/kg lipid) for adverse health effects in marine mammals (Kannan *et al*. 2000), levels of total PCBs (based on *Aroclor 1254*) were calculated for all animals in this study. This threshold value, based on low grade physiological effects in experimental studies on a range of marine mammal species, was used not as an absolute value, but as a guide to determine whether levels of PCB exposure in individual animals are likely or not to exert significant biological (immunotoxic) effect based on empirical experimental data (Kannan *et al*. 2000). In porpoises with total PCB levels below 17 mg/kg lipid, there was no significant difference in total PCBs concentrations between the physical trauma and infectious disease groups. In porpoises with total PCBs above 17 mg/kg lipid, levels were significantly elevated in the infectious disease mortality group. These findings are therefore consistent with the proposed marine mammal threshold of 17 mg/kg lipid in blubber for PCB-induced adverse health effects including immunosuppression (Kannan *et al*. 2000). Since the 17 mg/kg total PCBs threshold value is based on low-grade physiological effects (Kannan *et al*. 2000), and given that some porpoises in the physical trauma group had higher total PCBs levels than 17 mg/kg, a true threshold level of total PCBs in harbour porpoises sufficient to

induce immunosuppression and predispose to infectious disease mortality at the individual or population level is likely to exceed 17 mg/kg total PCBs.

Considered collectively, these findings suggest that, above an estimated threshold of biological toxicity (17 mg/kg lipid), a causal (immunosuppressive) relationship exists between blubber total PCBs levels and animals that died of infectious disease that is not explained by nutritional loss of lipid mass having a concentrating effect on blubber CBs levels. Other studies of UK-stranded harbour porpoises have shown statistical links between quantitative indices of thymic involution (a potential biomarker of PCB-induced immunotoxicity) and elevated blubber $\Sigma25$CBs concentrations (particularly above the 17mg/kg lipid total PCBs threshold) that are consistent with PCB-induced immunosuppression (Jepson 2003).

c) Relationships between trace metals and health status in UK-stranded porpoises

In terms of potential toxicity, Hg is likely to pose the greatest threat of all the metals examined in this study. Most of the dietary Hg ingested by piscivorous marine mammals is in the organic form of methylmercury, which is the most toxic form (Law 1996). Hepatic mean Hg levels were also considerably higher than other toxic metals such as Cd and Pb in this study, and Hg is known to have greater capacity for bioaccumulation and toxicity (Langston 1990, Law 1996, Aguilar *et al*. 1999). Furthermore, Hg has indirectly been associated with disease and mortality in several marine mammal studies (De Guise *et al*. 1994, Siebert *et al*. 1999, Bennett *et al*. 2001). Although some other metals in this study such as Ag, Cu and Pb were also associated with disease status, the total hepatic concentrations of individual metals were often low and these associations were generally explained by the confounding effects of other more significant factors. Hepatic concentrations of As and Cu were actually significantly higher in the physical trauma group, not the infectious disease group, and these higher levels probably reflect recent dietary input within this group of relatively healthy animals. Therefore, it is the interactions between levels of Hg, Zn, disease status and nutritional status, together with the potentially detoxifying interaction of Se with Hg (including the molar Hg:Se ratio), that warrant the greatest scrutiny.

Zinc was also significantly higher in the infectious disease group and was negatively correlated with both indices of nutritional status, but not with age. Little is known about the significance of differing tissue levels of Zn in marine mammals (Augier *et al*. 1993). Zinc is an essential element for immune function and redistribution and elevation of hepatic Zn concentrations in humans is a result of acute-phase protein synthesis (Scott 1985; Hambridge *et al*. 1986; Amdur *et al*. 1991). Elevated Zn levels have been associated with infectious disease in other studies of harbour porpoises and may represent a response to infectious disease rather than a contributory cause (Bennett *et al*. 2001). The strong negative correlation between hepatic Zn levels and nutritional status could also represent redistribution of Zn bound to metalloproteins in muscle tissue during muscle proteolysis associated with loss of nutritive condition as has been shown in other mammals (Spencer *et al*. 1985, Krämer *et al*. 1993).

The interactions between Hg, Se and molar Hg:Se ratio, disease status, and both indices of nutritional status were complex. Although Hg levels were significantly elevated in the infectious disease group, Hg was also positively correlated with age and negatively correlated with nutritional status. The forward stepwise multiple regression analyses demonstrated that hepatic levels of Hg were significantly and

independently correlated with disease status (significantly higher in animals dying of infectious disease) and independently and positively correlated with age (attributed to bioaccumulation). Therefore, the possibility of redistribution of Hg from other tissues (e.g. muscle) to the liver due to loss of nutritional status, or the elevation of liver Hg concentrations due to loss of liver mass during catabolism may explain some, but probably not all, of the association between elevated Hg levels and animals dying of infectious disease.

Before an assumption of toxicological causality can be made between elevated Hg levels and infectious disease mortality, the role of Se in the detoxification of Hg must be considered. Hepatic Se levels were also strongly correlated with Hg levels, a phenomenon that has been shown in a considerable number of marine mammal studies including harbour porpoises (Paludan-Muller *et al*. 1993, Law 1996, Bennett *et al*. 2001). In this analysis, the molar Hg:Se ratio was also positively correlated with age (and hepatic Hg levels) thus providing further evidence of this phenomenon. Interestingly, the molar Hg:Se ratio was also significantly greater in the infectious disease group, an association previously demonstrated using a smaller dataset and suggested as potential evidence for the partial failure of the Se-based detoxification of Hg, possibly leading to Hg-induced immunosuppression or other toxicity (Bennett *et al*. 2001). However, the mean molar Hg:Se ratio in animals dying of infectious disease was still less than 1:1 in both studies, and although 31 porpoises had molar Hg:Se ratios greater than 1:1, only 13 of them died from infectious disease while 18 died of physical trauma.

In conclusion, these metals analyses clearly demonstrated a significant and independent association between hepatic Hg levels and infectious disease mortality that was not confounded by either age or (loss of) nutritional status. There was also a significant association between molar Hg:Se ratio and infectious disease mortality, that was not confounded by age or nutritional status. Therefore, despite the existence of a Se-dependent Hg detoxification mechanism in cetaceans, these results are not inconsistent with the hypothesis that chronic Hg exposure in harbour porpoises may be associated with susceptibility to infectious disease, possibly mediated by a partial failure of the Se-dependent Hg detoxification as evidenced by the higher Hg:Se ratio in the infectious disease group. In order to investigate these associations further it would be desirable to ascertain exactly what proportion of hepatic Hg was successfully detoxified by Se (or other detoxification mechanisms), and what proportion of Hg remained non-detoxified, in both physical trauma and infectious disease groups. Unfortunately, it was not possible to quantify the proportions of detoxified and non-detoxified Hg from the contaminant data in this study. Ultimately, the direction of causality of the associations between Hg and Hg:Se ratio, disease status and nutritional status require further research for full elucidation.

d)   Relationships between polybrominated flame retardants (PBDEs), butyltins (BTs) and health status in UK-stranded porpoises

Levels of exposure to butyltins and polybrominated diphenyl ether (PBDE) flame retardants were relatively low, at least compared to other potentially toxic contaminants such as PCBs and mercury. It is therefore perhaps not surprising that no associations were found in this study between ΣBTs or Σ15BDEs levels and infectious disease mortality in these UK-stranded harbour porpoises. However, samples sizes were smaller than the reported investigations of relationships between

health status of stranded porpoises and their exposure to PCBs and heavy metals (Jepson *et al*. 2005a; Jepson 2003). Larger samples sizes, particularly for PBDEs, would therefore be desirable to test these associations more rigorously and to account for future increases in PBDEs levels due to their continued use as flame retardants.

A partial ban of butyltins in antifouling paints was introduced in 1987, and the use of these compounds in marine paints is to be phased out completely by 2008. It is therefore unlikely that levels of butytins in marine fauna will significantly increase in future. Although levels of PBDEs were also low in porpoises in this study, the use of PBDEs as flame retardants is now widespread. Since these compounds share similar physico-chemical properties (lipophilic, bioaccumulative and resistant to environmental degradation) to PCBs, it is possible that these compounds will continue to bioaccumulate in marine food chains and for levels to escalate over time. Ongoing monitoring of PBDEs in marine biota, and particularly in marine top predators like harbour porpoises, is clearly warranted.

3.3 SUMMARY OF ADDITIONAL (PEER-REVIEWED) RESEARCH ACTIVITY

Additional collaborative research activity resulted in a range of peer-reviewed scientific publications between 2000 and 2004 (see **Annex D**). Some of these publications are summarised here.

ANATOMY

The (immuno)histological development of the testis in the harbour porpoise has been recently described in detail (Holt *et al*. 2004) and follows earlier work on the testicular and ovarian development in the harbour porpoise (Karakosta *et al*. 1999). This research provides valuable baseline data from which to study potential links between cetacean reproductive development and endocrine disrupting chemical (pollutant) exposure.

DISTRIBUTION, ABUNDANCE AND POPULATION STRUCTURE

Recent studies of trends in the frequency and spatial and temporal distribution of particular cetacean species, in conjunction with cetacean sightings data, have provided important data relating to changes in cetacean habitat usage that may in turn carry implications for the management and conservation of these species. A mitochondrial DNA study at Aberdeen University, using samples collected from UK-stranded bottlenose dolphins, suggests that the Moray Firth population has less genetic diversity than evidenced from samples collected from bottlenose dolphins found elsewhere in the UK (Parsons *et al*. 2002). A second collaborative study involving the Sea Mammal Research Unit (St Andrew's University) and Aberdeen University used both sightings and strandings data to demonstrate an expansion of the range of the "Moray Firth" bottlenose dolphin population moving both north and south of the Moray Firth region. Even more recently, a combined cetacean

sightings/strandings study in north-west Scotland showed increases in some warmer water species such as striped dolphins in recent years and declines in some colder water species such as the white beaked dolphin, possibly in response to long-term changes in ocean temperatures (MacLeod *et al.* 2005).

DIETARY STUDIES

A range of dietary studies have been conducted at Aberdeen University using samples collected from stranded cetaceans in Scotland. Analyses and identification of prey species from the stomachs of stranded bottlenose dolphins show that they eat a wide variety of species that is probably only limited by availability and size (Santos *et al.* 2001). A second paper suggested regional variation in the diet of sperm whales stranded in the North Sea (Santos *et al.* 2002). Another study showed that minke whales have a wide variety of prey species and that sandeels form an important component of their diet at certain times of the year (Pierce *et al.* 2004). Finally, a fourth paper demonstrated seasonability in the diet of harbour porpoises consistent with harbour porpoises being opportunistic feeders (Santos *et al.* 2004).

MICROBIOLOGY

Publications on cetacean microbiology, many produced in collaboration with a number of research institutions, have been generated between 2000 and 2005 from UK cetacean strandings including a number on Brucella species infection in marine mammals. These include molecular typing of marine mammal *Brucellae* that further demonstrates their distinction from terrestrial species (Bricker *et al.* 2000); the further molecular classification of marine mammal *Brucellae* (Cloeckaert *et al.* 2001); and chronic meningoencephalitis associated with *Brucella* sp*.* infection in live-stranded striped dolphins (*Stenella coeruleoalba*) (Gonzalez *et al.* 2002). This condition has subsequently been found in other species of cetacean and a new manuscript reporting these findings is in preparation. Some of these studies have impacted significantly upon theories for the evolution of this major zoonotic genus. A commissioned review of the extent of *Brucella* infection in marine mammals covering geographic spread, species affected, microbiology, serology, epidemiology, zoonosis and potential hazards for other species has also been undertaken (Foster *et al.* 2002). Brucella infection has also been reported in marine mammals off the Cornish coast (Dawson *et al.* 2004) and within the lungworms from a harbour porpoise that stranded in Cornwall (Perrett *et al.* 2004).

Other studies include the establishment of a new species within the *Pasteurellaceae* for an organism recovered from the uterine tract of porpoises (Foster *et al* 2000a); a letter extending the animal host range, including porpoises, of *Plesiomonas shigelloides* (Foster *et al* 2000b); the establishment of a new *Streptococcus*-like species for an organism recovered from porpoises and seals (Hoyles *et al.* 2000); the establishment of a new Gram positive bacterial genus for an organism recovered from porpoise and deer (Lawson *et al.* 2001). This *Arcanobacterium pluranimalium* has also been established as a cause of infection in sheep where it has been associated with spontaneous abortion and other conditions (Foster *et al.* 2001). Other publications include the establishment of a new *Actinomyces* species for an organism recovered from cetaceans and seals (Hoyles *et al.* 2001); the

establishment of a new species within the genus *Campylobacter* for an organism recovered from porpoises and seals (Foster *et al.* 2004) and the establishment of a species of *Streptococcus* for an organism recovered from seals (Lawson *et al.* 2005) that has subsequently been recovered from a harbour porpoise.

POTENTIAL IMPACTS OF ANTHROPOGENIC NOISE ON CETACEANS

The emergence of gas and fat embolism in cetaceans, including a mass stranding of beaked whales in the Canary Islands in 2002 linked to a military exercise using mid-frequency active sonar, has challenged conventional concensus about the adaptations of some species of deep-diving marine mammals and suggested a potential mechanism for the phenomenon of mass cetacean strandings (mainly involving beaked whales) involving an acoustically-induced condition similar to decompression sickness in human beings and experimental mammals (Jepson *et al.* 2003; Fernandez *et al.* 2004). These studies were reported in collaboration between pathologists working on cetacean strandings in the Canary Islands and in the UK. The emergence of gas and fat embolism as a potential causal mechanism in beaked whale mass strandings is gaining increasing support within the scientific community (Crum *et al.* 2005; Fernandez *et al.* 2005; Cox *et al* in press; Rommel *et al.* in press). The single-stranded cases of acute and chronic gas and fat embolism in UK-stranded cetaceans have already been described in this report (see also Jepson *et al.* 2005b).

TOXICOLOGY

In addition to the toxicological studies already presented in this report, a number of other publications on contaminant exposure in cetaceans from UK waters were generated during the period of this report using tissue samples from UK-stranded animals and toxicology data derived from the CEFAS Burnham Laboratory, Burnham-on-Crouch, Essex.

Two studies summarised the current state of contamination of offshore cetaceans by organotin compounds derived from TBT-based antifouling paints (Law *et al.* 1999; 2000), and fed into discussions of a ban on the application of these products to all vessels (enacted in 2005). Other studies summarised the current state of organochlorine contamination of pelagic cetaceans feeding offshore remote from sources (Law *et al.* 2001); reported levels of polybrominated diphenyl ether (BDE) flame retardants in porpoises and cormorants (with quite high levels in porpoises) (Law *et al.* 2002); summarised the current state of contamination with persistent organohalogen compounds of UK marine mammals (Law *et al.* 2003); and established concentrations of BDE flame retardant compounds in a wide range of cetacean species, including some feeding in deep offshore waters, and showed the widespread distribution of these compounds in the environment feeding into the EU risk assessment (Law *et al.* 2005).

4.0 DISCUSSION AND CONCLUSIONS

No unusual cetacean or turtle mass-mortality events occurred in the UK between 2000 and 2004. Between 1st January 2000 and 31st December 2004 3197 UK-stranded cetaceans of 18 species were reported and the annual numbers of these strandings increased on an annual basis from 420 in 2000 to 788 in 2004. These are the highest annual numbers of cetacean strandings recorded within the UK since records began in 1913. Consistent with previous years, the most common UK-stranded cetacean species in the 2000 to 2004 period was the harbour porpoise (*Phocoena phocoena*) of which 1596 were recorded. The progressively increasing annual numbers of UK-stranded cetaceans reported between 2000 and 2004 was predominantly due to increasing numbers of both common dolphins (*Delphinus delphis*) and harbour porpoises reported stranded in south-west England. The annual number reported of most other cetacean species have shown relatively small inter-annual fluctuations. However, subtle shifts in the relative abundance and distribution of strandings of southern "warmer" and northern "colder" waters species, possibly attributable to climate change and increasing oceanic temperature, have also occurred over time. More specifically, there has been a relative increase in stranding frequency of striped dolphin (*Stenella coeruleoalba*) and a relative decrease in stranding frequency of white beaked dolphins (*Lagenorhynchus albirostris*) in north-west Scotland since 1992 (MacLeod *et al.* 2005).

The main cause of death of the vast majority of common dolphins and harbour porpoises stranded in SW England between 2000 and 2004 inclusive was entanglement in fishing gear (bycatch). This annual and seasonal (mainly December-March) phenomenon of increased numbers of stranded harbour porpoises and common dolphins (comprising mainly bycatches) in recent years has received considerable campaigning and media focus. Although a range of factors such as changes in cetacean distribution, change in population size, increase in bycatch and other causes of mortality or variations in climatic factors could be involved, the trend of progressively increasing reports of both harbour porpoises, common dolphins and even striped dolphins in the same region suggest that the main cause of these increasing stranding reports is not biological. The most likely cause of the increased reports of cetacean strandings in southwest England is increased coastal vigilance for strandings. The relatively high number of cetacean strandings reported (mainly common dolphin and harbour porpoise bycatches) particularly in south-west England since the early 1990s has undoubtedly lead to a greater awareness of the bycatch issue within the region and is likely to have precipitated increased vigilance for cetacean strandings in this coastal area.

