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REPORT OF THE WORKING GROUP ON MARINE MAMMAL ECOLOGY (WGMME)

9-12 MAY 2005

SAVONLINNA, FINLAND

International Council for the Exploration of the Sea

Conseil International pour l'Exploration de la Mer

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1 Introduction

1.1 Participation

The Working Group on Marine Mammal Ecology (WGMME) met at the Metsähallitus, Natural Heritage Services, Savonlinna, Finland from 9-12 May 2005. The following list of individuals participated in all or part of the meeting (see Annex I for addresses).

Luis Arregi	Spain
Penina Blankett	Finland[5]
Wolfgang Dinter	Germany
Eero Helle	Finland[1]
Ivar Jüssi	Estonia
Mart Jüssi	Estonia
Olle Karlsson	Sweden[2]
Jouni Koskela	Finland[3]
Iwona Kuklik	Poland
Mervi Kunnasranta	Finland
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Per Risberg	Sweden[4]
Meike Scheidat	Germany
Tero Sipil≅	Finland
Krzysztof Skóra	Poland
Olavi Stenman	Finland[6]
Mark Tasker	UK[2]
Michail Verekin	Russia
Gordon T. Waring (Chair)	USA
Håkan Westerberg	Sweden[5]

[1] Present Monday & Tuesday; [2] Present Tuesday through Thursday; [3] Present Monday; [4] Present Tuesday; [5] Present Monday through Wednesday; [6] Present Wednesday and Thursday

The Working Group members were welcomed by Tero Sipilä, and Matti Määttä, Director Metsähallitus Natural Heritage Services, Savonlinna, Finland. The WG reviewed the Terms of Reference (TORs) and a work schedule was adopted.

1.2 Terms of Reference

The **Working Group on Marine Mammal Ecology [WGMME]** (Chair: Gordon T. Waring, USA) will meet from 9 May to 12 May 2005 in Savonlinna, Finland to:

a) report on the populations of seals and harbour porpoise in the Baltic marine area, including the size and structure of the populations, distribution, migration pattern, reproductive capacity, effects of contaminants on the health status, and additional mortality owing to interactions with commercial fisheries by sub-region (bycatch, intentional killing),

b) develop further the response to the European Commission standing request regarding fisheries that have a significant impact on small cetaceans and other marine mammals:

  i) review any new information on population sizes, by catches or mitigation measures and suggest relevant advice, ii) review the usefulness of available prey data to quantify marine mammal-prey interactions for multispecies modelling purposes, and provide recommendations for future sampling schemes for quantification of marine mammal-prey interactions;

iii) review information on common dolphins, including:

a) size, status and trends on the NE Atlantic population(s) (or possibly sub-populations);

b) bycatch in fisheries, including fleet composition by gear type, fishing effort, and bycatch rates;

c) mitigation measures and advice, including level of priority

This information should be disaggregated according to scale area as appropriate depending on the distribution of common dolphins populations and the "distribution" of the fisheries.

c) for each marine mammal species affected by fishing, compile data (in excel spreadsheet format) which quantifies the seasonal distribution and abundance at spatial scales, where possible, that correspond to ICES rectangles for the North Sea. The data will be submitted to REGNS secure website in preparation for the REGNS integrated assessment workshop from 9-13 May 2005. These data should, where possible, be for the period 1984-2004 to assess trends. Also where possible, provide information on diet and variation/change of this for all species described;

d) start preparations to summarize the size, distribution and incidental catches of marine mammal populations in the ICES areas (VII - X);

e) begin preparations for a future Workshop (associated with WGMME meeting) on health and immune status, disease agents and links to environment quality;

f) develop a Cooperative Research Report on threats to marine mammal populations based on a compilation of prior reports of this and former marine mammal working/study groups;

The Group will report to ACE at 18 May for the attention of ACE.

1.3 Justification of Terms of Reference

a) This request is a biannual request from HELCOM. This request to ICES should address the following five fundamentals of a potential conservation plan:

i) The Group should identify possible target and limit reference points for grey seals that would satisfy the provisions of the Habitat Directive, while considering the uncertainty inherent in assessing the population trends, birth rates and total mortality. Risk levels to explore could be 1, 2.5 and 5%;

ii) Population growth rates that under different assumption about total mortality that would be needed for maintaining status quo to with high probability allow the population to continue to increase towards a future target. A growth rate could in its self be an interim target in the conservation plan;

iii) Information on indicators for health for the population birth rates, contaminants etc;

iv) Evaluation of habitat protection and seal sanctuaries in the Baltic and possible need for more use of such;

v) Identification of gaps in monitoring of the population and by catches.

b) This work is required in relation to MoU between the European Commission and ICES. This also addresses Goal 1 of the ICES Strategic Plan.

c) This has been requested by WGECO/REGNS to provide marine mammal data for the REGNS integrated assessment in 2005.

d) Comprehensive information on cetacean abundance, distribution and interactions with fisheries in ICES areas VII-X has not been available for review at prior WGMME meetings. This work will provide the first comprehensive review of cetacean abundance, by catch, and stranding. This addresses Goal 1, 2 and 5 in the ICES Strategic Plan.

e) Marine mammals are upper trophic level predators that accumulate high levels of pollutants. This work is needed to develop workshop terms of reference and identify participants. This addresses Goal 2 in the ICES Strategic Plan.