Although mid-water pelagic trawl fisheries aimed at bass have been widely suggested to be the principal cause of the increasing number of common dolphin and harbour porpoise bycatches reported in south-west England between 2000 and 2004, there is a poor correlation with these increasing reports of strandings between 2000 and 2004 and temporal trends in the pelagic pair-trawl bass fishing effort. In contrast to trends in cetacean strandings in southwest England, effort in the pelagic pair-trawl bass fishing increased significantly between 1995 and 1999 and then reached a plateau between 2000 and 2004 (SMRU, 2004, *in prep*). Similarly, the seasonal

distribution of both harbour porpoise and common dolphin bycatch (with a seasonal peak in strandings/bycatches in January for both species) does not particularly correlate with the seasonal peak in fishing effort in March of the mid-water pelagic bass-fishery. Stomach content analyses from common dolphin bycatches stranded in the SW of England during the 1990s and 2000s have shown a decline in the proportion of mackerel, but no bass have ever been found (SMRU, 2004, *in prep*). In addition, although scientific observers in the bass-trawl fishery have recorded over 300 common dolphins since 2000, no bycatches of harbour porpoises have been recorded in this fishery. The mid-water pelagic bass fishery is therefore unlikely to be responsible for the increase in harbour porpoise strandings reported in southwest England.

Analysis of body-length and age data from harbour porpoise strandings demonstrated highly significant regional morphometric (i.e. standardised body length/size measurements) variation characterised by greater body-length and body-length-to-age ratios in porpoises stranding in south-west England compared to other regions, suggesting that at least two distinct harbour porpoise populations or stocks exist within UK waters. Genetic studies using mitochondrial DNA (Walton 1997) and nuclear DNA micro-satellites (Andersen *et al.* 1999) suggested some significant differences in harbour porpoises from the Irish/Celtic Seas and the North Sea, but none suggesting that porpoises in south-west England were particularly distinct from other UK regions. A genetic study of UK-stranded harbour porpoises using microsatellite loci has, however, found significant differences between porpoises in southwest England and other parts of the UK (Thatcher 2005). This evidence of morphometric, genetic and spatial isolation of the Celtic Sea porpoise stock suggests that it may be particularly vulnerable to regional decline or disappearance if previously quantified levels of unsustainable bycatch in Celtic Sea gillnet fisheries (Tregenza *et al.* 1997) are maintained.

The frequency and proportion of other forms of physical trauma, specifically fatal attacks on harbour porpoises by bottlenose dolphins, have increased significantly in recent years in areas of sympatric distribution in west Wales and south-west England and are not considered to be due to either increased observer effort or changes in diagnostic criteria (Deaville *et al.* 2005). The distribution of stranded harbour porpoises killed by bottlenose dolphins off north-east Scotland has increased in geographic range in recent years, reflecting a change in the distribution of resident bottlenose dolphins (Wilson *et al.* 2004). Although a number of hypotheses have been proposed to explain these violent and often fatal interactions (Ross and Wilson 1996), their cause(s) has not been established.

Mortality due to infectious disease was rare in all species examined apart from harbour porpoises. Most fatal infectious disease cases in harbour porpoises were caused by endemic infections with parasitic, bacterial and mycotic agents and are consistent with findings in stranded harbour porpoises from Germany (Siebert *et al.* 2001) and Belgium/Northern France (Jauniaux *et al.* 2002). Most cases of infectious disease mortality in harbour porpoises occurred in juvenile individuals around 1 and 2 years of age, a period when most porpoises have developed parasitic infections. Lower numbers of fatal parasitic infections in later years may be partly due to acquired immunity following repeated parasitic exposure. Although cetacean morbillivirus (CMV), a viral agent capable of causing cetacean epizootic mortality, was identified in two UK-stranded harbour porpoises in 1990 (Kennedy *et al.* 1992,

Kennedy 1998), distemper due to CMV infection has not been diagnosed since in any UK-stranded cetacean. The low seroprevalence of antibodies to morbilliviruses in harbour porpoises, common dolphins and other UK-stranded cetaceans examined between 1989 and 1999 (Van Bressem *et al.* 1998, 2001) suggests that CMV is not endemic in cetacean species such as harbour porpoises and common dolphins found in UK waters, and supports the suggestion that these species may be accidental hosts for cetacean morbilliviruses (Van Bressem *et al.* 1998).

Other potential primary pathogens, such as bacteria of the genus *Brucella*, are now known to infect a wide range of marine mammals including harbour porpoises in UK and other oceanic waters (Foster *et al.* 1996; Ross *et al.* 1996; Jepson *et al.* 1997; Foster *et al.* 2002). Although relatively few fatal cases of brucellosis were recorded in harbour porpoises in this study, there have been a number of cases of fatal *Brucella* sp. meningoencephalitis in pelagic dolphins in Scottish waters. Since the major association with *Brucella* spp. infection in terrestrial animals is abortion and infertility, and there is widespread evidence of exposure to *Brucella* spp. in marine mammals (Foster *et al.* 1996; Ross *et al.* 1996; Jepson *et al.* 1997; Foster *et al.* 2002), it is possible that *Brucella* infection is having a significant, but hitherto undetected, impact on cetacean populations. The zoonotic potential of these organisms has been further realised with the reporting of two cases of human neurobrucellosis with intracerebral granuloma caused by a marine mammal *Brucella* spp. (Sohn *et al.* 2003).

Associations between infectious disease mortality and elevated blubber levels of polychlorinated biphenyls (PCBs) (Jepson *et al.* 1999; Jepson *et al.* 2005a; Bull *et al.*, in press) and hepatic levels of mercury (Hg) (Bennett *et al.* 2001) identified in UK-stranded harbour porpoises (including many individuals in this study), suggest that these persistent, bioaccumulative and immunosuppressive pollutants (especially PCBs) influence the susceptibility to, and pathogenesis of, potentially fatal infectious diseases. Harbour porpoises have high-energy requirements for thermoregulation (given their small body mass and temperate habitats) and annual reproduction at sexual maturity (McLellan *et al.* 2002), and it is plausible that a suite of species-specific additional stressors such species-specific high-intensity parasite infestations and chronic PCB exposure, acting additively or synergistically, may influence the epidemiology and pathogenesis of infectious disease in harbour porpoises in European waters. Future risk assessment-type approaches (e.g. Schwacke *et al.* 2002) may enable the prediction of past, present and future PCB-induced harbour porpoise mortality in European populations of known size and PCB exposure using dose-response curves derived from empirical pathological and toxicological studies of stranded animals.

Although samples sizes and therefore statistical power varied between different contaminant groups in these studies, associations have been demonstrated between elevated PCBs and Hg levels and thymic atrophy (Jepson 2003), elevated PCBs and infectious disease mortality (Jepson *et al.* 2005a), and elevated liver Hg and molar Hg:Se ratios and infectious disease mortality that consolidate findings in earlier studies utilising smaller datasets (Jepson *et al.* 1999, Bennett *et al.* 2001). In particular, the associations between PCBs and infectious disease mortality were highly significant above a proposed threshold of toxicity (17 mg/kg total PCBs lipid weight) in aquatic mammals (Kannan *et al.* 2000), but absent at sub-threshold levels, and provided the most compelling evidence that these persistent, bioaccumulative and immunosuppressive pollutants (especially PCBs) influence the susceptibility to,

and pathogenesis of, potentially fatal infectious diseases in exposed harbour porpoises (Jepson *et al.* 2005a).

Levels of other (less toxic) metals, butyltins and polybrominated diethylethers (PBDEs) were generally considerably lower than for PCBs and mercury, and were not associated with infectious disease mortality. The low prevalence of neoplasia within this study was also consistent with the low levels of carcinogenic compounds such as radionuclides (Watson *et al.* 1999) and polycyclic aromatic hydrocarbons (Law & Whinnett 1992) that have been recorded in harbour porpoises in UK waters. The long-term nature of these investigations has enabled significant temporal declines in levels of some contaminants to be detected in UK harbour porpoises. In addition to the morphometric variation in stranded harbour porpoises from different UK regions, spatially specific variation in contaminant exposure (e.g. elevated lead levels in Welsh porpoises) was also evident. Collectively, these findings suggest that porpoises in UK waters exhibit some degree of philopatry and possibly population structuring as has been shown in other harbour porpoise studies (e.g. Westgate & Tolley 1999).

During the period of this report a new disease entity characterised by acute and chronic tissue injury associated with systemic gas (and in some cases fat) embolism was described in a small number of UK-stranded cetaceans (Jepson *et al.* 2003; Fernandez *et al.* 2004; Jepson *et al.* 2005b). Although the aetiology and pathogenesis of the gas and fat emboli in these UK cases remains uncertain, the findings provide the first evidence that bubbles can form *in vivo* in cetaceans and persist through time. The similarities of these lesions with decompression sickness (DCS) in other mammalian species, and their higher prevalence in deep-diving species like Risso's dolphins (*Grampus griseus*) and beaked whales, suggests that a DCS-like mechanism could be involved in their pathogenesis. Although acute and widely disseminated gas and fat embolic lesions consistent with DCS have been identified in beaked whales that mass stranded during naval exercises using active sonar (Jepson *et al.* 2003; Fernandez *et al.* 2004; Fernandez *et al.* 2005; Cox *et al.* in press), these UK findings demonstrate that gas embolism can occur in species other than the deep divers and possibly in circumstances other than mass strandings. Further research should investigate the incidence, aetiology and pathogenesis of bubble lesions, and undertake retrospective and prospective examination of cases for gas emboli and associated lesions, which may have been historically under-reported in marine mammal pathology.

In addition to the results of the studies described in this report, the marine mammal database and tissue archive developed in the UK since 1990 continues to support a broad range of parallel and inter-disciplinary research activity. Currently, over 13, 500 records and 50, 000 tissue samples are held as a strategic resource that supports a range of research activity including basic life history and biology, diet, morphometrics, systematics, distribution, population modelling and stock structure, disease epidemiology, pathology, microbiology, toxicology, parasitology and other studies. A list of peer-reviewed publications and national and international research collaborations that utilise tissues and data derived from UK-stranded cetaceans are provided in **Annex D and E**. Wherever possible (and subject to Defra approval), collaborative research activities of high scientific quality are supported by facilitating access to data and tissues collected from post-mortem examinations of UK-stranded

cetaceans, although the short-term goals of some research proposals must be balanced against the finite nature of archived tissue samples.

In conclusion, the results of these investigations have identified and monitored trends in a number of threats to the health and conservation status of cetaceans in UK waters. The bycatch of dolphins and porpoises in commercial fisheries, particularly but not exclusively in waters around south-west England, are a longstanding welfare and potential conservation concern. Although levels of many pollutants such as lead (Pb), nickel (Ni) chromium (Cr) and several organochlorine pesticides have demonstrated significant temporal declines during the period of this study, the most toxic compounds (PCBs and mercury) were found in the highest concentrations and will probably pose an ongoing threat to the conservation status of coastal and piscivorous cetaceans such as harbour porpoises and bottlenose dolphins inhabiting industrialised regions for decades to come. Newer chemical pollutants emerging in the marine environment such as polybrominated flame retardants, the long-term effects of climate change on cetacean distribution and health, are other issues that require ongoing monitoring of cetacean strandings in UK and European waters. Future studies should also investigate how gas bubble formation can occur in diving cetaceans, including the characterization and quantification of any contributory factors (e.g. dive behaviour, depth of complete lung collapse, critical levels of nitrogen tissue supersaturation, acoustic exposure) to bubble development (Cox *et al.*, in press).

5.0 RECOMMENDATIONS FOR FUTURE RESEARCH

- Individual and population-level impacts of cetacean bycatch are undoubtedly important welfare and conservation concerns and research into cetacean bycatches and strandings are ongoing requirements of conservation agreements such as ASCOBANS. The UK Government is already funding a broad range of research activity (principally co-ordinated by the Sea Mammal Research Unit) into the assessment and mitigation of cetacean bycatch. Data and tissues collected from UK cetacean strandings contribute significantly to some of these (mainly Defra-funded) research activities on an ongoing basis. Future bycatch-related research activities using strandings data should include long-term monitoring of numbers of by-caught strandings in the UK and their relation to changes in fishing effort, forensic examination of net marks to infer specific types of fisheries/fishing gear involved through a comparison of directly observed bycatches and stranded individuals, assessment of diet and life history parameters (e.g. age/sex structure; individual growth rates, age at sexual maturity, fecundity) in by-caught and stranded carcasses, and investigations on how these parameters may change over time in response to bycatch and other ecological factors. Morphometric and genetic data from strandings can also help to infer population structure (especially for harbour porpoises) and life history and other biological data (controlled for health status/cause of death) can continue to contribute to modelling components of management models for bycatch such as the EU- and Defra-funded SCANSII project.

- Investigate emergence of new contaminants and trends in cetacean contaminant exposure.   These research activities not only enable trends in specific contaminants to be monitored but also enable investigation of newer emerging contaminants of potential importance in the marine environment (such as newer polybrominated flame retardants).   Data deriving from the UK Cetacean Strandings programme has already fed into the EU risk assessments of flame retardant compounds and contributed to a ban on the production and use of the penta-mix and octa-mix polybrominated diphenyl ether flame retardant formulations. The assessment of hexabromocyclododecane (HBCD) is ongoing, and high concentrations of HBCD have already been shown in UK-stranded porpoises providing compelling evidence for its persistence. Monitoring pollutant levels in key species such as harbour porpoises also enable specific targets for these compounds to be set. For example, specific targets for the reduction of PCBs and other contaminants have been set in the Species Action Plan for the harbour porpoise (under the UK Biodiversity Action Plans). The data are also important in discussions with other governments in fora such as OSPAR where it contributes to our national perspective on the significance of strategies on the control of chemicals. Finally, new studies of radionuclide levels in coastal species like harbour porpoises would be of considerable value to assess exposure from existing nuclear power stations plus any new stations commissioned in the future.

- Further investigate potential relationships between pollutant exposure and health status in cetacean populations. Some work has already been done on UK-stranded harbour porpoises but much more can be done in this and possibly other species such as bottlenose dolphins to develop dose-response curves for contaminant-related effects such as infectious disease mortality and reduced reproductive success (e.g. Schwacke *et al.* 2002; Jepson *et al* 2005). More

detailed investigations of these associations with larger pathological/toxicological datasets from stranded and by-caught animals may enable future risk-assessment-type predictions of population-level effects of specific pollutants (e.g. PCBs) on cetacean populations in UK/European waters of known size and pollutant exposure. Assessing population-level impacts of pollutants on cetacean health and fertility are key components of management strategies, especially for species with small population size and high pollutant exposure.

- Considerable advances in molecular diagnostic techniques used to characterise known and novel pathogens have been made in recent years. The collection and archiving of tissues and data within the UK Cetacean Strandings Programme represents a valuable resource for the surveillance of emerging disease threats and facilitates the investigation of the origin and nature of emerging pathogens. Some pathogens already identified (e.g. Brucella) also have the potential to infect humans and cause disease. These new molecular diagnostic techniques will permit improved future screening and early detection of emerging pathogens of potential conservation and zoonotic significance and will aid the rapid investigation and diagnosis of any future marine mammal mass mortality events.

- A major issue for years to come will be the possible effects of climate change on cetacean distribution, prey availability and cetacean health (including exposure to novel pathogens). A 2004 workshop report on Habitat Degradation from the Scientific Committee of the International Whaling Commission Scientific Committee Workshop recommended that continued long-term research monitoring of cetaceans and key biotic and abiotic features of the environment are a critical aspect in understanding the impact of habitat degradation (including climate change) on cetacean populations. Effects potentially related to climate change are already being observed in the marine environment in UK and other European waters (e.g. MacLeod *et al.* 2005). Cetacean strandings provide one of the few mechanisms of investigating trends in key parameters such as cetacean diet, health status and pathogen exposure over the short, medium and longer time-scales in parallel with other biotic and abiotic factors involved in climate change. The UK already has one of the most advanced strandings networks in Europe with good baseline data to compare with findings from future investigations.

- Although there are growing concerns related to the potential impacts of anthropogenic noise on cetaceans, the investigations in this report have not found any confirmed evidence of noise-related impacts on cetaceans in UK waters. However, the absence of robust biomarkers of exposure to noise, together with the absence of similarly robust biomarkers of noise-related effects, makes the investigation of any cause-effect relationships relating to noise exposure very problematic. Nonetheless, Defra are currently funding a small feasibility study on post-mortem examinations of UK-stranded cetacean ears in collaboration with Dr Ursula Siebert's research group in Busem, Germany. If this feasibility study is successful, the potential exists for this research collaboration to develop into a larger study to investigate potential auditory effects of anthropogenic noise on cetacean hearing using ears collected from stranded cetaceans in UK and European waters.

- The emergence of gas embolism as a pathological entity in cetaceans raises a number of important questions in relation to the aetiology, pathogenesis and potential biological impact of this new phenomenon, and the possibility that

acoustic factors (like naval sonar) may play a role in bubble formation and beaked whale mass strandings has received considerable scientific and political focus (see Jepson *et al.* 2003, 2005, Fernandez *et al* 2004, 2005; Cox *et al.* in press). Future examination of stranded animals should investigate the species range, prevalence, individual and population-level effects (if any) and the factors involved in the aetiology, pathogenesis and incidence of cetacean gas embolism, including the potential role of high-intensity anthropogenic noise activity. A key component of these research goals will be the prospective and retrospective investigation of gas (and fat) embolic lesions in pathological studies of cetaceans, which may have been historically under-reported in marine mammal pathology.