1.4 Acknowledgements

WGMME thanks Tero Sipilä and Matti Määttä, Director, Metsähallitus Natural Heritage Services, Savonlinna, Finland for their excellent hospitality and support to the meeting. We also thank Arne Bjorge (Norwegian Institute of Marine Research), Rohan Cosgrove (BIM), Tero Härkönen, (Swedish Museum of Natural History), Carl Kinze (xx), Alice Mackey and Simon Northridge (SMRU, University of St Andrews), Luca Mirimin (University College Cork), Ada Natoli (Durham University), Graham Pierce (University of Aberdeen), Vincent Ridoux (CRMM, Université de La Rochelle), Begoña Santos, Monica Silva (University of St Andrews), Ingrid Tulp (Netherlands Institute for Fisheries Research) and Karen Stockin, for providing information and/or reports for use by WGMME.

The Chair also acknowledges the diligence and commitment of the participants, which ensured that the extensive Terms of Reference for this meeting were addressed.

2 Report on Baltic seal and harbour porpoise populations

Term of Reference a) report on the populations of seals and harbour porpoise in the Baltic marine area, including the size and structure of the populations, distribution, migration pattern, reproductive capacity, effects of contaminants on the health status, and additional mortality owing to interactions with commercial fisheries by sub-region (bycatch, intentional killing),

2.1 Grey seal Halichoerus grypus

2.1.1 Population discreteness, distribution and migration

Movements and site fidelity of Baltic grey seals have been studied using photographic identification of individuals (“photo-id”). Seals with distinctive pelage markings were photographed on the major summer haul-out sites. Profile photographs of the head and neck were matched using a software program to generate a database of capture histories from 1995 to 2000. The majority of the re-sightings were made in the area where the animals were originally identified, suggesting that Baltic grey seals exhibit a high degree of site fidelity during the summer. Furthermore, the proportion of re-sightings made within the same area showed only a slight decline over time, suggesting that fidelity to a particular site may last over several years. Movements between adjacent areas were relatively common while movements between the different Baltic sub-basins were rare (Karlsson *et al*. in press).

Sattellite telemetry of six grey seals from the southern Baltic (Dietz *et al*. 2003) and 17 individuals from the Central and North Baltic (Sjöberg et al. 2003) show the capablity for long distance movements. A Danish study showed that grey seals made extensive movements, up to 850 km, away from Rødsand to Sweden, Germany, Estonia and Latvia. These dispersal patterns were reflected in the calculated Kernel home ranges, where seals that dispersed farther from the tagging site had large estimates of area use (home range). The corresponding Kernel home range for grey seals was 51,221 km^2 ranging from 4,160 to 119,583 km^2 for five out of the six grey seals (Dietz *et al.* 2003). The seals in Sjöbergs study (Sjöberg et al. 2003) tracked during the summer and autumn showed fidelity to one or two haulout sites. Most daily distances was less than 10km however longer movements were carried out, four out of 12 seals moved more than 150 km at least once. Seals tracked during winter and early spring spent more time at sea and ranged over larger areas. One of four seals tracked during this period moved over 100 km in 24 hours. The tracked grey seals exhibit some degree of site fidelity, but for most seals fidelity is shown to a general area not a single haulout.

2.1.2 Effects of contaminants

Extremely high levels of both persistent organic pollutants and heavy metals have been measured in the Baltic seals. Exceptional concentrations of PCB and DDT, which are thought to be the greatest threat to the Baltic seals, were over 100 mg/kg in blubber of the seals in the late 60's and the 70's. During the years 1996-1998 average sum PCB and sum DDT levels were 66 mg/kg and 38 mg/kg in liver in the ringed seals, while the corresponding contaminants levels were 28 mg/kg and 8 mg/kg in liver in grey seals, respectively. These levels are still 3 to 100 times higher than in seals living in relatively unpolluted areas (Nyman 2000). Results indicate that although the contaminant levels in the Baltic seals have decreased since in the 1970's, the levels are still high, especially in ringed seals. Mercury concentrations exceeding 100 mg/kg fresh liver weight were measured in the seals in the 1970's (Herva and Häsänen 1972, Kari and Kauranen 1978). The concentrations of heavy metals are still exceptionally high and no clear patterns of decrease in levels have been observed (Jonsson *et al.* 1996, Fant *et al.* 2001).