- The continued maintainance and development of the UK Cetacean Strandings and *Poseidon* databases, together with the national tissue archives, undoubtedly plays a very strategic role by supporting a wide and inter-disciplinary range of present and future research activities. The data and tissue collection protocols can also be readily adapted to meet future research needs (e.g. via the addition of extra tissue samples to cetacean post-mortem protocols). The critical mass of data and tissues already held by the UK Cetacean Strandings programme, in conjunction with its flexibility to adapt to future research needs, is undoubtedly one of the great strengths of the programme.

- Defra has already indicated a desire to develop a dedicated website for the Cetacean Strandings Project. Such a website, if developed in the future, would be an excellent means of disseminating information derived from the Strandings Project, establishing collaborative links with other scientific organisations, and educating the public.

Acknowledgements

ANNEX A: REFERENCES CITED

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ANNEX B

Materials and Methods

Reporting and collection of marine mammal strandings

The reporting, retrieval and transportation of marine mammal carcasses within England and Wales involves the integration of a number of localised reporting centres with the Institute of Zoology and the Natural History Museum (see below). IoZ works closely with the NHM and other organisations to ensure cetacean carcasses (and a small number of seal carcasses) suitable for post-mortem examination are promptly and successfully retrieved. The decision about whether to subject a carcass to post-mortem is based on the state of decomposition and whether it can be secured safely prior to collection and transportation to a laboratory for post-mortem examination. The relevant public health considerations of handling stranded marine mammal carcasses are stressed to those individuals and organisations that are involved with the day-to-day reporting and recovery of stranded marine mammal carcasses.

Central reporting centres

- *The Natural History Museum, Cromwell Road, London, SW7 5BD.*
  Tel: 020 7942 5155. Records all UK cetacean strandings (since 1913) and accords reference numbers. Transports some cetacean carcasses within England to the Institute of Zoology for post-mortem examination. The UK Cetacean Strandings webpage is http://www.nhm.ac.uk/research-curation/projects/strandings/
- *Institute of Zoology, Zoological Society of London, Regent`s Park, London NW1 4RY.*
  Tel: 020 7449 6672. Receives and records seal strandings within the UK and administers national reference numbers.
- *Scottish Agricultural College, Inverness, Stratherrick Road, Inverness, IV2 4JZ.*
  Tel: 01463 243030. Receives marine mammal strandings reports and arranges transportation of carcasses for post-mortem examination within Scotland.
- *Marine Environmental Monitoring, Penwalk, Llechryd, Ceredigion SA43 2PS. National Strandings Line for Wales.*
  Tel: 01348 875000. Co-ordination of all marine mammal and turtle strandings in Wales. Administers national marine turtle strandings/sightings database. Funded separately under contract to the Welsh Assembly (contract ref: FC 73-02-273).

Regional Reporting Centres

- *Cornwall Wildlife Trust, Five Acres, Allet, Truro, Cornwall TR4 9DJ.*
  Tel: 0845 2012626. Co-ordinates all marine mammal strandings within Cornwall.
- *National Marine Aquarium, The Fish Quay, Plymouth PL4 0LH.*
  Tel: 01752 600301. Reporting and collection of some marine mammal carcasses within the Dorset/Devon area.

Reporting of live cetacean strandings

There has been increasing concern about the welfare of live marine mammal strandings within the UK in recent years. Although a number of rehabilitation centres have now been established for the nursing of sick or injured seals and seal pups (such as the RSPCA Wildlife Hospital in Norfolk), there was until recently little provision for cetaceans. Across the UK, the RSPCA (England and Wales) and the SSPCA (Scotland) are the first point of contact for members of the public reporting live-stranded or distressed marine mammals. The national hotline numbers are **0870 5555999** [RSPCA] and **0870 7377722** [SSPCA].

In recent years, the UK-based organisation *British Divers Marine Life Rescue* (BDMLR) has seen a large expansion of its activities including the training of over 2,000 BDMLR-trained Marine Mammal Medics. The BDMLR Marine Mammal Medic training course, and associated *Marine Mammal Medic Handbook*, has been approved by the Royal College of Veterinary Surgeons and all trained Marine Mammal Medics are fully insured. BDMLR also organise special marine mammal rescue workshops for veterinary surgeons. The BDMLR UK hotline is: **01825 765546**. Further details of the organisation can be found on the BDMLR website at: http://www.bdmlr.org.uk/.

A coalition of several welfare and rescue groups or societies called the *Marine Animal Rescue Coalition (MARC)* has been established for some years to provide a forum for the exchange of information, expertise and resources, and to help develop a more coherent and integrated strategy of response to live strandings of marine mammals. This forum is usually attended by strandings researchers from IoZ and SAC along with representatives from a broad range of other animal rescue and welfare organisation other organisations such as RSPCA and BDMLR. Data and expertise on pathology of live stranded marine mammals developed by the UK Marine Mammal Strandings Programme continues to provide informed input to this forum.

Pathology laboratories

The majority of post-mortem examinations were carried out at one of the following pathology laboratories.

- ***Scottish Agricultural College (Veterinary Science Division), Drummondhill, Stratherrick Road, Inverness, Scotland, IV2 4JZ.***
- Institute of Zoology (Zoological Society of London), Regent's Park, London, NW1 4RY.
- Veterinary Laboratories Agency (Truro), Polwhele, Truro, Cornwall, TR4 9AD.
- Department of Veterinary Pathology, University of Liverpool, Neston, Wirral, L69 3BX.

Standard protocols for post-mortem examination, tissue sampling and data recording

The fundamental basis of this contract is that each laboratory conducts post-mortem investigations, tissue sampling and recording of observations using standard procedures (Law 1994). Carcasses were routinely transported to one of the four pathology laboratories listed above. In cases where carcasses were too large or too difficult to retrieve, post-mortem investigations were conducted *in situ* at the stranding site.

Virological examinations. Routine samples from standard tissue sites on the carcass were preserved at -70°C (or -20°C at SAC Inverness) together with samples of any lesions suspected to be of possible viral origin.

Bacteriological examinations. Routine samples were taken from specified tissues for bacteriological culture, together with any lesions suspected to be of bacterial (or mycotic) aetiology, and examined for bacterial and mycotic organisms using standardised protocols. All cultures were incubated at 37°C. In most laboratories, Columbia blood agar with sheep blood was incubated in 10% CO_2, observed daily for four days and at frequent intervals thereafter until 14 days before discarding. Farrell's medium was observed at 3-4 days incubated in 10% CO_2 and frequent intervals thereafter until 14 days. The 14-day period was chosen because Brucellae recovered from seals are occasionally slower growing on Farrell's medium. Other media routinely utilised include MacConkey agar incubated aerobically and observed after 24 and 48h and, for respiratory tissues only, chocolate agar incubated in 10% CO_2 for isolation of *Haemophilus* spp. Anaerobic cultures were normally only performed on significant lesions using Oxoid anaerobic agar, and anaerobic agar with nalidixic acid and vancomycin as a Gram-negative selective medium. Cultures were observed at 2 and 5 days, and longer in some cases.

Histological examinations. Tissues collected for histological examination from post-mortem examinations conducted in Scotland were routinely processed and examined at SAC Edinburgh or the Moredun Institute of Neuropathology. Tissues collected for histological examination from post-mortem examinations conducted in England and Wales were sent to commercial histological laboratories for routine processing. Dr. John Baker conducted histological examinations of tissues collected during post-mortem examinations conducted at Veterinary Field Station (University of Liverpool) and all other cases from England and Wales were examined at IoZ. All standard formalin-fixed tissue samples collected from post-mortem examinations conducted under this contract were archived at IoZ or SAC.

Age and sex class determination
Age was determined mainly by quantification of growth-layer groups from analyses of decalcified tooth sections (Lockyer 1995a). Where data from teeth ageing was not available for sexually immature harbour porpoises, age was estimated from body length (see Lockyer 1995b) as follows: <105cm = 0yo; 105-120cm = 1yo; 121-130cm = 2yo, and >130cm = 3yo. For some data analyses, harbour porpoises were classified as neonates (body length ≤ 90cm), juveniles (sexually immature with a body length > 90cm), and adults (sexually mature). Sexual maturity was established using gonadal appearance and histological evidence of spermatogenesis in testes.

Toxicological analyses.
All tissue samples were stored at –20°C prior to preparation, and subsequently until analysed. Tissue samples were analysed for a range of trace elements, organochlorine pesticides, chlorobiphenyl and brominated diphenyl ether congeners according to previously established and fully validated protocols. The trace elements determined in liver samples were Cr, Ni, Cu, Zn, As, Se, Ag, Cd, Hg and Pb. Determinations were made using established methodology based upon nitric acid digestion with microwave heating followed (except for mercury) by analysis using

inductively-coupled plasma/mass spectrometry. Mercury was analysed using either atomic fluorescence detection or cold-vapour atomic absorption spectrophotometry, following reduction with tin (II) chloride (Jones & Laslett, 1994).

Blubber samples were analysed for organochlorine pesticides and metabolites, and a range of 25 chlorobiphenyl (CB) congeners, using established methodology based upon analysis by gas chromatography with electron-capture detection (Allchin *et al.*, 1989). This method was modified in the light of the recommendations which followed the intercomparison programme organised under the auspices of the International Council for the Exploration of the Sea (de Boer *et al.*, 1992, 1994; de Boer & van der Meer, 1998), and has been further validated for the analysis of biological tissues by participation in the QUASIMEME laboratory proficiency scheme (de Boer & Wells, 1997). The blubber concentrations of 25 individual CB congeners, expressed as mg kg^{-1} wet weight, were measured using standardised methodology (Allchin *et al.* 1989; Law 1994, Jepson *et al* 2005). The sum of the concentrations of the 25 CB congeners determined (Σ25CBs) were then converted to a lipid basis (mg kg^{-1} lipid) using the proportion of hexane extractable lipid (%HEL) in individual blubber samples. The individual (IUPAC) CB congeners analysed were numbers: 18, 28, 31, 44, 47, 49, 52, 66, 101, 105, 110, 118, 128, 138, 141, 149, 151, 153, 156, 158, 170, 180, 183, 187, 194. For the calculation of total PCBs (based on the *Aroclor 1254* formulation), fish of six species (cod, dab, flounder, lemon sole, plaice and whiting) (n = 118) were analysed on both a congener basis (using the ICES7 congeners CB28, CB52, CB101, CB118, CB138, CB153, and CB180) and on a formulation basis as *Aroclor 1254*. The two sets of data were then plotted and the regression established. The resultant conversion factor of 3 [total PCB concentration (as *Aroclor 1254*) = 3.0 x sum of ICES7 congeners (lipid wt)] was determined with a +/- 5% standard error. Since the CB profiles in fish and marine mammals are similar, being dominated by the recalcitrant congeners CB138, CB153 and CB180, this conversion factor (from ICES7 congeners to total PCBs) can be applied to harbour porpoises as well as fish (CEFAS, unpublished data). Regression analysis of sum 25CBs on ICES7 CBs for all cetaceans analysed 2000-2003 (*n*=199) gives a conversion factor of 1.53 with a standard error of 0.0036 (0.23%).

The brominated diphenyl ether (BDE) congeners (10 to 14 in number as the programme developed for these contaminants) were determined in the same blubber extracts as were used for the organochlorine compounds by means of gas chromatography/electron-capture negative ion mass spectrometry, monitoring bromine ions at 79 and 81 Daltons (de Boer *et al.*, 2001). All analyses were conducted under an analytical quality protocol requiring the analysis of blanks and reference materials alongside each batch of samples. Further details of method performance are given elsewhere (Law *et al.*, 1991; Law, 1994; Law *et al.*, 1997).

For statistical analysis of toxicology data, blubber *p, p'*-DDE/ΣDDTs ratio (DDE/tDDTs) and the hepatic molar Hg:Se ratio were calculated. Two quantitative indices of nutritional status (relative body-weight and mean blubber thickness) were calculated from the residuals from the best-fit regression line of log(body-weight) on log(body-length), and from the mean of the three standard (dorsal, lateral, ventral) blubber thickness measurements, respectively (Law 1994). The stranding locations were divided into the following areas to test for regional variation in all contaminant levels: South-west England (Devon, Cornwall, Somerset and Gloucestershire); East

coast of England (Northumberland to Essex); Wales and North-west England; the English Channel (Kent to Dorset); and Scotland.

Analysis of variance and regression models (Sokal & Rohlf 1995, SYSTAT 1998) were used to test for associations between levels of Σ25CBs, ΣDDTs, dieldrin, α-HCH, γ-HCH, HCB, Σ15BDEs (mg kg^{-1} lipid), Ag, As, Cd, Cr, Cu, Fe, Ni, Pb, Se, Zn, ΣBTs (mg kg^{-1} wet wt) and hepatic molar Hg:Se ratio (the dependent variables in the models) and the following (independent) parameters: age, sex, nutritional status, region of stranding and year of stranding. Intercorrelations between selected individual contaminants or contaminant groups were also investigated using similar statistical methods. For logistical reasons the quantity of data for each contaminant differed, and complete data were not available for all variables (e.g. age). Therefore, sample sizes for the statistical tests differ. Statistical analyses were performed using the natural logarithms (ln) of all continuously distributed data to meet the assumption of normality required by the statistical models (Sokal & Rohlf 1995). Statistical analyses of subsets of these porpoises were also examined where deemed appropriate to investigate further any statistically significant associations.

Tissue archiving. Tissue specimens collected for research and archive are stored at both -20°C and -80°C and in 10% neutral buffered formalin or 70% alcohol at the Institute of Zoology and SAC Inverness or sent to collaborating institutions for research purposes.

Tissue archive and Poseidon central database

The national marine mammal tissue archive and 'Poseidon' central database were developed under previous contracts to IoZ for marine mammal strandings investigation. The tissue archive provides an extensive range of tissues for retrospective analyses and currently over 50,000 individual samples have been collected during the course of the project, the majority of which are housed at the two main sites (IoZ and SAC). The type, reference number and location of each sample are tracked through Poseidon, a relational database specifically developed for this purpose. Poseidon is currently run on a Microsoft FoxPro 6.0 platform, although it is in the process of being transferred to a bespoke Access pathological database. Poseidon also holds records on over 13,500 individual strandings of cetacea, pinnipeds and recently marine turtles, of which nearly 2,800 have been investigated at post mortem. The central database was also established to ensure that the information collected by collaborating laboratories and individuals is recorded centrally, accurately, securely and in a standard manner. Most of the data currently held in the Poseidon database was generated under contract to the UK Government (Defra), although some data were generated independently by research collaborations with non-Defra-funded individuals or organisations. The IoZ also maintains the records for all seal species stranded throughout the UK, allocating seal stranding numbers as required and monitoring any unusual patterns of mortality.

Reporting of data

Data resulting from this contract has been routinely reported to the Department through annual contract progress reports. Reports have also been submitted for the UK national reports to ASCOBANS. Annual data on numbers of strandings of each cetacean species examined, together with the main causes of death, have also regularly been

submitted to the Department's Digest of Environmental Statistics (Wildlife Chapter). Occasionally, data have also been submitted to the Department for the purposes of answering Parliamentary Questions. A number of research papers have also been published within the scientific literature (see section **Annex D**).

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ANNEX C

C-1 BALEEN WHALES (BALAENOPTERIDAE)

Minke whale (*Balaenoptera acutorostrata*)

Spatial and Temporal Distribution of UK Strandings
The minke whale is the smallest but most commonly encountered baleen whale in UK waters and has on average ranked as the fifth most commonly stranded species from the entire NHM strandings record. It ranked fifth in the 1990-1994 period with 39 reported strandings but both its ranking and absolute numbers have increased (to third and 83 animals, respectively) in the 5-year period under review. Sheldrick *et al.* (1991) noted that strandings and sightings were recorded most frequently from northern Britain including the northern North Sea, although it occurred all round the UK. This distribution pattern around the north of the UK with a few records from the North Sea coast is consistent with recent years. There were 19 reported strandings in 2000, the highest annual figure on record, and 2002 and 2003 each had 18 records.
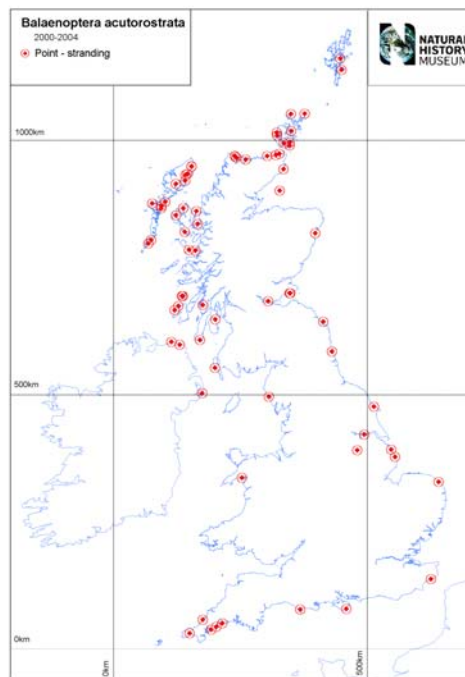
Figure 1: Distribution of minke whales reported stranded from 2000-2004.

Table 1. The monthly distribution of the minke whale for 2000-2004.

J	F	M	A	M	J	J	A	S	O	N	D
10	3	1	4	7	14	9	8	8	8	6	5

Sei whale (*Balaenoptera borealis*)

In September 2001 a sei whale stranded alive on the Lancashire coast and later died. It was one of only 13 recorded in the UK since 1913 and was the first to be reported since1990.

Fin whale (*Balaenoptera physalus*)

<u>Spatial and Temporal Distribution of UK Strandings</u>
The fin whale has also shown evidence of a slow increase since the 1980s with the highest number of strandings (n=11) occurring within the five years since 2000. Few had been reported since the 1940s and the current frequency has only appeared in earlier records between 1915 and 1925 (**Figures 2a and 2b**).

This species of whale occurs sporadically in strandings, usually with only one or two records per year, but in 2000 there were 4 and in 2004 another 7 reports were received. The total NHM database record for stranded fin whales amounts to only 61 animals. Historically most were found between 1915 and 1925 with another 7 records in the 1940's, but with very low instances at other times until the 1980's when strandings began to reappear.

Harmer (1927) noted that the early records up to 1926 were evenly distributed all around UK and Eire coasts and included the North Sea with individuals probably passing through the Straits of Dover. This pattern of distribution has persisted in more recent years with reports ranging from Kent to the Shetland Isles (**Figure 2a**). **Table 2** shows the seasonal distribution of fin whale strandings from 1980-2004.
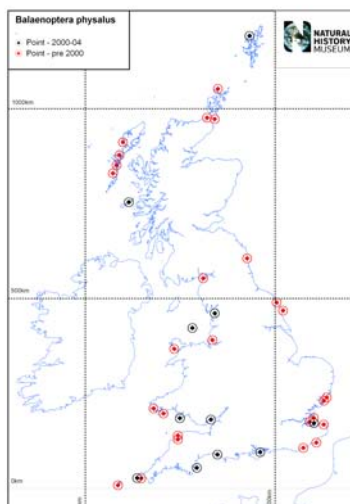
Figure 2a: All UK records of stranded fin whales 1913 2004
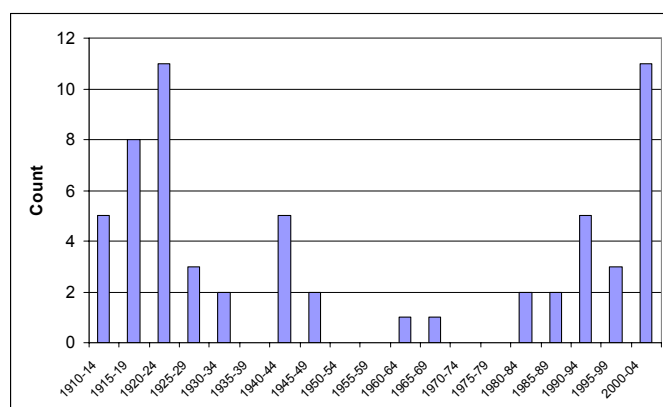
Figure 2b. Frequency of fin whale strandings in the UK 1910-2004

Table 2. Monthly distribution of fin whales (1980-2004)

J	F	M	A	M	J	J	A	S	O	N	D
2	2	0	0	0	0	4	2	5	5	2	1

Other data

Although it was not possible to establish the sex of many of the carcases, there has been a notable shift in the gender distribution for those animals whose sex was determined. In the early part of the record 70% were males, but since 1981 only three males have been recorded while the remaining 77 % were female.

Humpback whale (*Megaptera novaeangliae*)

Spatial and Temporal Distribution of UK Strandings

Among the more unusual species in the British record, the humpback whale (*Megaptera novaeangliae*) was reported on five occasions in the last five years, the only previous records having been two single strandings in the 1980's. Reports suggest that humpback whales tend to occur more frequently in northern British coastal waters, but in March 2001 the first stranding on the English coastline was recorded at Pegwell in Sandwich Bay, Kent. This live-stranded humpback whale was the most southerly record of stranding of this species in the UK. On 13th September 2004 the first record for Northern Ireland was received from the Giant's Causeway, Co. Antrim, as well as another from the Orkney Isles, Scotland in February of that year.

Other data

Unfortunately the sex and length data were not available for either of the animals stranded in 2004. Examination of the testes from the Kent stranding revealed that this male was immature and put its age at between one and three years. Based on their recorded lengths, both of the animals stranded in the 1980's were very young individuals (**Table 3**). Studies by Winn and Reichley (1985) suggest that the average length at birth is 4-5m, and the mean length at sexual maturity is 11.58m for males and 12.09m for females.

Table 3. Length data for all UK records of stranded humpback whales.

SWNo	Date	County	Sex	Metres
1982/38	16 Oct 1982	South Glamorgan, Wales	M	8.66
1985/42	23 Aug 1985	Highland, Scotland	F	5.99
2001/60	21 Mar 2001	Kent	M	10.22
2001/205b	07 Sep 2001	Highland, Scotland	-	-
2003/98b	15 Feb 2003	Lothian, Scotland	-	13

	27	Feb			
2004/91d	2004		Orkney Islands, Scotland	-	-
	13	Sep			
2004/274b	2004		Antrim, N.Ireland	-	-

The monthly occurrences of humpback whales in British coastal waters have been in February-March and August-October which suggests that a seasonal migration may take place in spring and autumn when these whales can be found off the UK. Their rarity in British and Irish waters and long absence from the NHM's strandings records until comparatively recently, may be indicative of a recovery in humpback numbers.

C-2 DOLPHINS (DELPHINIDAE)

Short-beaked common dolphin (*Delphinus delphis*)

Spatial and Temporal Distribution of UK Strandings
The spatial distribution of common dolphin stranding reports between January 2000 and December 2004 inclusive is shown in **Figure 3a**. Each year reports of dolphin carcasses which are too decomposed, incomplete or inaccessible for examination and retrieval are received. Numbers of these reports have risen in parallel with those for common dolphins (**Figure 3b**) and as their geographical distributions are so similar, it is highly likely that most unidentified carcasses in the south-west of Britain were those of common dolphins.
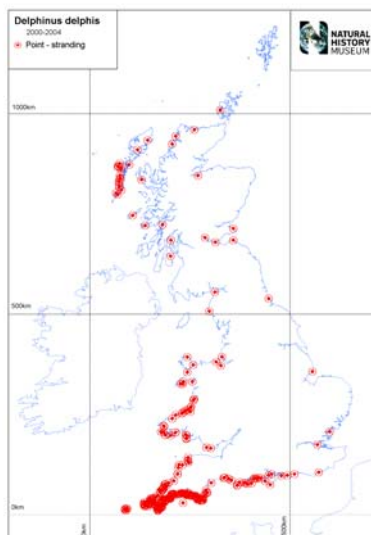
Figure 3a: Distribution of UK-stranded common dolphins reported (2000-2004)
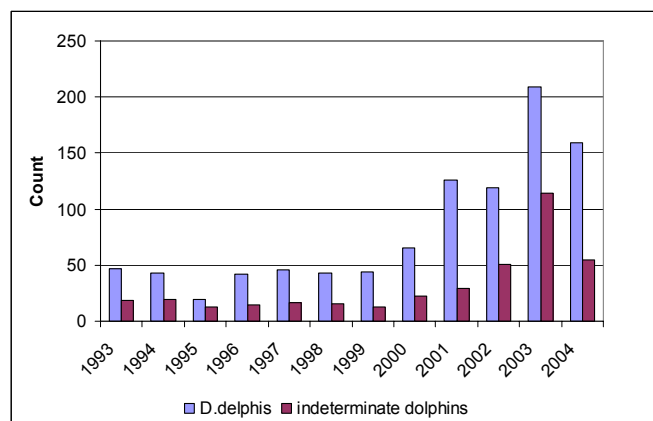
Figure 3b. Frequency of strandings common dolphin vs indeterminate dolphins 1993-2004

Figure 4 shows that common dolphin strandings averaged only two or three per year until the 1970's, before reports rose steadily to around a dozen annually by the end of the 1980's. Although numbers were low in the early records this species contributed around 8% of all strandings, but a progressive decline from 1913-1966 was noted by Fraser in 1974 with a proportion of only about 3% for the latter part of this period (**Figure 5**). When reporting effort increased in 1990, annual reported strandings rose to around 40 for that decade. For the last five years, from 2000-2004 that figure has risen to an average of 136, agsin possibly due to increased reporting effort. The number of common dolphin strandings reported peaked at 209 in 2003.

Table 4 shows that the seasonal distribution for common dolphins has changed with time. For example records from 1913 to 1966 show high numbers in February and similar figures in August and September. From 1967 to 1989 the highest numbers occurred between October and March, with a peak in January. Over the next 15 years the majority of strandings were in the first three or four months of each year with peak numbers recorded in February, January and then March, respectively for each of the successive five-year intervals. But low records from May to July have always been a feature and this has long been associated with the departure of common dolphins from our shores in early summer for the reproductive season.

Monthly figures for the last 15 years suggest that the period with a lower incidence of stranding now extends later into the year. This may indicate that the dolphins are remaining offshore for longer. Movements of common dolphin populations are likely to be linked to the offshore migration of mackerel to their spawning grounds in spring and early summer (see below). Live sightings reports from these areas are needed to substantiate this suggested pattern of behaviour.
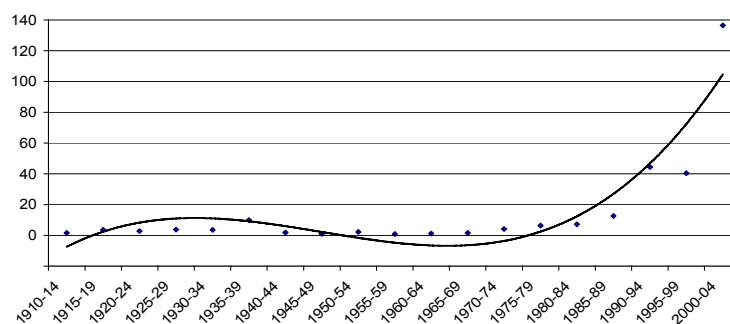
Figure 4. Annual average number of reported common dolphin strandings per 5-year period (1910-2004)
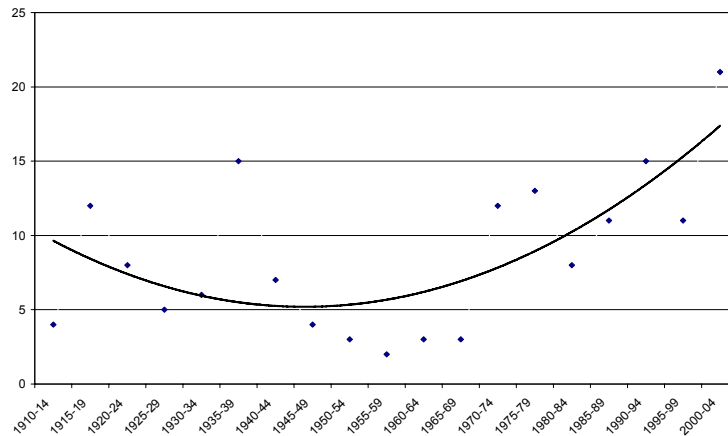
Figure 5. Percentages of common dolphin per 5 year periods (1910-2004)

Table 4. Comparison of monthly frequencies of common dolphin strandings expressed as relative percentages of each of five periods spanning the total dataset. The peak figures for each period are emboldened.

	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
1913-66	4	**15**	9	8	7	4	7	**15**	**14**	7	4	5
1967-89	**16**	12	**11**	7	2	7	3	5	5	**10**	**11**	**10**
1990-94	12	**23**	20	**10**	5	0	3	4	6	5	6	6
1995-99	**21**	17	17	12	3	6	5	3	3	3	5	5
2000-04	**23**	11	**27**	11	3	1	3	2	1	2	6	**10**

Scottish strandings

MacLeod *et al*. (2005) have suggested that, for NW Scotland, there has been a relative increase in sightings and strandings of common dolphins (whose range extends south and westwards) since 1981 compared with a decline in reports of white-beaked dolphins (*Lagenorhynchus albirostris*), whose range extends north and eastwards in the same region.

Monthly strandings figures show that common dolphins occur in Scottish waters all year and that most are found in the winter months, peaking around February and March (**Table 5**). Prior to 1990 there was a distinct maximum in February, but in the last 15 years the highest numbers have occurred a month later.

In England and Wales the lowest number of strandings, which coincide with a presumed offshore migration, occurs from May to July.

Table 5. **Monthly frequency of common dolphin strandings recorded in Scotland**.

	J	F	M	A	M	J	J	A	S	O	N	D
1913-89	2	**21**	1	4	8	5	0	1	2	1	3	6
1990-94	3	3	**6**	2	1	0	0	0	0	1	1	1
1995-99	3	4	**6**	3	5	6	2	1	3	2	2	2
2000-04	3	5	**7**	2	4	2	3	0	3	2	1	5

(peak numbers are emboldened)

Common dolphin/striped dolphin

The shape and body size of these two species are very similar and if decomposition or other damage has led to the loss of identifiable external characteristics, it can be difficult to distinguish the two. There are two well-defined palatal grooves on the ventral surface of the rostrum of common dolphins which can be identified by touch, but it is not always possible to obtain information on their presence or absence.

Numbers vary from year to year depending on the condition and accessibility of specimens, but the overall number of strandings (21) from 2000-2004 is similar to the previous five-year period when 25 animals were recorded under this category.

Long-finned pilot whale (*Globicephala melas*)

Spatial and Temporal Distribution of UK Strandings
The presence of pilot whales is usually recorded annually, but this species has occurred in widely fluctuating numbers as a result of its propensity to occasionally strand in massive groups. In 1950 two such events occurred involving 97 and 148 animals respectively. This variability in stranding has led to similar changes in its rank in abundance. It has often been the most frequently stranded species of cetacean on the British coastline, exceeding the number of porpoises in those years. But in recent years there have been no recorded instances of mass stranding involving more than two pilot whales. The most recent event involving more than two animals was in April 1992 when seven whales stranded together, and the last sizeable mass event occurred in January 1985.

In the species rankings for the last three five-year periods pilot whales have been third twice and fourth for 2000-2004 with 87, 78 and 75 animals recorded,

respectively. During this fifteen year period harbour porpoises and common dolphins were more abundant and in the five-year period of review there were also more minke whales stranded than pilot whales.

In the winter of 2001 three dead pilot whales were seen floating in the same area as 16 common dolphin carcases off Looe in Cornwall. This suggested that they may be prone to by-catch from the same seasonal fishery that affects large numbers of dolphins in the south-west. Three more pilot whale carcases were found at sea off Cornwall the following year.

Only one stranding of this species has been recorded in the North Sea in the last five-year period and there are indications that it has occurred there less regularly recently than, for example, during the period from 1948 to 1966.

Risso's dolphin (*Grampus griseus*)

Spatial and Temporal Distribution of UK Strandings
The NHM database holds 112 records of Risso's dolphin strandings between 1913 and 1989, an average of just over one animal per year. Data for the five years from 2000 to 2004 clearly show a west coast distribution with the highest concentrations in Scottish waters (**Figure 6**). Since 1992, Scottish strandings have accounted for 76% of all British records. The most recent five years have been consistent with this, both in terms of distribution and frequency with fluctuations in numbers from year to year within the expected range.