Grey seals are ingesting less PCB and DDT compounds than Baltic ringed seals. The differences in levels of toxins could be explained by differences in their diets. The toxic load in grey seals, however is still very high, when compared to seal populations from other areas. The toxic effects of environmental contaminants could be causing divergence in vitamin levels in the Baltic seals. A-vitamin levels are lowered and E-vitamin levels elevated in correlation with PCB- and DDT loads in the tissues of studied seals. The vitamin A accumulation in the seals is poorly known and more research should be conducted on the vitamin dynamics (Nyman 2000, Nyman *et al.* 2001, 2002, 2003, 2005, Routti *et al*. 2005, in press). Heavy metals are not known to have any detrimental effects on the Baltic grey seals (Fant *et al.* 2001).

2.1.3 Health status

The general health status of Baltic grey seals has improved, but many diseases occur in the population. Most of these maybe considered normal especially in the older age classes (Liskins and Pilats 2005, Westerling *et al.* 2005), but high age also means a long time of exposure to pollutants. Colonic ulcers caused by hookworms (*Corynosoma* sp.) occur frequently through age classes. The colonic ulcers can be lethal in some cases. The overall high prevalence of this lesion seems to be unique for the Baltic grey seal and ringed seal populations. In addition, this condition is more common in the Gulf of Bothia than in the Baltic proper (Bäcklinand Bergman 2005).

12 stranded grey seals on the Mecklenburg-Vorpommern coast of Germany were recovered for examinations in the period 1998-2003. Some of the older animals (up to 40 years), in particular, showed leiomyomas, occlusion and stenosis of the uterus, loss of bone substance, fibrosis and multifocal calcification of the kidneys and the adrenal glands, heavy parasitic

burdens, thrombosis and sclerosis of blood vessels and severe necrotic splenitis. Parasitological investigations revealed *Pseudoterranova decipiens* in the stomach, and *Corynsosoma strumosum*, or *semerme* in the intestine. Potential pathogenic bacteria found were *Escherichia coli*, a- and b-hemolytic Streptococci and *Clostridium perfringens*. In addition, zoonotic bacteria *Erysipelothrix rhusiopathiae* could be cultivated (Harder *et al*. 2004)

2.1.4 Reproductive capacity

The frequency of uterine occlusions /stenoses and leiomyomas had decreased and the pregnancy rate clearly increased in mature females from 1977-86 to 1987-96 (Bergman 1999). The positive trends have continued since then. In tissues collected from animals shot by local hunters, which maybe considered as a random sample in respect to reproductive capacity, no occlusions, stenoses or leiomyomas were found (in females aged 3-37 years) in 2001-2004. The pregnancy rate was 81%, increasing from 68% at 3-8 years to 87% at over 8 years of age. Thus reproductive capacity of the Baltic grey seal in the Gulf of Bothnia seems to be normal (Helle *et al.* 2005).

2.1.5 Current abundance and survey methodology

In Finland aerial censuses aided by aerial photographs were used, whereas counts from boats and land were used in the other countries (Sweden, Estonia, Latvia, Lithuania, Poland, Germany and Russia). The essential use of these haul-out counts is to monitor the population abundance, as the censuses are carried out with largely comparable methods. Census results are considered as relative indices of abundance, which are smaller than the true population size.

Annual numbers of grey seals counted were: 9,700 in 2000, 10,300 in 2001, 13,100 in 2002, 15,950 in 2003, and 17,640 in 2004. The distribution of grey seals by sea area from the 2004 count were as follows:

- Bothnian Bay and North Quark 1,330
- Sea of Bothnia excluding Åland achipelago 870
- Waters around SW Finnish archipelago including Åland 7,735
- Gulf of Finland 870
- Western Estonia 2,690
- Swedish Baltic proper south of Gulf of Bothnia down to 58°N (northern tip of Gotland) 3,900
- Swedish Baltic proper south of 58°N, 245 (Halkka *et al*. 2005).
- Recent observations of grey seals in Polish waters show no increase in the number of individuals occurring in the Southern Baltic during the last 5 years (2000-2004) (Kuklik and Skóra, 2005)

Photoidentification of grey seals was used to estimate the population size. The estimate for the total Baltic population is 15,631 (95% C.I 9,592 to 19,005) in 2000. This estimate is based on a value for annual survival of identification markings of 0.9035, which was also estimated using the photo-id data. The estimate is subject to an unknown, but probably small, upward bias resulting from the risk of failure to identify all individuals in the photographs used for the analysis. An estimated minimum of 15,950 seals were counted at moulting haul-outs in 2003, which provides a lower bound on the population size in that year, and allowing for growth of the population, represents 80% of the photo-id point estimate (Hiby *et al.* submitted)

2.1.6 Current information on by catches and human-induced mortality

A network of fishermen are paid to keep detailed journals of seal damages in Sweden. At present the system covers approximately 5 % of the fishing effort (Tärnlund 2005). The information includes data on by catches. Preliminary 2004 data indicate that approximately 300 grey seals are by caught in the Swedish Baltic fishery. This is a 25 % decrease, since 2001. The decrease is partly due to the introduction of seal-protected salmon traps and partly to a decreased effort in the gillnet fishery. An increase in the grey seal abundance in the Baltic proper is indicated by a doubling of the by catch per unit of effort in the cod gillnet fishery. No recent by catch data are available from Finland, Russia and Estonia.