Earlier NHM database entries do show sporadic occurrences of Risso's dolphins in the English Channel and in the northern part of the North Sea, but there have been no North Sea records since a stranding at Lossiemouth in 1992. With the exception of a solitary stranding at Cleethorpes in Lincolnshire in 1933, the other North Sea strandings have all been restricted to the east coast of Scotland, mainly on the most northerly shores.
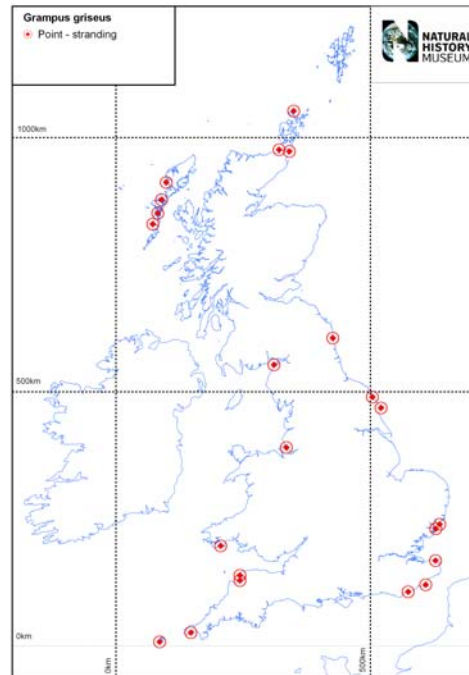
Figure 6: All UK stranding records of Risso's dolphin 2000-2004

Killer whale (*Orcinus orca*)

<u>Spatial and Temporal Distribution of UK Strandings</u>
Although sightings data show killer whales in UK waters throughout the year (Reid *et al.* 2003), strandings of killer whales are not reported every year on UK shores. The NMH database holds stranding records of 85 killer whales for the period from 1913 to 2004 which represents an average of just less than one per year. In the period under review three strandings were reported; one each in 2001, 2002 and 2003. Most strandings occur in the north and west of Scotland, but this species has been found around the entire coastline. The three most recent strandings were all recorded in England. The animal reported in 2002 stranded on the Humberside coast in December but managed to free itself on the rising tide. Its sex was unknown and the length was estimated at about 6 metres. The 2001 animal, an adult male, also stranded alive but later died on a sandbank in the Mersey estuary in October. The third animal was reported in May 2003 by yachtsmen off Cumbria where it was seen floating about three miles from the shore. This too was a male, judged to be about 5 metres in length. These length data suggest that all three animals were adult. The NHM records show that about a third of killer whale strandings are very young animals. Also, from the strandings data available, there appear to be more males (29 strandings) than females (17 strandings) in UK waters. Live sightings and stranding records both indicate that killer whales occur off British coasts throughout the year.
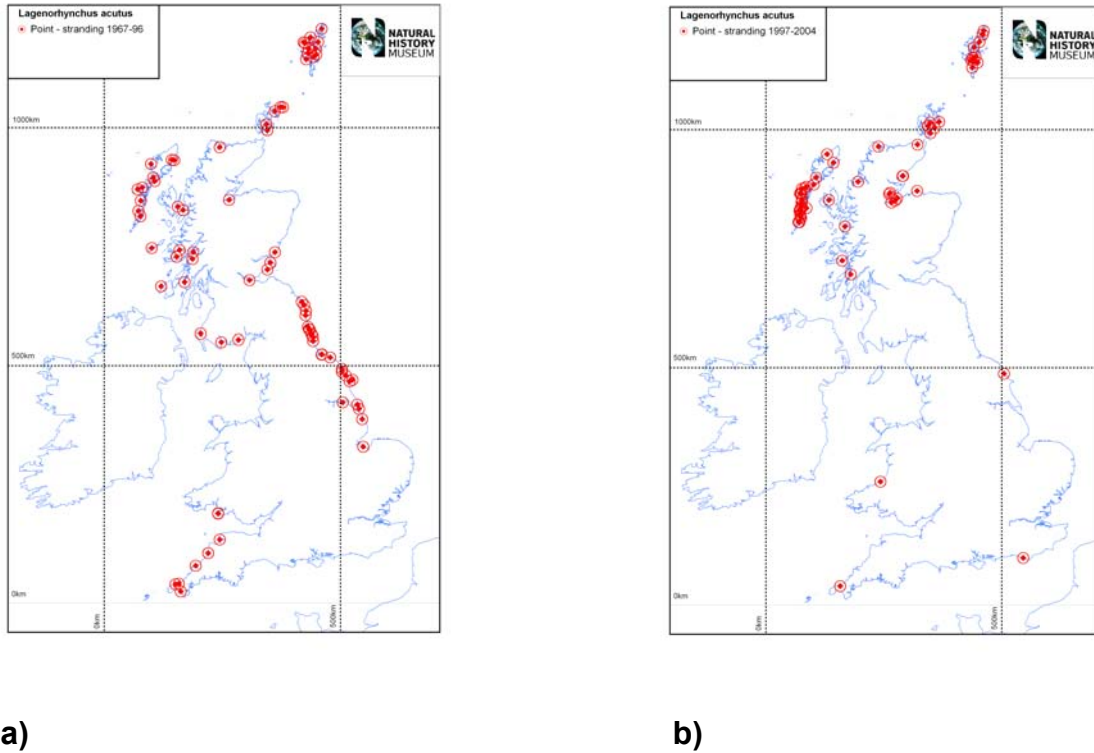
Atlantic white-sided dolphin (*Lagenorhynchus acutus*)

Spatial and Temporal Distribution of UK Strandings
Lagenorhynchus acutus is a boreal species restricted in distribution to the northern North Atlantic. This species is gregarious, and mass stranding (an event involving two or more animals at the same time and place) is not uncommon. In the 1920s, three separate events, all in the Shetland Isles, each involved over 30 animals. But since then the reports of mass strandings have all been of much smaller groups. In the last five years three mass strandings were recorded, involving six, five and two animals, respectively.

Annual stranding totals have fluctuated over the course of our historical record, partly as a consequence of mass stranding events in some years. For most of the period from 1913 the average figure has been about two reports per year for this species. But as with many of the smaller cetaceans, the Scottish records increased from 1992 with the establishment of a systematic reporting programme. As a result, the average has risen to about nine animals over the last 15 years. But notwithstanding the increased reporting effort, there has been a marked change in the ratio of the two species of *Lagenorhynchus* (Atlantic white-sided dolphin and white beaked dolphin) in British waters in recent years, based on the evidence from strandings data. The large mass stranding events in the 1920s, noted above, affected the ranking order of species at that time, but for most of the period from 1913 to 1966 stranding reports of white-sided dolphins was only about 30% of those for white-beaked dolphins in the UK. The NHM strandings record from 1967 to 1989 gives a comparison of 60%, but this has risen still further to 85% in the last 15 years from 1990 to 2004. Indeed for the last ten years of this period the numbers of each of the two species have been more-or-less the same although in some years, reports of white-sided dolphins have outnumbered white-beaked dolphins.

Between 1913 and 1966 approximately a third of records were on the English east coast of the North Sea and the rest mainly around the Scottish northern isles. **Figure 7a** shows that a similar proportion occurred in the North Sea from 1967 to 1996, with the remainder more-or-less evenly distributed between south-west England and Wales and four main areas of Scotland, namely Shetland, Orkney and the Western Isles and the mainland west coasts of Strathclyde and Highland. The number of strandings in the south-west of the British Isles in this period was unusual; there have been only sporadic records at other times and it is noteworthy that the only significant numbers of strandings of white-sided dolphins in the region were also reported within the same period by Sheldrick *et al.* (1991). Comparison of **Figures 7a and 7b** shows that strandings on the Scottish west coast had a more southerly distribution at that time than more recent records indicate. From 1967 to 1996 ten records were from the regions of Dumfries and Galloway and Strathclyde and only four from the more northerly region of Highland. From 1997 to 2004 two strandings occurred in Strathclyde while 18 were reported from Highland. At the same time a solitary record in Yorkshire was the only recent evidence from strandings that this species still occurs in the North Sea. There have been no further reports since that record in May 1998. **Figure 7b**, for the period 1997-2004, illustrates the recent concentration of strandings on offshore islands with the predominance in the Western Isles.

Monthly stranding frequencies of Atlantic white-sided figures show little variation throughout the year (**Table 6**).

a)   b)

Figure 7. **Distribution of Atlantic white-sided dolphin stranding reports showing changes between a) 1967-96; and b) 1997-2004**

Table 6. Monthly frequencies of Atlantic white-sided dolphins for the period 1913-2004.

J	F	M	A	M	J	J	A	S	O	N	D
18	20	21	21	18	11	20	17	16	20	14	9

White beaked dolphin (*Lagenorhynchus albirostris*)

Spatial and Temporal Distribution of UK Strandings
White beaked dolphins occur in the northern North Atlantic in cold temperate and subarctic waters. A population is believed to exist off the west coast of Ireland (Reid *et al*. 2003) which may account for occasional strandings in the south-west of England, but most of the British strandings have occurred on the east coasts of England and Scotland.

In a status review of marine mammals in the southern North Sea, Klinowska (1987) described the white-beaked dolphin as the third most commonly reported species in

North Sea sightings and strandings records. Its position in the entire British strandings record has varied with time from as low as tenth, at around the time of the Second World War, to third in the late 1960s. During much of the last 30 year period it has ranked about fifth in the UK but with a drop in position over the last decade. Although there have been substantially more records in the last fifteen years, largely as a result of improved reporting procedures in Scotland since 1992, the number of reports has declined since the start of this period. Since 1991 numbers have fluctuated between 5 and 19 per year with an annual average of 9 over the last 5 years.

But the most important trend, which has become more apparent since about 2000, is the change in the geographical distribution of strandings (see **Figures 8a and 8b**). Describing the earliest British stranding records from 1913 to 1926, Harmer (1927) reported that most strandings of white beaked dolphin were from the North Sea where they occurred in all months, but least commonly in December and January. In the southern North Sea strandings usually occurred in early spring (February-April) or late autumn (October-November). A greater concentration in June-August from Yorkshire to the extreme north of Scotland suggested a migration northwards in summer. In subsequent reviews Fraser (1934, 1946 and 1953) described east coast strandings that were in accordance with the earlier period. For the period 1948 to 1966 the distribution still conformed to the previously established pattern, with most strandings of this species on the east coast as far south as London (Fraser, 1974). Fewer, but a still significant number, were reported from the west coast, as in previous years, with more sporadic occurrences in the English Channel. It was noteworthy that in that period the low number of strandings in June and August and the lack of records from July was in contrast to earlier years.

Sheldrick *et al.* (1991) reported numbers as high in January-March as in July-September for the period from 1967 to 1990. All July and August strandings were north of a line between the Humber and Mersey, providing further evidence of a summer migration to the north. Sheldrick *et al.* (*ibid.*) also noted that more strandings occurred in the south-west of England.

An increase in reporting effort in Scotland from 1992 almost certainly gave rise to an increase in the number of white-beaked dolphins as more of the smaller cetaceans were being recorded. Some were found in north-west Scotland that year, but the bulk of strandings still occurred on the east coast from Norfolk to the north-east of England and Scotland. But at the same time more regular strandings began to occur in the northern island groups of Orkney and Shetland. There was an absence of records from the southern North Sea from 1997-1999 when Lincolnshire was the southernmost limit of strandings (**Figure 8b**). In 2000 there were further indications that the distribution of this species had moved northwards with all reports coming from Orkney, Shetland or the Highland region. In 2001, again all occurrences of this species stranding were in Scotland, including a mass stranding event in the Shetland Isles involving eight animals.

In 2002 and 2003 the majority of reports were Scottish, but records from the east coast indicated that this species was still present in the most northerly part of the North Sea. However, there were no records from the south. Indeed the southernmost records, both from Northumberland, were significantly further north than those in the

latter part of the 1990s which occurred in Lincolnshire. The Northumberland strandings were found at the end of September and in April, respectively. The other four records in 2003, all from Scotland were in January (2), October and December, providing further evidence that the main population was remaining further north in winter. The 2004 stranding record showed seven Scottish occurrences and three on the north-east coast of England, the southernmost animal stranding at Spurn Head in Yorkshire. So for the last eight years there have not been any reported strandings south of Lincolnshire, and none since 1983 between Sizewell, in Suffolk, and the Thames estuary.

a)   b)
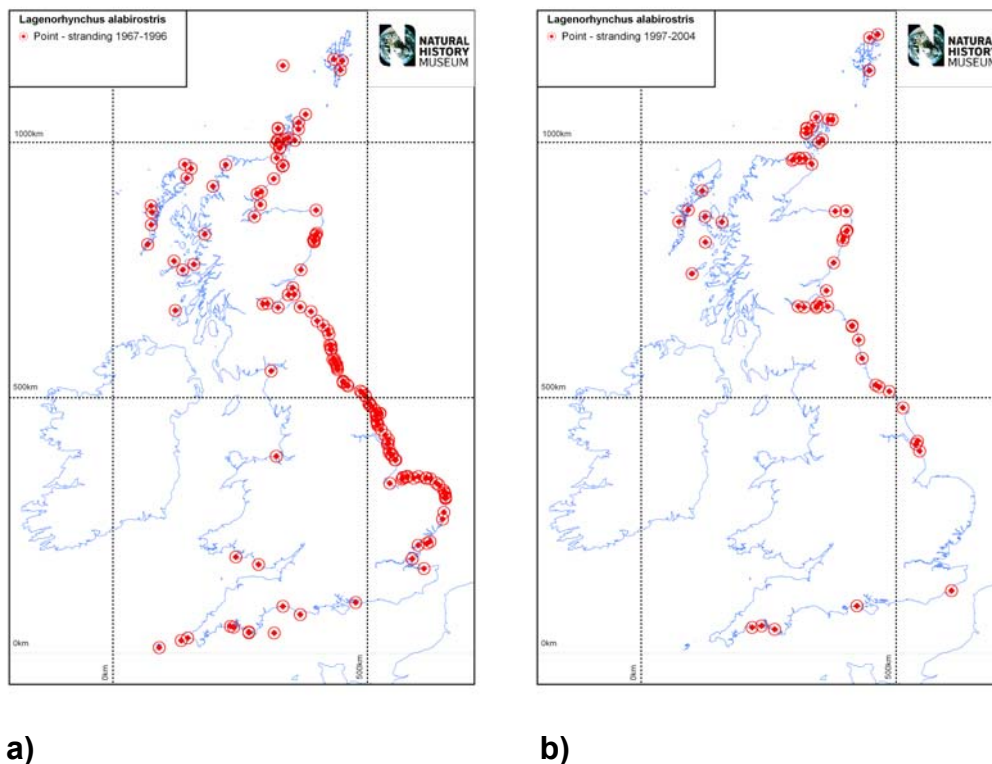
Figure 8. **Distribution of strandings of white-beaked dolphins showing changes between: a) 1967-96; and b) 1997-2004**

In a study on the possible effects of climate change on cetacean distribution in the north-west region of Scotland, MacLeod *et al.* (2005) cite this dolphin as one of the species that may show evidence for this. Certainly the indications from the NHM records from the last five years, and possibly in the period leading up to this, suggest that there has been a northward movement of the North Sea population when compared with historical data. The reason for this is not known and while a rise in surface water temperatures may have been a contributory factor, a change in availability of its prey species of fish is equally (and possibly more) likely.

***Lagenorhynchus* sp. indet.**

As with other groups of unidentified cetaceans it is usually an advanced state of decomposition which prevents identification to species level, and the numbers vary

accordingly. Seven unidentified *Lagenorhynchus* specimens were reported for 2000-2004 compared with six in the previous five years. Retrieval of the entire skull, or teeth alone, can be used to distinguish the two species. Although this is not always possible, one such skull specimen was obtained from the carcase of SW2000/161 which is yet to be examined.

Inferences regarding the more likely identity of indeterminate *Lagenorhynchus* specimens can be made based on the geographical location of the strandings. However, this may be unreliable because of the natural overlap in distribution of the two species which occur in British waters. There are also recent indications of changes to their distribution patterns around the UK which are discussed above.

Striped dolphin (*Stenella coeruleoalba*)

Spatial and Temporal Distribution of UK Strandings
The similarity in size and shape to common dolphins may have produced some confusion in the identification of these two species on occasion but any early rarity in the record of striped dolphin was not due to lack of awareness of the possibility of its being found amongst British strandings (Harmer, 1927). The striped dolphin has never been particularly abundant in the strandings records and indeed the first authenticated record for this species did not occur until 1923. There were three more single records in the 1930s but no more until a further two were found in the 1970s. Strandings were then found in most years of the next decade and since 1990 it has been reported regularly, in gradually increasing numbers. For the past ten years it has ranked as the sixth most frequently stranded cetacean on our coast.

The number of striped dolphins ranged between 6 and 14 per year between 2000 and 2004, with a total of 54 for the period. The spatial distribution of striped dolphin strandings between 2000 and 2004 is shown in **Figure 9**. The annual range and total was broadly similar to that of the previous five years. Their abundance has always been greatest in south-western waters, but Muir *et al.* (2000) reported a spread in distribution in the 1995-1999 quinquennium to include Scotland. MacLeod *et al.* (2005) considered its presence in north western Scottish waters, along with changing patterns for other species, to be possibly indicative of climate change although reports in the most recent five-year period from northern parts of Britain have declined. The only part of the country where the number of striped dolphin stranding reports has continued to rise is in the south-west (**Figure 10**).

In 1974 Fraser commented that the euphrosyne (striped) dolphin was abundant in warmer waters than those surrounding the British Isles and related the paucity of strandings on British shores at that time to their peripheral situation to the main populations. The seasonal distribution of striped dolphin reports for 1913-1989 and for 1990-2004 is shown in **Table 7**.

Table 7. Monthly frequency of striped dolphin strandings recorded around Britain.

	J	F	M	A	M	J	J	A	S	O	N	D

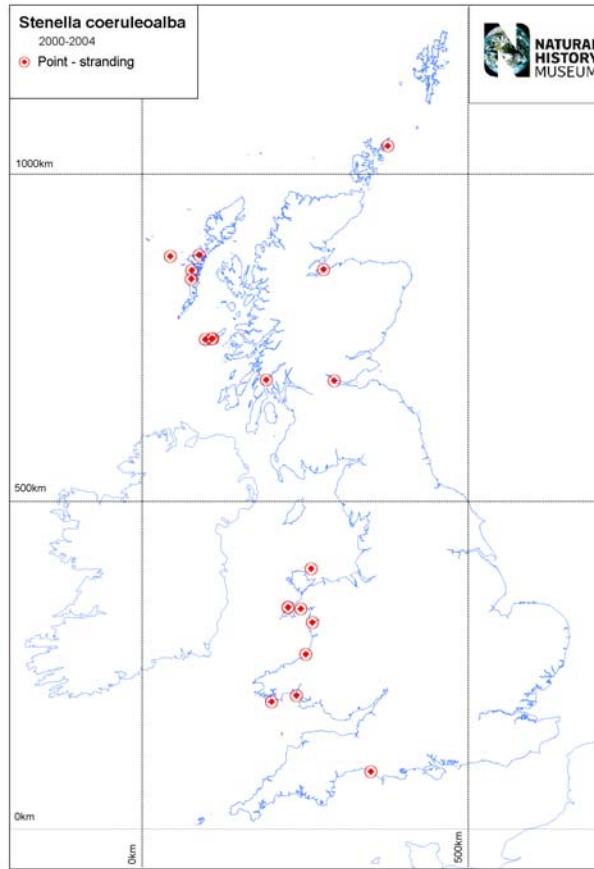| 1913-89	4	7	0	3	0	0	3	2	3	1	3	4
1990-04	19	27	11	9	6	6	10	16	15	5	9	12

Figure 9: Distribution of stranded striped dolphins reported (2000-2004)

Figure 10. **Regional trends in strandings of striped dolphins**. The North/South boundary for the regions occurs at 400km northing on the British

National Grid (BNG) (Mersey and Humber estuaries). The East/West boundary varies from 250km easting on the BNG along the northern coast of Scotland to 400km along the southern coast of England.

Bottlenose dolphin (*Tursiops truncatus*)

Spatial and Temporal Distribution of UK Strandings
The spatial distribution of UK-stranding reports of bottlenose dolphin is shown in **Figure 11**. This is a species whose abundance in the stranding record has changed significantly with time and one for which the geographical distribution of strandings has also undergone some marked changes. Broadly, bottlenose dolphins represented a greater proportion of all strandings in the earlier part of the record than they have been since the 1970's. Between the 1940's and the 1960's these dolphins were the second or third most commonly reported cetaceans but their ranking has dropped to tenth or eleventh over the past 15 years. From 1948 to 1966 when this dolphin was prominent among strandings it was reported in concentrations on the north-western British coast around the Irish Sea and in the south-east along the English Channel and East Anglian shores (Fraser, 1974). Apart from the extreme south-west coast of Scotland bordering the Irish Sea, there was only one other record for Scotland, from the Moray Firth. At that time, and during the previous ten years, the southern North Sea accounted for around a quarter of all strandings of this dolphin, with a comparative scarcity in the northern North Sea area.
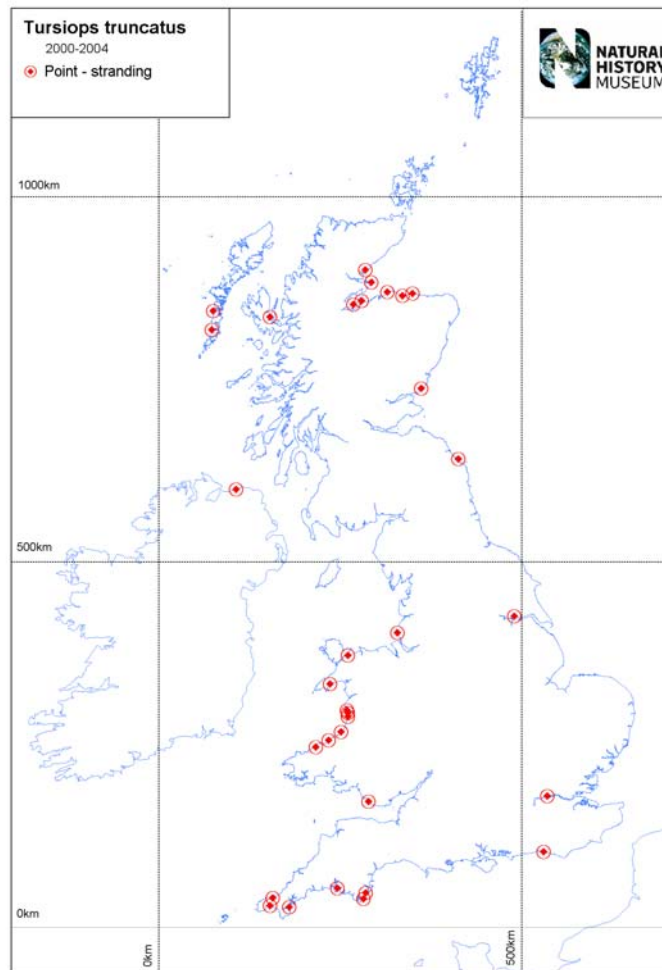
Figure 11: Distribution of UK-stranded bottlenose dolphin reports (2000-2004)

Figure 12 shows the geographical changes that have taken place from 1960 to 2004. There is a drop in reports from the north-west and a more gradual decline in the south-east. By contrast, the erratic increase in strandings in the south-west, including Wales, and the rise in reports from the north-east since 1990 respectively reflect the bottlenose dolphin populations that now appear to be concentrated in the Cardigan Bay and Moray Firth regions. The increase in the latter case may be due in part to improved reporting procedures for Scotland since 1992. This may also have led to a greater frequency of reports from the Hebridean region.
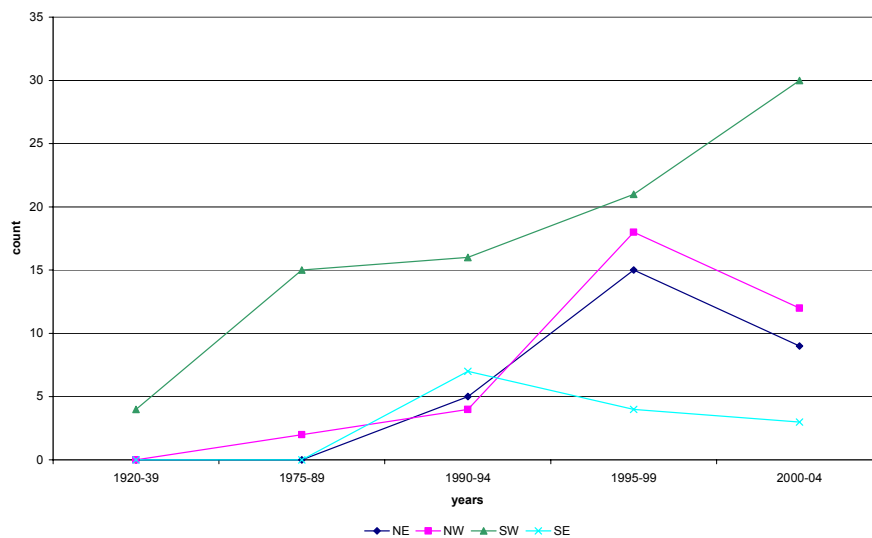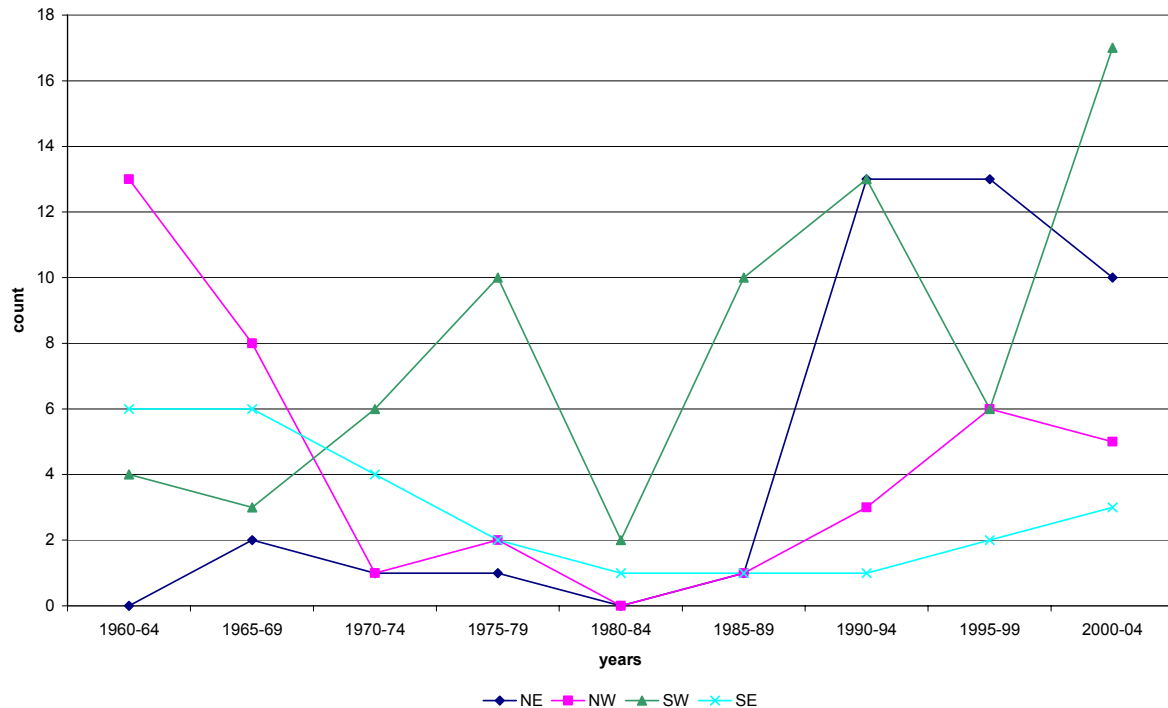
Figure 12. Regional trends in strandings of bottlenose dolphins. The North/South boundary for the regions occurs at 400km northing on the British National Grid (BNG) (Mersey and Humber estuaries). The East/West boundary varies from 250km easting on the BNG along the northern coast of Scotland to 400km along the southern coast of England.

Fraser (1974), for 1913-66, showed a peak in stranding frequency in the summer months of June (Jul-Aug) September with the same pattern having been maintained for 1948-66 as in earlier years. But Sheldrick *et al.* (1991) noted a reduction in summer prominence for the 1967-90 period. The monthly incidence since then has returned to the earlier pattern, with a peak around July and August, but with high occurrences also recorded in March and October (**Figure 13**). It is not yet clear whether these spring and autumn prominences reflect in any way the recent changes in geographical distribution of this species.
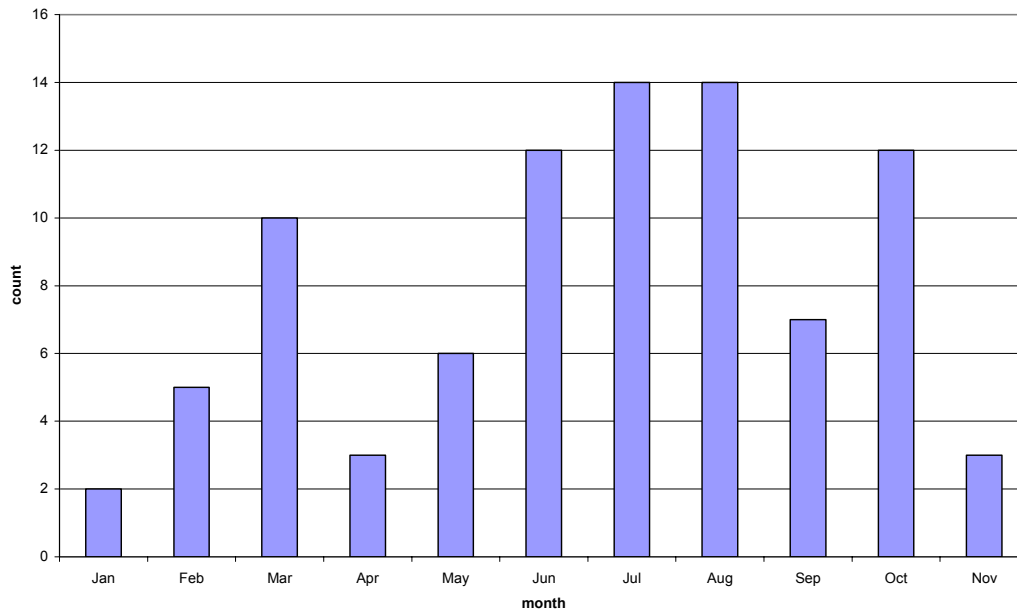
Figure 13. Monthly stranding of bottlenose dolphin for the years 1991-2004
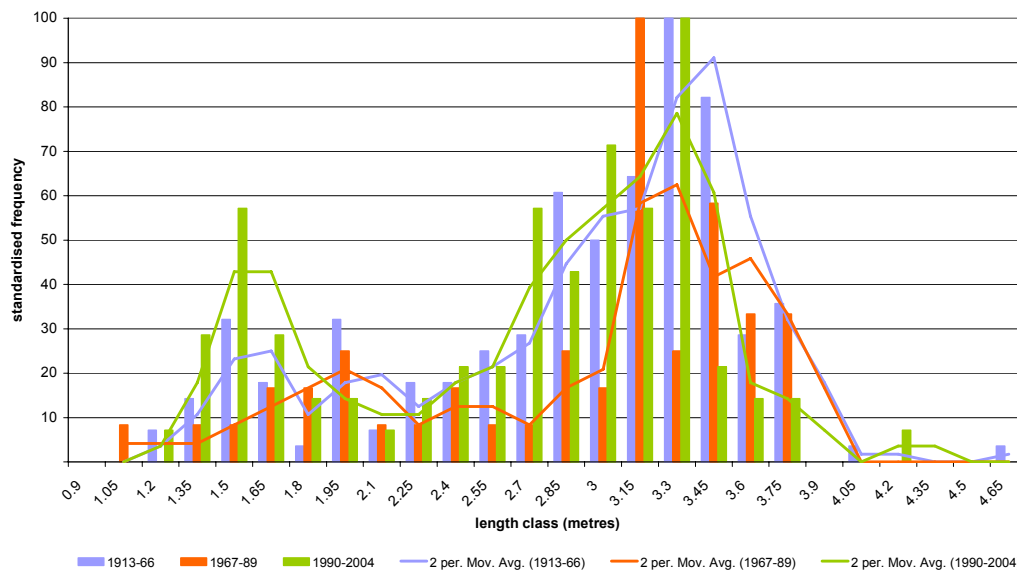
Figure 14. Length frequency classes of bottlenose dolphins

Other data

Fraser also illustrated the bimodal pattern produced by the length frequencies of stranded bottlenose dolphins. This is a pattern which is still clearly seen in later time periods (**Figure 14**). In these figures the frequencies have been adjusted in relation to the maximum variate for each period under review, to facilitate comparisons. The lower cluster in each case represents mortality of young animals while the peak in

the more prominent group probably indicates mortality due to old age. Klinowska (1991) described a degree of segregation within nearshore populations of bottlenose dolphins, based on sex and age. Subadult males usually form bachelor groups with one or two adult females, while the majority of females with calves tend to occupy the more productive areas of the community home range. The consistent paucity of immature animals in the British strandings data may reflect a tendency for immature dolphins to spend time away from the main breeding populations in UK waters. **Figure 14** indicates a higher proportion of juveniles in the period from 1990-2004.

Unidentified dolphins

The number of unidentified dolphins recorded by the NHM reached a high of 114 in 2003 when the number of common dolphin strandings was also at its peak. These reports relate to carcases too decomposed, incomplete or inaccessible for examination and retrieval. Most were found in south-west England between January and March, and many bore injuries consistent with entanglement in fishing gear (broken rostrum, lacerations, damage to fins, flukes and flippers etc.). It is highly likely that the carcases were those of common dolphins and the significance of these unidentified animals in relation to the possible total number of common dolphin strandings is indicated elsewhere.

It is noteworthy that a large number (102) of unidentified dolphins were recorded in 1992 when mass-mortality of common dolphins was correlated with a high incidence of fishery interaction. For the remainder of that decade the level of unidentified dolphins did not exceed 20 per year. But in 2000 the numbers began to rise in parallel with the number of common dolphin deaths. In 2001 and 2002 the distribution of unidentified stranded dolphins extended further east from the main area of concentration on the coasts of Devon and Cornwall. Stronger prevailing weather conditions in those years may have contributed to this spread. It is clear that carcases which have drifted for longer will arrive ashore in a more decomposed condition and be less readily identifiable. The high proportion of animals in this category compared with fresher identifiable common dolphin strandings found in the more eastern counties implies that they possibly died in south-western waters.