A limited protective hunting has been allowed in Sweden north of 58^{o} N. Table 2.1 shows the number of licenses given and the actual number of grey seals killed during the period 2001-2004. There is no official hunt carried out in Estonia or Russia. The extent of any illegal hunting is unknown.

Table 2.1. Number of licences issued and seals shot in 2001-2004.

	Sweden			Finland/mainland			Finland/Åland			Total
Year	Licenses	Number shot	%	Licenses	Number shot	%	Licenses	Number shot	%	Number shot
2001	150	57	38%	100	60	60%	89	54	61%	171
2002	150	79	53%	180	92	51%	156	95	61%	266
2003	170	79	46%	230	128	56%	171	82	48%	289
2004	170	81	48%	395	135	34%	232	152	66%	368

2.1.7 Current population status

The 2000-2004 annual estimates indicate an increasing trend in the size of the Baltic grey seal population. This trend, however, should not be use to express the true rate of increase, because an increase of such magnitude over the period of observation is biologically unrealistic in the grey seal. It implies unrealistic fecundity/survival rates. Increases in census experience and efficiency, change in seal haul-out behaviour, and increasing number of annual replicated counts, are all factors that may have played a role in the observed increase in the numbers of seals.

2.2 Baltic ringed seal Phoca hispida botnica

2.2.1 Population discreteness, distribution and migration

Presently, Baltic ringed seals are found in four main areas: the Bothnian Bay, Gulf of Finland, Archipelago Sea and Gulf of Riga (Miettinen *et al*. 2005) (Figure 2.1). A population genetics study has shown that there are no genetic differences between these four stocks (Palo *et al.* 2001). There is no new information on movements and migrations of the species in the Baltic sea since 2003 (ICES 2003), but number of ringed seal sightings in Polish coastal waters have increased from an average of 5 in the 1980s and the 1990s to ten since 2000 (K. Skóra pers. comm).

Pilot studies of breeding distribution of ringed seals in the Gulf of Riga in 2004 and 2005 have shown that seal distribution can be linked to the location of certain ice types and formations and can vary according to ice conditions.

2.2.2 Effects of contaminants

Ringed seals are still suffering from exceptionally high concentrations of persistent organic pollutants (POPs), such as PCBs and DDT compounds. Ringed seals are ingesting more PCB and DDT compounds than grey seals, and this corresponds to elevated levels of contaminants in their tissues. The higher levels of DDT in ringed seals compared to the grey seals could be explained by differences in their diets. The toxic effects of environmental contaminants could be causing divergence in vitamin levels between Baltic seals and reference seal populations from other seas. A-vitamin levels are lowered and E-vitamin levels elevated in correlation with PCB- and DDT loads in the tissues of studied seals. However, the vitamin A accumulation in the seal is poorly known and more research should be conducted on the vitamin dynamics (Nyman 2000, Nyman *et al.* 2002, 2003, 2005, Routti *et al.* In press). Heavy metals are not known to have any detrimental effects on the ringed seals (Fant *et al.* 2001).

2.2.3 Health status

Except for studies on uterine occlusions, the general health status of Baltic ringed is not well known. However, there are some diseases and parasites, which have been documented in the Baltic ringed seals. Especially the incidence of heartworms (*Dipetalonema spirocauda*) is quite typical for ringed seals (Westerling *et al.* 2005).

2.2.4 Reproductive capacity

Reproductive capacity and disorders in the Baltic ringed seals have been studied in the Bothnian Bay since the 1970's. The frequency of uterine occlusions peaked in the late 70's (60% of mature females). Since 1991, there has been the strong age dependency in the frequency of uterine occlusions: 11% at 3-10 years, 35% at 11-20 years and 83% over 20 years. The ringed seal population is still suffering from uterine occlusions, although recovery has slowly taken place. Recently (1995-2004) 23% of mature females have been affected by uterine occlusions. Pregnancy rate of healthy females was 76 % in 1996-2004 (Helle *et al.* 2005).

2.2.5 Current abundance and survey methodology

Standard aerial surveys (method in Härkönen and Lunneryd 1992) have been carried out in Gulf of Bothnia in 2003-2005 and Gulf of Riga in 2003. However, during the study ice conditions were poor, therefore survey coverage was incomplete. Surveys in Finnish and Estonian sea areas of the Gulf of Finland in 2003 and in Russian territorial waters in 2004 were not successful. In Archipelago Sea in SW Finland horizontal observations (i.e. from ice level) were used in 2002 - 2004 to estimate the minimum population size during annual molt (Miettinen *et al*. 2005), aerial observations were carried out in 2005 (A. Halkka pers. com).

Surveys results were:

- Gulf of Bothnia: 3205 in 2003 and 4748 in 2004 (Swedish Museum of Natural History, unpubl.)