C-3 PORPOISES (PHOCOENIDAE)

Harbour porpoise (*Phocoena phocoena*)

Spatial and Temporal Distribution of UK Strandings
The spatial distribution of harbour porpoise strandings is shown in **Figure 15**. Harbour porpoises represent approximately half of all cetaceans that strand annually in the UK. In the period under review, the percentage has fluctuated more than usual with 59% in 2004 compared with only 41% in 2003 when many common dolphins stranded. When systematic recording was introduced in 1990, reports of harbour porpoise strandings increased steadily for a number of years, but from 1996 to 2000 the figure was remarkably stable (**Figure 1**). Since 2001 the number of reports of porpoise strandings has increased significantly from 259 to 474 in 2004. **Figure 16** gives a regional analysis of reports of strandings over the last 45 years and clearly

illustrates the extent of recent rises. The north-east region comprises much of the coast of England and Scotland bordering the North Sea. It is the only quadrant to show any decline and is in contrast to the continued steep increase in absolute numbers in the last five years in the other regions.
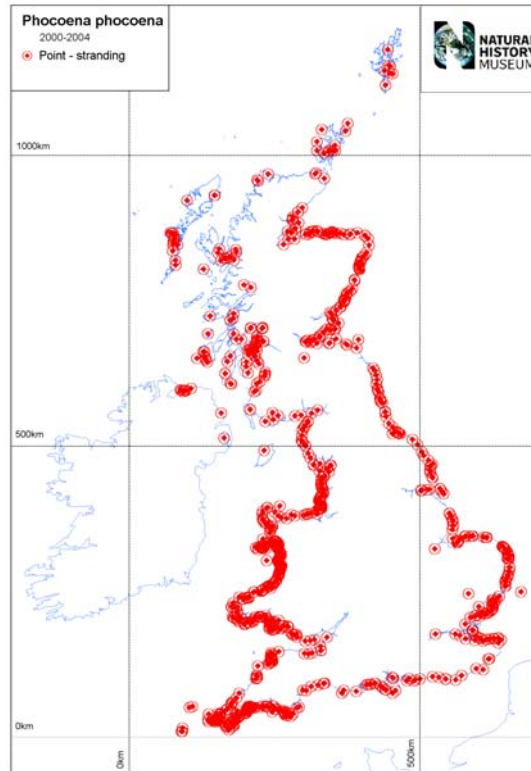
Figure 15: Distribution of UK-stranded harbour porpoise reports (2000-2004)
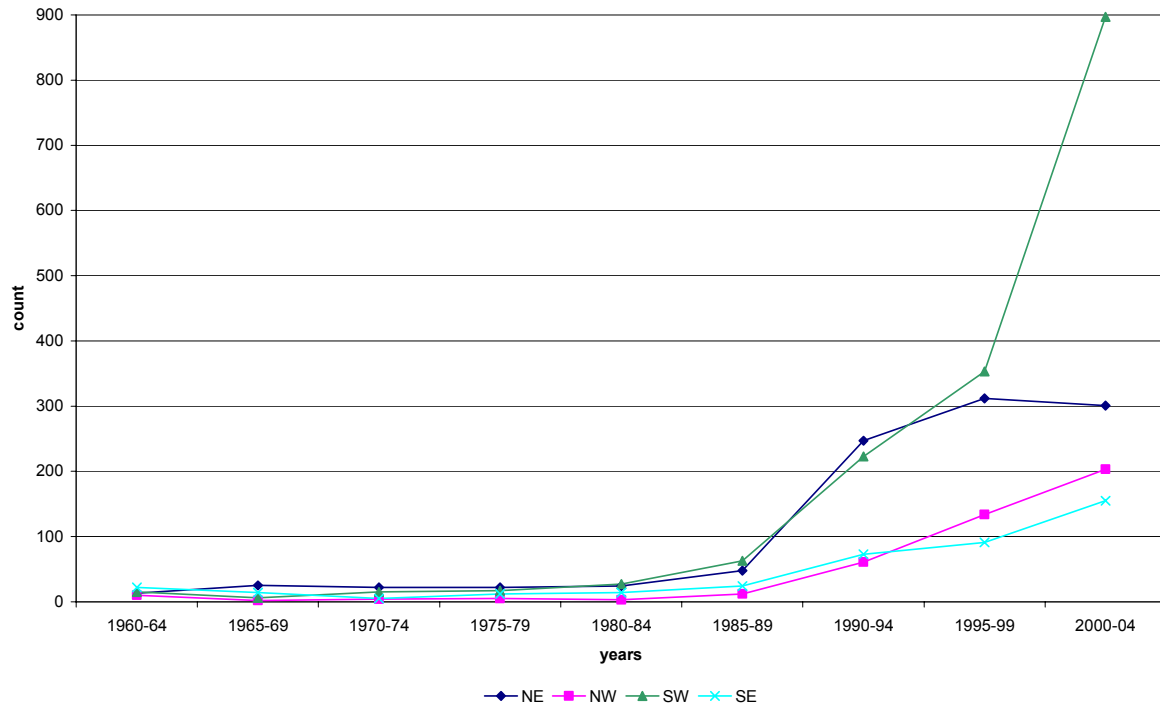
Figure 16: Regional trends in reports of strandings of harbour porpoises. The North/South boundary for the regions occurs at 400km northing on the British National Grid (BNG) (Mersey and Humber estuaries). The East/West boundary varies from 250km easting on the BNG along the northern coast of Scotland to 400km along the southern coast of England.

The most pronounced rise in stranding reports has been in the south-west of England where numbers have increased from a total of 353 for the five year period from 1995-1999 to 897 for 2000-2004. The reasons for this recent change and the longer term trend since the 1970's are not clear. More detailed examination of the data reveals a seasonal difference in the distribution of strandings between Wales and SW England. The Welsh reports occur mainly in the summer months while strandings in south-west England predominate in winter (**Figure 17**). The summer peak is thought to coincide with parturition in this species so the data indicate that this event in communities on the west coast of the UK is centred on Wales.

Although these data suggest a possible north-south seasonal migration between SW England and Wales, differences in ratios of body length to age have been detected between harbour porpoises from SW England and other UK regions (including Wales) (Jepson 2003). Specifically, analysis of regional variation in body-length data in harbour porpoise strandings between 1990 and 2002 demonstrated that porpoises stranding in Southwest England had significantly longer body-lengths compared to porpoises from other regions ($F_{3,571} = 13.1$, $p<0.001$) (Jepson 2003). Analysis of regional variation in the residuals from the best fit regression line of ln(body-length) on ln(age) during the same period also demonstrated highly significant morphometric variation between porpoises stranding in South-west England and those stranding in other regions ($F_{3,528} = 18.9$, $p<0.001$) (**Figure 18**) (Jepson 2003).
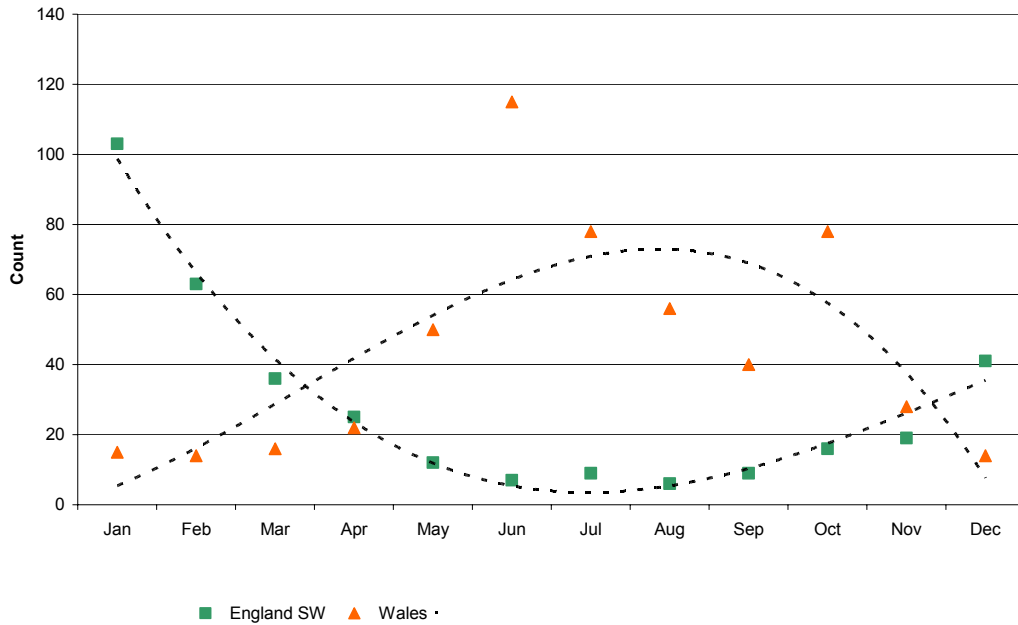
Figure 17. Comparison of monthly numbers of harbour porpoise strandings in Wales and in the south-west counties of England.
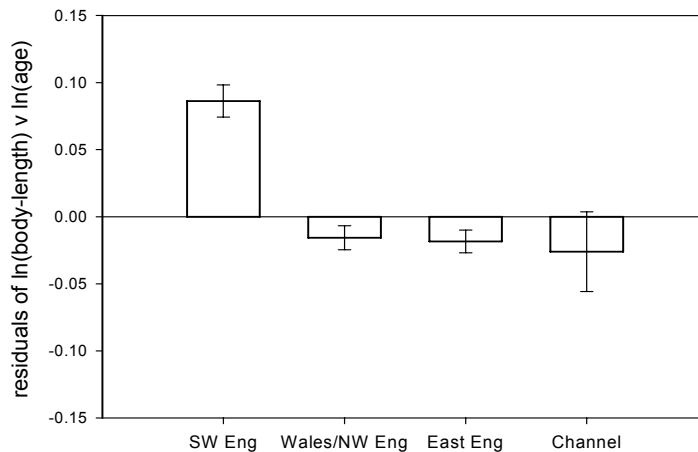
Figure 18   Plot of residuals of from the best fit regression line of ln(body-length) on ln(age) against region. Bars = 1 S.E.

C-4 SPERM WHALES (PHYSETERIDAE)

Pygmy sperm whale (*Kogia breviceps*)

Pygmy sperm whales (*Kogia breviceps*) have appeared more frequently in the strandings record since 1980, peaking between 1995 and 1999 (Muir *et al.*, 2000), but in the most recent five years only one report of this whale was received. Since

the NHM began keeping records in 1913 there have been nine reports of stranded pygmy sperm whales, the first in 1966. The distribution included strandings on the south-west coast of Scotland, south-west England and south Wales. The most recent and southernmost record occurred at Thurlestone, Devon in January 2002. This was consistent with previous NHM data which shows that the majority of strandings have occurred in late autumn and the winter months.

These animals are difficult to separate from their congener, dwarf sperm whale, at sea, but the limited sightings data for this species indicate that there may be a resident population in deep water south and west of the UK (Reid *et al.* 2003).

Sperm whale (*Physeter catodon*)

Spatial and temporal distribution of UK strandings
The sperm whale is distributed widely in the oceans of the world with its main concentrations in warmer waters, but its range extends into colder waters with indications that old males have a preference for these higher latitudes. One stranding was reported from the east coast of Scotland at Queensferry, Lothian in June 2002. In 2003, seven of the eight sperm whales stranded that year were seen in the North Sea with two as far South as Norfolk. These strandings all occurred between January and May. Two of the strandings in 2004 were also found on North Sea coasts in Norfolk and Lincolnshire in January and March, respectively.

Most records for this period of review are consistent with the established distribution of sperm whales, although the record from Cornwall in February 2001 is more unusual. The very decomposed nature of this carcase may indicate that it had drifted some distance before stranding.
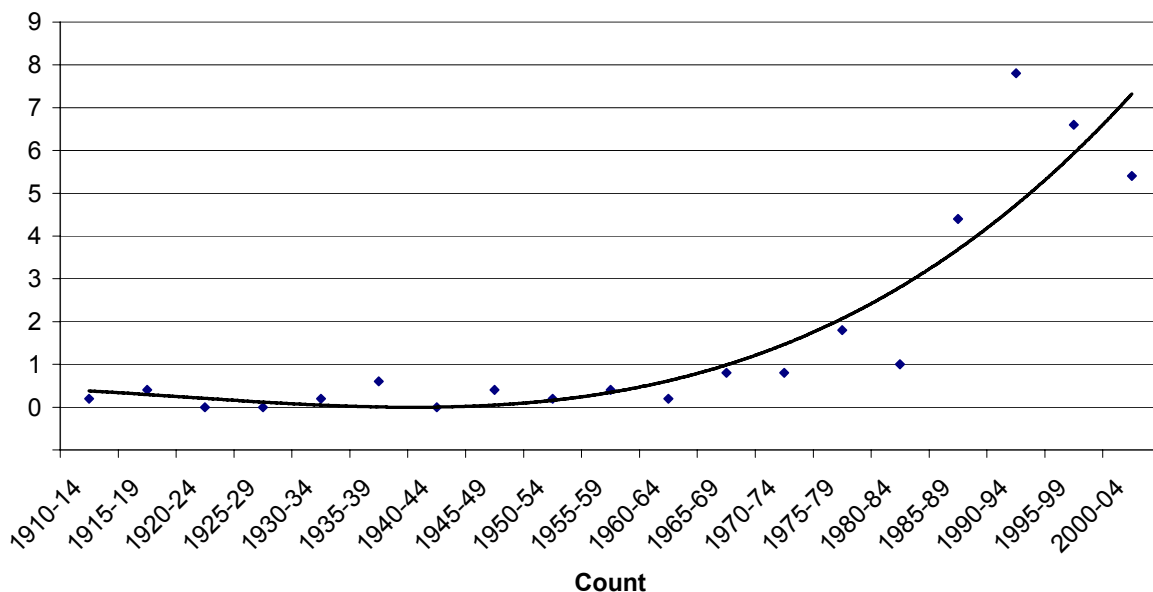
Figure 19. Annual average records of sperm whale strandings in 5 year periods for 1910-2004

In the last ten years the number of reported strandings has been stable with expected annual fluctuation, although the total of only three in 2004 was unusually low. The longer term trend (**Figure 19**) shows that there has been a steady rise in the number of reports since the 1960's, which may indicate a gradual sperm whale population recovery from whaling, or that an increasing number of animals are moving into higher latitudes.

C-5 BEAKED WHALES (ZIPHIIDAE)

Few strandings of beaked whales occurred during the period of this report (**Table 8**). There were 11 Sowerby's beaked whales (*Mesoplodon bidens*), seven Northern bottlenose whales (*Hyperoodon ampullatus*) and six Cuvier's beaked whales (*Ziphius cavirostris*) reported between 2000 and 2004.

Unidentified toothed whales and other cetaceans

It is often not possible to identify carcases which are very inaccessible or in an advanced state of decomposition. The numbers of unidentified odontocetes and cetaceans that could not be identified as either toothed or baleen whales are given in **Table 1** of the main report. The number of such reports naturally varies from year to year. In 2003, a year in which the number of unidentified dolphins was high, more unidentified toothed cetaceans were also reported, mainly from the Cornish coast between January and March. Because of their size and the nature of their injuries it is likely that these carcases were connected to the by-catch-related dolphin and porpoise deaths in the south-west of England.

Table 8: List of UK beaked whale strandings reported 2000-2004.

SWno	Species	Sex	Metres	Date	County
2000/2b 1	Mesoplodon bidens	F	4.14	07 Jan 2000	Fife, Scotland
2000/64a 1	Mesoplodon bidens	-	4.37	22 Mar 2000	Orkney Islands, Scotland
2000/150b 1	Mesoplodon bidens	F	4.98	08 Sep 2000	Lincolnshire
2001/81e 1	Mesoplodon bidens	F	4.5	14 Apr 2001	Highland, Scotland
2001/156a 1	Mesoplodon bidens	-	4	15 Jul 2001	Orkney, Scotland
2001/210b 1	Mesoplodon bidens	M	4.59	18 Sep 2001	Highland, Scotland
2002/201 1	Mesoplodon bidens	F	4.44	07 Jun 2002	Cornwall
2003/280a 1	Mesoplodon bidens	F	4.73	04 Jul 2003	Highland, Scotland
2004/240c 1	Mesoplodon bidens	F	3.78	04 Aug 2004	Highland, Scotland
2004/287 1	Mesoplodon bidens	F	-	24 Sep 2004	Londonderry, N.Ireland
2004/289 1	Mesoplodon bidens	M	3.96	27 Sep 2004	Ceredigion, Wales
2002/125a 1	Ziphius cavirostris	F	6.25	18 Mar 2002	Strathclyde, Scotland
2002/222 1	Ziphius cavirostris	M	5.65	20 Jun 2002	Norfolk
2002/247 1	Ziphius cavirostris	-	5	07 Jul 2002	Cornwall
2003/401i 1	Ziphius cavirostris	-	6	18 Dec 2003	Western Isles, Scotland
2004/36a 1	Ziphius cavirostris	-	-	26 Jan 2004	Strathclyde, Scotland
2004/170b 1	Ziphius cavirostris	F	-	23 May 2004	Western Isles, Scotland
2000/47b 1	Hyperoodon ampullatus	-	6.7	12 Mar 2000	Western Isles,

						Scotland
2001/73b 1	Hyperoodon ampullatus	-	-	?? Apr 2001	Orkney, Scotland
2001/204c 1	Hyperoodon ampullatus	F	5.8	04 Sep 2001	Strathclyde, Scotland
2001/251b 1	Hyperoodon ampullatus	F	7.5	09 Nov 2001	Orkney, Scotland
2003/390 1	Hyperoodon ampullatus	-	-	04 Dec 2003	Devon
2004/125 1	Hyperoodon ampullatus	M	4.5	29 Mar 2004	Lancashire
2004/239 1	Hyperoodon ampullatus	-	4.57	01 Aug 2004	Humberside

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ANNEX D: Peer-reviewed Scientific Publications (1990-present)

The following publications in peer-reviewed scientific journals have been generated using data or tissue samples derived from the UK Cetacean Strandings Programme.