- Gulf of Finland: No valid population estimates are available for the 2003 - 2005 period.

- Archipelago Sea: Observed population size in 2004 was 120 – 140 individuals (Miettinen *et al*. 2005). WWF-Finland Baltic seal group conducted ringed seal flight surveys in the Archipelago Sea from 31 March to 8 April 2005. The ice period was exceptionally short as ice began to form in the outer Archipelago Sea in the end of February, and most of the ice had disappeared by the time of the second survey. More than 40 adult ringed seals and 6-7 pups were observed during the surveys. The pups were situated openly on the ice as snow accumulation had not been sufficient for the formation of lairs. No population estimate is possible based on the survey as an unknown proportion of seals haul out during the end of the breeding season, but the population appears to be very small (with a maximum of few hundreds) as indicated by earlier data based on boat-based surveys of seals on skerries during molting time (A. Halkka pers. com).

- Gulf of Riga: 579 (101 SE) individuals, based on 2003 survey.

2.2.6 Current information on by catches and human-induced mortality

The Swedish reporting system (see 2.1.6) shows some by catches in the Bay of Bothnia. Approximately 30 animals were by-caught, mainly in whitefish fish traps. This level is low as compared to the grey seal by catch. The ringed seals are found offshore during the summer when most of the inshore gillnet fishery takes place, but move inshore during the autumn when they are exposed to the trap fishery. No data available from the Finnish fishery in the Gulf of Bothnia or from the southern distribution range.

Approximately 5-10 ringed seals from the Bothnian Bay are taken annually for research purpose in Finland.

2.2.7 Current population status

Ringed seal stock in the Gulf of Bothnia has been increasing at 5% per year (T. Härkönen, unpubl., Swedish Museum of Natural History).

Relative abundance estimates in the Gulf of Finland (counts during the ice-free period) indicate low, but stable numbers of ringed seals. Given the low population numbers – hauling out population of only 150-170 individuals (Stenman *et al.* 2005) the population is endangered.

Studies in the Archipelago Sea have only recently started and thus the population status can not be established.

The ringed seals in the Gulf of Riga have probably suffered a population decline between 1996 and 2003, but there is no recent data to evaluate the current situation.

The Archipelago sea ringed seal population seems thus to be distributed mostly to the eastern part of the area. Restricted distribution, small size and apparent status of a demographically distinct sub-population indicate that the Archipelago sea ringed seal population should be considered as a threatened subpopulation of Baltic ringed seals.

The ringed seal in the southern distribution range of Baltic Sea (Gulf of Riga and Archipelago Sea) is sensitive to ice conditions during breeding period, so mild winters can significantly affect the reproductive success of these populations.

2.3 Harbour seal *Phoca vitulina*

2.3.1 Population discreteness, distribution and migration

There is no recent data on population discreteness, distribution and migration of the Kalmarsund population of the harbour seal. A satellite telemetry study of harbour seals in Kattegat was carried out in the years 2000 -2002 at Rodsand seal sanctuary in Denmark (Dietz *et al.* 2003). The harbour seals remained within 50 km of the tagging site year-round. The average Kernel home range (95% fixed Kernel) of the harbour seals was 394 km^2 ranging from 237 to 709 km^2.

2.3.2 Contaminant load and health status

2.3.2.1 Kattegat

No specific studies of contaminant load in harbour seals have to been conducted during the reporting period (2003-2004) so indices about health status have to be drawn from general health status of all seals in the area (grey seals, ringed and harbour seals). Levels of environmental contaminants (mainly PCBs and DDT) in the seals prey have decreased during the last decades. However, in some areas, the decline of PCBs has stabilised. Organochlorines had negative effects on the reproductive capacity of both ringed seals and grey seals from the 1960s to the 1980s (Helle 1986; Bergman and Olsson 1986; Bergman 1999). Such effects are also suggested for harbour seals in the period 1977-1989 (Härkönen *et al*. 2002). The disease complex described by Bergman (Bergman and Olsson 1986; Bergman 1999) is rarely seen in recent years and only in old individuals, but the prevalence of intestinal ulcers has increased during last decade. Intestinal ulcers may be fatal if the intestine is perforated leading to peritonitis. The epizootic in 1988 and 2002 killed a large portion of the population, mortality in some areas exceeded 50%. During the seal epizootic in 1988 more than 1000 lower jaws were collected in the Kattegat, Skagerrak and the Baltic. Subsequent analyses revealed a high prevalence of alveolar exostosis, not found at all in reference material collected 1850-1930. Similar changes in Baltic grey seals were thought to be indicative of organochlorine pollution (Mortensen *et al*. 1992, Härkönen *et al*. 2002).