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- Baker, J.R., Jepson, P.D., Simpson, V.R. & Kuiken, T. (1998) Pathology of grey seals (*Halichoerus grypus*) found dead on the coasts of England, Wales and the Isle of Man. *Veterinary Record* **142**, 595-601.
- Bennett, P.M., Jepson, P.D., Law, R.J., Jones, B.R., Kuiken, T., Baker, J.R., Rogan, E. & Kirkwood, J.K. (2001) Exposure to heavy metals and infectious disease mortality in harbour porpoises from England and Wales. *Environmental Pollution* **112**, 33-40
- Bones, M., Neill, B. & Reid, R.J. (1998) Fraser's Dolphin *Lagenodelphis hosei* stranded in South Uist: first record in U.K. waters. *Journal of the Zoological Society of London* (1998) **246,** 460-461
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- Bull, J.C., Jepson, P.D., Ssuna, R.K., Deaville, R. & Fenton, A. The Relationship between Polychlorinated Biphenyls in Blubber and Levels of Nematode Infestations in Harbour Porpoises, *Phocoena phocoena. Parasitology* (in press)
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- Dawson, C.E., Perrett, L.L., Davison, N.J., Quinney, S., Simpson, V.R., (2004) Brucella infection in marine mammals off the Cornish coast. *Veterinary Record* **155,** 32
- Drout, V., Berube, M., Gannier, A., Goold, J.C., Reid, R.J. and Palsboll, P.J. (2004) A note on genetic isolation of Mediterranean Sperm Whales, *Physeter*

macrocephalus, suggested by mitochondrial DNA. *Journal of Cetacean Research and Management* **6(1)** 29-32

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- Foster, G, Ross, HM, Naylor, RD, Collins, MD, Ramos, CP, Garayzabal, FF, Reid, RJ (1995) Cetobacterium ceti gen-nov, sp-nov, a new gram-negative obligate anaerobe from sea mammals. *Letters In Applied Microbiology* **21**, 202-206

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Additional Report writing

- Dr Paul Jepson authored a 2004 report to the *UK Defence Science and Technology Laboratory (Dstl)* on "*In vivo* gas bubble disease in cetaceans".
- Dr Paul Jepson co-authored a 2005 International Convention of the Exploration of the Seas (ICES) ad-hoc working group report on the "Impact of Sonar on Cetaceans". Advice deriving from the report was commissioned by the European Commission.

Scientific Meetings attended

Staff of the UK Cetacean Strandings Project attended meetings of the UK Biodiversity Action Plan Cetacean and Turtle Steering Group from 2000-2003.

Staff of the UK Cetacean Strandings Project also presented research findings at several scientific conferences, mainly organised by the European Cetacean Society (ECS) and the Society of Marine Mammalogy (see Conference Proceedings) including a workshop on Cetacean Strandings Networks at the ECS annual scientific conference (April 2005).

The co-ordinator of scientific research (Dr Paul Jepson) also attended the following meetings in relation to the impacts of anthropogenic noise on marine mammals:-

- US Marine Mammal Commission Technical Workshop on "Understanding the impacts of anthropogenic noise on beaked whales" held in Baltimore, USA (April 2004)
- Marine Board & National Science Foundation workshop on impacts of noise on marine mammals (September 2004)
- An International Workshop (US Marine Mammal Commission/UK JNCC) on Policy on Sound and Marine Mammals (September 2004)
- A Defence Science and Technology Laboratory (Dstl) beaked whale workshop in London (October 2004)
- IWC Workshop on Habitat Degradation (inc noise effects) in Italy (November 2004)
- Dstl beaked whale workshop held at SMRU in UK (November 2004)
- Invited speaker at UK IACMST (Inter-Agency Committee on Marine Science and Technology) Working Group on Underwater Noise and Marine Life (February 2005)
- Beaked whale/sonar workshop at ECS annual scientific conference (April 2005)
- ONR/Dstl-funded Intergovernmental conference on Noise and Marine Mammals in Italy (May 2005) (invited participant)
- Scientific meeting to begin developing an International Strategy for Scientific Research on the Effects of Noise on Marine Mammals, held at Tubney House, Oxford (October 2005) (invited participant)

Geoff Foster (SAC Inverness) attended the following meetings:-
- Brucellosis 2003 in Pamplona. Spain (September 2003)
- EC COST Action on Brucellosis in Tromso (2002)
- VTEC 2003, Edinburgh. Analysis of faeces from birds and marine mammals in Scotland for verocytotoxin-producing *Escherichia coli* O157 (June 2003).

Geoff Foster also organised and chaired an EC COST meeting on marine mammal brucellosis in Inverness (November 2004).

Teaching/training

Student Theses completed
A number of postgraduate student theses were completed in the 2000-2004 period of this report using resources collated by the UK Cetacean Strandings Research Programme.

PhD theses

- Jepson, P.D. (2003) Pathology and toxicology of stranded harbour porpoises (*Phocoena phocoena*) in UK waters. *PhD thesis.* Royal Veterinary College (University of London). Pp.1-221. July 2003.

MPhil theses

- Thatcher, O.R. (2005) Genetic population structuring and relationships between heterozygosity and susceptibility to parasitic disease in the harbour porpoise *Phocoena phocoena* population of Great Britain. *MPhil thesis.* Magdalene College, University of Cambridge. Pp.1-143. July 2005.

MSc theses

- Goldsworthy, C. (2000) Study into the association between chronic exposure to polychlorinated biphenyls and thymic involution and cystic change in harbour porpoises (*Phocoena phocoena*) from British waters. *Submitted in part fulfilment of the requirements for the degree of Master of Science in Wild Animal Health, University of London, 2000*

- Villa-Garcia, Gracia (2002) Histopathological study of the harbour porpoise (*Phocoena phocoena*) mammary gland: Observations on development and tissue concentrations of polychlorinated biphenyls (PCBs)

- Elliott, P.P. (2003) The prevalence of fat embolism in UK cetacean strandings mortalities: Formalin-fixed lung tissue archives stained with Oil Red O. *Submitted in part fulfilment of the requirements for the degree of Master of Science in Wild Animal Health, University of London, 2003*

- Parades Gonzales, Jorge (2003) Ultrasonographic and physical anatomical comparision between axial and optical dimensions in the eye of a harbour porpoise (*Phocoena phocoena*). *Submitted in part fulfilment of the requirements for the degree of Master of Science in Wild Animal Health, University of London, 2003*

- Ssuna, R. (2004) Investigating the relationship of polychlorinated biphenyls in blubber with levels of nematode infestations and cause of death in harbour porpoises (*Phocoena phocoena*). *Submitted in part fulfilment of the requirements for the degree of Master of Science in Wild Animal Health, University of London, 2004*

- Provan, H. (2005) The condition scoring of stranded cetaceans using direct and indirect techniques. Submitted in part fulfilment of the requirements for the degree of Master of Science in Wild Animal Health, University of London, 2005

- Zachariah, A. (2005) Evidence of leptospirosis in phocid seals of coastal United Kingdom. Submitted in part fulfilment of the requirements for the degree of Master of Science in Wild Animal Health, University of London, 2005

Lectures

During the period of this report, Dr Paul Jepson gave two lectures per academic year to students registered for the MSc courses in Wild Animal Health and Wild Animal Health co-taught by the Institute of Zoology (Zoological Society of London) and the Royal Veterinary College (University of London). Practical demonstrations of post-mortem examinations on marine mammal carcasses were also conducted at the Institute of Zoology by staff working on the Cetacean Strandings project.

ANNEX E: Other Collaborative Research Activity

A number of collaborative research activities were supported by access to data and/or tissue samples collated by the Defra-funded UK Marine Mammal Strandings Programme.

- *Institute of Zoology (Zoological Society of London)/Cambridge University.* A *NERC*-funded PhD/MPhil studentship investigating the population structure of harbour porpoises in UK-waters using genetic (microsatellite) markers was completed in 2005. The research was supervised jointly by the Institute of Zoology (Drs. Simon Goodman and Paul Jepson) and Cambridge University (Dr. Bill Amos).

- *Institute of Zoology (Zoological Society of London).* The final report of a *WWF-UK*-funded investigation into relationships between indices of testicular development and endocrine disrupting chemical exposure in UK-stranded harbour porpoises, utilising data and tissue samples collected by the UK Marine Mammal Strandings Project, was completed by the Institute of Zoology in 2003 (Dr. Peter Bennett, Prof. Bill Holt, Dr. Paul Jepson) (see Holt et al. 2004).

- *Institute of Zoology (Zoological Society of London).* A collaborative research programme to develop population-based models for the dynamics of parasites and their mammalian hosts, utilising data from UK-stranded cetaceans (specifically harbour porpoises), was initiated in 2004 in collaboration with Dr. Andrew Fenton and Dr. James Bull (see Bull et al., in press).

- *Professor Antonio Fernandez, Facultad de Veterinaria Universidad de Las Palmas de Gran Canaria, Spain.* Pathological investigations into gas and fat embolism in cetaceans (see Jepson et al. 2003, 2005; Fernandez et al 2004, 2005).

- *Professor Antonio Fernandez, Facultad de Veterinaria Universidad de Las Palmas de Gran Canaria, Spain.* Immunohistochemical investigations of intracytoplasmic inclusions in cetacean hepatocytes.

- *Professor Antonio Fernandez, Facultad de Veterinaria Universidad de Las Palmas de Gran Canaria, Spain.* Immunohistochemical studies of muscle fibre types and interstitial skeletal muscle fat globules in shallow and deep-diving cetaceans.

- *Dr. Simon Northridge, Sea Mammal Research Unit, Gatty Marine Laboratory, University of St. Andrews, St Andrews, Fife.* Teeth and stomach contents from cetaceans stranded in England and Wales are routinely sent for teeth ageing and stomach content analysis respectively. This biological data from UK stranded cetaceans forms an integral part of additional Defra-funded research on cetacean by-catch co-ordinated by the Sea Mammal Research Unit.

- *Dr. Phil Hammond, Sea Mammal Research Unit, Gatty Marine Laboratory, University of St. Andrews, St Andrews, Fife.* Provision of data from UK cetacean strandings for modelling component of 2005-2006 SCANSII research project (SCANSII has multiple funding sources including EU and Defra).

- *Dr. Ailsa Hall, Sea Mammal Research Unit, Gatty Marine Laboratory, University of St. Andrews, St Andrews, Fife.* Odds ratio analyses of the impacts of polychlorinated biphenyls (PCBs) on populations of harbour porpoises (see Hall et al., in review).

These analyses have the potential to develop into future risk assessment-type analyses of the population-level impacts of PCBs in porpoise populations of known size and PCB exposure.

- *Dr. Ailsa Hall, Sea Mammal Research Unit, Gatty Marine Laboratory, University of St. Andrews, St Andrews, Fife.* Blubber and lung samples from UK-stranded seal and cetacean species are being analysed as part of a bioinformatics study of leptin (a multifunctional hormone involved in body fat regulation and respiratory function). The investigation is trying to determine how leptin structure has evolved in species with very different diving capabilities and physiologies.

- *Dr. Ailsa Hall, Sea Mammal Research Unit, Gatty Marine Laboratory, University of St. Andrews, St Andrews, Fife.* Marine mammal immune function studies. Liver samples from a range of UK-stranded marine mammal species are being used for exploratory studies of innate immunity, particularly the expression of antimicrobial peptides such as cathelicidins and definsins.

- *Moredun Research Institute, Pentlands Science Park, Bush Loan, Penicuik, Midlothian, EH26 0PZ, Scotland.* Expert neurohistopathological studies on cetacean tissues from Scottish cetaceans.

- *Dr. Graham Pierce, University of Aberdeen, Department of Zoology, Lighthouse Field Station, George Street, Cromarty, Ross-shire IV11 8YJ.* Collaboration on life history, dietary and toxicological studies of harbour porpoises and other cetaceans (including internationally collaborative EU-funded *BIOCET* project completed in 2005).

- *Dr. Paul Thompson, University of Aberdeen, Department of Zoology, Lighthouse Field Station, George Street, Cromarty, Ross-shire IV11 8YJ.* Collaboration on biological and genetic studies of harbour porpoises and bottlenose dolphins.

- *Dr. Krishna Das, Laboratory for Oceanology, MARE Center, B6c, Liège University, B-4000 Liège, Belgium/Dr. Ursula Siebert, Forschungs- und Technologiezentrum Westkueste Hafentoern D-25761 Buesum Germany.* A collaboration with these two research institutes was established in 2004 to investigate potential thyrotoxic histopathological effects of persistent organic pollutants (such as PCBs) on thyroid microanatomy.

- *Dr. Ursula Siebert, Forschungs- und Technologiezentrum Westkueste Hafentoern D-25761 Buesum Germany.* On 1st October 2004, the Institute of Zoology began a small (Defra-funded) 2-year project to examine the feasibility of using formalin-fixed auditory tissue (ears) collected from UK-stranded cetaceans to investigate potential auditory impacts of anthropogenic noise exposure. The research is in collaboration with the Forschungs und Technologiezentrum Westkueste, Buesum, Germany.

- *Dr. Seamus Kennedy, Veterinary Sciences Division, Department of Agriculture for Northern Ireland, Stormont, Belfast.* Immunoperoxidase screening for the presence of morbillivirus antigen in fixed samples of cetacean lung and other tissues.

- *Dr. Charlie Dalley, Lab Testing Department, Veterinary Laboratory Agency, New Haw, Addlestone, Surrey, KT15 3NB.* Serological studies to assess exposure to *Leptospira* spp. in UK-stranded marine mammals (both seal and cetacean). Collaboration initiated in 2005 with the Institute of Zoology (Mr. Arun Zachariah,

Dr. Paul Jepson and Dr. Clyde Hutchinson), Sea Mammal Research Unit (Drs. Ailsa Hall and Paddy Pomeroy) and University of Aberdeen (Dr. Paul Thompson).

- *Dr Andrew Kitchener, Royal Museum of Scotland, Edinburgh, Scotland.* Recording all marine mammal stranding events in Scotland. Marine mammal skulls and scapulae are sent to Dr. Kitchener for marine mammal morphometric studies.

- *Dr Alistair MacMillan, Brucella Section, Veterinary Laboratories Agency, New Haw, Addlestone, Surrey, KT15 3NB.* Serological studies to assess exposure to *Brucella* spp. and typing of *Brucella* isolates (see Ross et al 1996; Jepson et al 1997; Foster et al. 2002).

- *Dr Jacques Godfroid, Centre d'etude et de la Recherches Veterinaires et Agronomiques, B-1180 Brussels, Belgium.* Molecular typing of *Brucella* isolates.

- *Professor David Collins, University of Reading, Reading RG6 6AP.* Sequencing novel of bacterial isolates.

- *Drs Marie-Francoise Van Bressem and Tom Barrett, Institute of Animal Health, Pirbright, Surrey.* Serological surveys of UK-stranded cetacean and pinniped sera for antibodies to morbilliviruses (see Van Bressem et al 1998, 2001).

- *Dr David Williams, Royal Veterinary College (University of London), Hawkshead Campus, Herts.* Ultrasonographic and physical anatomical comparison of axial and optical dimensions of the harbour porpoise eye.

- *Dr. Richard Morris, Department of Veterinary Pre-clinical Studies, University of Liverpool.* Anatomical studies of the innervation of the cetacean skin.

- *Dr. Peter Ditchfield, Department of Archaeology, University of Oxford.* Samples of bone, skin and blubber are supplied for studies of stable isotopes in marine mammal tissues.

- *Dr John Goold, University of Bangor, Bangor, Wales.* A small number of cetacean carcasses stranded in Wales are used as demonstration material to marine mammal MSc students at Bangor University. Tissues and data derived from these animals are also used to support MSc thesis projects as part-fulfilment of the MSc in Marine Mammal Science.

- *Dr John Pinnegar, CEFAS Lowestoft Laboratory, Pakefield Road, Lowestoft, Suffolk, NR33 0HT.* Stable isotopes analyses of pinniped and harbour porpoise tissues from the central and southern North Sea as part of a *Defra*-funded R&D project (MF0323) centred on the Dogger Bank and central North Sea and looking at the importance of sandeels to various predators (mostly bird and fish).

Collaborative research on UK/European pinnipeds

Although the current Defra-funded Cetacean Strandings contract is exclusive to UK-stranded cetaceans and marine turtles, earlier contracts under the UK Department of the Environment (later the Department for Environment, Transport and Rural Affairs) in the 1990s had a slightly wider remit covering UK-stranded marine mammals (both cetaceans and pinnipeds). To facilitate this, pinniped post-mortem protocols were developed in 1991 and pinniped data and tissues were included in national data/tissue archives. This research background in disease surveillance of UK

pinnipeds has provided a solid platform for additional collaborative research activity on European pinnipeds.

- *Pathological investigations of UK-stranded pinnipeds.* Since 1990, UK-stranded seals have been recorded and allocated individual reference numbers by the Institute of Zoology (IoZ). In addition, a small number of post-mortem examinations of UK-stranded seals are conducted annually by IoZ and the Scottish Agricultural College, Inverness (SAC). These low-level investigations provide some baseline data on health status, disease prevalence and enable the archiving of seal tissues for future research.

- *Mass mortality of Caspian seals (Phoca caspica) in 2000.* In 2000, scientists from the IoZ, SMRU (St Andrew's University), SAC Inverness, Erasmus University Rotterdam and Department of Agriculture and Rural Development (Northern Ireland) investigated a mass mortality of Caspian seals in 2000 (see Kennedy *et al.* 2000; Kuiken *et al.* in press).

- *2002 European Phocine distemper epizootic.* A European epizootic of phocine distemper killed over 22,000 common seals (*Phoca vitulina*) between April 2002 and early 2003. A Defra-funded national research investigation into the impact of PDV on UK seal populations was co-ordinated the IoZ (contract ref: WM0301) in collaboration with SMRU (University of St Andrew's), SAC Inverness, Marine Environmental Monitoring and The Natural History Museum. The final report of this investigation (compilers: Lawson and Jepson 2004) can be found at: http://www.defra.gov.uk/science/project_data/DocumentLibrary/WM0301/WM0301_1655_ABS.pdf

- *Serological studies of exposure to poxvirus, herpesvirus and Toxoplasma in UK seals.* This 2002-2003 investigation was conducted as a collaborative study between the IoZ and the Department of Veterinary Pathology, University of Liverpool (Professor Malcolm Bennett).