Since 1996 in total 11 common seals were found on the German Baltic Sea coast of Schleswig-Holstein. Investigations were made by the Research and Technology Center Westcoast (University of Kiel). The animals were of various ages and in varying states of decomposition. Pathological findings included gastroenteritis due to infection of kryptosporidia, suppurative myositis, and hepatitis, abscessation in the muscles and stomach wall with septicemia, bronchitis, and endometritis. One harbour seal found in 2002 died due to Phocine Distemper Virus. Morbillivirus infections were not found in any other year in Schleswig-Holstein. Between 1998 and 2003, carcasses of 27 common seals were found on the coast of Mecklenburg-Vorpommern. Of these, 7 were PDV-positive in 2002 (Harder *et al*. 2004).

2.3.2.2 Kalmarsund

No specific studies of contaminant load in harbour seals has been carried out during the reporting period, therefore indices about health status have to be drawn from the general health status of all seals in the area (grey seals, ringed and harbour seals) – see above.

2.3.3 Reproductive capacity

The low rate of population increase in the Kattegat area, compared to the Skagerrak prior to the last epizootic, is an indication of reduced reproductive capacity (Härkönen *et al*. 2002). No new information is available from the Kalmarsund region.

2.3.4 Current abundance and survey methodology

Aerial surveys are used in Sweden and Denmark to estimate the harbour seals population size. Occasional observation of harbour seals in German waters can not be used for population estimates but it provides information on distribution range of the species in the southern Baltic. Survey results are given in Table 2.2.

Table 2.2. Aerial survey results of harbour seals. Source: HELCOM Habitat 6/2004, 12/2.

YEAR	SEA AREA	COUNTED NUMBER	POPULATION ESTIMATE
2003	Danish Kattegat	1956	3431
2003	Danish Belt Sea	465	815
2003	Danish South Baltic	386	677
2004	Kalmarsund	361	555
2004	Makläppen	127	195
2004	Swedish Kattegat	2468	3797

2.3.5 Current information on by catches and human-induced mortality

There are no data on by catches of the Baltic harbour seal population (Kalmar Sound population). The Swedish reporting system (see 2.1.6) shows a total of 380 by catches in Skagerrak and Kattegatt for the Swedish fishery. Approximately half of this is in the eel fyke-net fishery and half in gillnets. No Danish by catch data is available.

In the Swedish area of Kattegatt 4 harbour seals have been shot to protect the local coastal fishery in 2004. No data are available from Denmark.

There is a concern regarding the introduction of harbour porpoise pingers in the ICES statistical rectangle 4160 from 1 June 2005 (Council Regulation (EC) No 812/2004). The pinger signal lies within the hearing range of harbour seals and the experience from trials with acoustic scaring devices for seals is that they tend to act as a “dinner bell” and attract seals to the fishing gear rather than deter them. If this happens the by catches of harbour seals from the small Baltic population, which resides in this rectangle, may increase. A Swedish study is initiated to follow the effect of pingers on seal damages and by catches.

2.3.6 Current population status

The Kalmarsund population was not affected by the PDV epizootic in 2002 and is increasing approximately 9.5% per annum (Härkönen *et al.* In press). The Kattegat population of harbour seals suffered mass mortality in 2002 (Härkönen *et al.* In press) and is recovering although the population recovery rate have not been established.

2.4 Harbour porpoises Phocoena phocoena

2.4.1 Population discreteness, distribution and migration

Voluntary sighting reporting programs have been in place for the last four years. In Finland, the Ministry of Environment is collecting information on incidental sightings based on the reporting form available on a public website. From 2001 to 2004, 17 sightings of 42 harbour porpoises were reported to the website. In Poland the data has been collected by Hel Marine Station, University of Gdańsk but no sightings were reported in 2003-2004. The Swedish Museum of Natural History has collected the reports on incidental sightings and several of them have been made along the coasts, all the way up to the Gulf of Bothnia.

In Germany, the environmental NGO GSM [Gesellschaft zum Schutsz der Meeressängetiere e.V.] annually distributes reporting forms for incidental sightings of harbour porpoises. Most of the observations come from the Kiel Bight (ICES area IIIc) and only very few reports come from the area IIId. In addition, information on incidental sightings have been collected by German Oceanographic Museum in Stralsund in area IIId, and by the FTZ [Forschump und Technologiazentrum, University of Kiel] the area IIIc. All data are compiled and stored in a database at the FTZ.

Historical data on harbour porpoise occurrence in Estonian waters have been collated. Most reports come from the 1930s, and the observations were widely distributed all along the Estonian coast (I.Jüssi, pers. comm.)

Several genetic and morphometric studies have concluded that the Baltic porpoises are a separate population different from those living in Kattegat and North Sea. A recent review of the population structure studies of the Baltic harbour porpoise, based on the result of the direct genetic studies, concluded that no statistically significant differences have been shown that justify a separate Baltic population (Palme *et al*. 2004). On the other hand such population/hypothesis can not be excluded and according to the precautionary principles the Baltic porpoises should be managed as a distinct population (Palme *et al*. 2004).

Despite the difficulties associated with a small sample size, a joint research project funded by Germany for the implementation of the Jastarnia Plan contains a subproject on the genetic differentiation of the harbour porpoises from the ICES area IIId (Baltic population). The available samples will be collected from the entire area to further analyse the genetic structure of harbour porpoises in the Baltic proper.

Within the Jastarnia project a GIS-database for information on the Baltic Sea harbour porpoise was created. The future the database will include information on effort and incidental sightings, strandings and bycatches (both recent and historical) from the ICES areas 22 (IIIc) and 24, 25 and 26 (IIId). Central and Eastern Baltic Sea (South of 56°N, East of 12°E). Information on acoustic monitoring of porpoises with towed or stationary hydrophones will also be included. The data will be presented by an interactive map on the internet, which will be located at www.balticseaporpoise.org. This database will then act as a forum and the data will be accessible for all researchers as well as the general public. Additional information regarding the project will also be found at this website, as well as contact details for all parties who have included data. Since the project was started in autumn 2004, Latvia, Germany, Poland and Sweden have included data, but more countries have showed a definite interest and are invited to contribute.

2.4.2 Effects of contaminants

Butyltins (BTs) and phenyltins (PhTs) were determined in the livers of marine mammals, including harbour porpoise, that were by-caught or stranded along the Polish coast of the Baltic Sea (Ciesielski *et al.* 2004). BT compounds were detected in all the liver samples, whereas PhTs were not detected in any of the samples. Age-related trends to accumulate BTs in immature porpoises were found. No male-female differences in BTs concentrations were observed. In comparison to butyltin levels in marine mammals from other geographic regions, the samples analyzed indicate a significant degree of tributyltin pollution along the Polish coast of the Baltic Sea (Ciesielski *et al.* 2004).

2.4.3 Health status

No new data were reported.

2.4.4 Reproductive capacity

No new data were reported.

2.4.5 Current abundance and survey methodology

Line-transect aerial surveys have been conducted (University of Kiel) in the German part of the Baltic Sea since 2002 and will continue until the year 2006. Harbour porpoises have been sighted east of the island of Ruegen during several flights. Data is still too scarce to allow interpretation in terms of seasonal patterns (M. Scheidat, pers. comm.).

PODs (Porpoise Detectors) have been deployed in the German, Polish and Estonian part of the Baltic Sea, respectively, since 2002 and 2003. The results of German studies indicate a decrease in click detection (porpoise positive days) from the western German waters of the Kiel Bight to the eastern Pommeranian Bight. It also gives some indication of seasonal changes in click activity. In Poland and Estonia few detections have been recorded so far. (I. Kuklik & I. Jüssi pers. comm.)

2.4.6 Current information on by catches and human-induced mortality

Preliminary data on recent levels of bycatch in the Swedish Kattegat/Skagerrak fishery (100 animals a year) are similar to what was reported in 2003. According to the Swedish reporting system for bycatch covering 5% of the Swedish Baltic fleet no bycatch has been reported in ICES IIId area (Westerberg, pers. comm.).

Eight bycatches were reported voluntarily by fishermen in Polish waters in years 2003-2004 (I. Kuklik, pers. comm.)

Latvia – bycatch of 2 harbour porpoises were reported between 2003 and 2004 (V. Pilats, pers. comm.).

No new data were available on the bycatch from other fisheries.

2.4.7 Current population status

No new data were available.

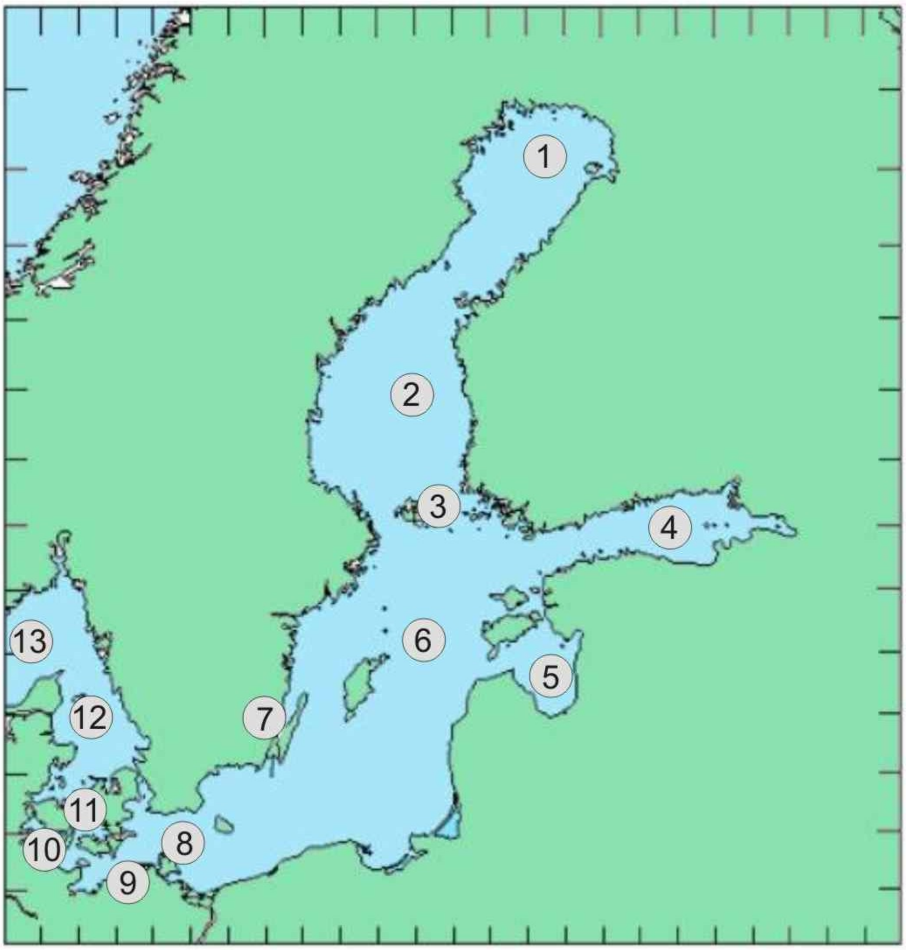

Figure 2.1 Names of sites mentioned in the text:

1 – Bothnian Bay, 2 – Bothnian Sea, 3 – Archipelago Sea and Åland Sea, 4 – Gulf of Finland, 5 – Gulf of Riga, 6 – Baltic Proper, 7 – Kalmarsund, 8 – Pommeranian Bight, 9 –Meclenburg – Voorpommen, 10 – Schleswig – Holstein and Kiel Bight, 11 – Belt Sea, 12 – Kattegat, 13 – Skaggerak

2.5 Fundamentals of a potential conservation plan for Baltic grey seal

The only biological reference point for the population level is the carrying capacity of the system. In the case of Baltic grey seal historical data show that the population has been 88 000 – 100 000 animals (Hårding and Härkönen 1999), and that the carrying capacity is at least at this level. However, it is uncertain if the environment could support a population of that magnitude. The WG did not identify a target population level, and noted that any target would need to consider socioeconomic impacts. The expertise required for the latter is outside the competence of the WG. The WG noted that a favourable conservation status, according to the Habitat Directive, is a stable or increasing population size well above any extinction risk and distribution throughout the natural range.

Defining a limit reference point requires a tool for risk analysis. Several models exist for this. One example developed to investigate different hunting regimes for the Baltic grey seal is given in Harding *et al*, in press. This model is based on a Leslie matrix with vital parameters appropriate for the population. Basic for the analysis is a choice of population level, judged to give a high risk for population extinction. In this model the level used is the standard choice of getting below 10% of the original population size, in this case 1000 females. By modelling the time development with the observed variability of demographic parameters the risk of reaching this level can be calculated. This is defined as the risk for quasi-extinction.

The model is very sensitive to the choice of some parameters. The population growth rate is a key value. Harding *et al*. (in press), used a value of 7.5 %, which is taken from the longest available monitoring series (1990-2003) for the Swedish region. An estimate for the entire Baltic region cannot be determined, since survey efficiency and effort have varied in some regions. An annual growth rate of 7.5% means the model becomes conservative in its predictions, as compared to using higher growth rates observed in the core distribution area.

As is discussed in 2.1.5 the growth rate varies for different areas of the Baltic and what is an appropriate value for the overall Baltic population should be analysed further. It is also important to improve the monitoring methods used in the Baltic countries. Several other parameters, as the fecundity rate, have been taken from Atlantic Ocean grey seal populations due to the lack of data from the Baltic. Obtaining life history data on the Baltic population is important in order to improve the model.

Harding *et al*. (In press), examined several scenarios. Assuming a population with 3000 females and no hunting regime the quasi extinction risk is 0.02 %, increasing to 2% and 7.5%, respectively, with annual hunts of 300 or 400 females. If more than 400 females are hunted, the risk for quasi extinction exceeds 10%. This can be compared to the present best estimate of the population size, which is approximately 20, 000 animals or 10, 000 females, assuming 50:50 sex ratio, and a hunting quota of 797seals in 2004, with 368 seals shot the same year.

Another similar kind of model has been developed within the EU research project FRAP (Development of a procedural Framework for Action Plans to Reconcile conflicts between large vertebrate conservation and the use of biological resources: fisheries and fish-eating vertebrates as a model case) (http://www.FRAP-Project.UFZ.DE). This model has also been applied to the Baltic grey seal and also shows a low extinction risk at the present population level.

A zero population growth rate will maintain the status quo. In the Baltic, however, there is a large uncertainty regarding the population growth rate due to variability in the monitoring programs. To detect a 5 % change from the 7.5 % level requires about 9 years of data with the present monitoring regime (Harding *et al*. in press). This means that a management must use an adaptive mode and incorporate the uncertainty in the population count.

The present system of monitoring grey seals only provides an index of the true population size, but a direct estimate of the approximate growth rate. For the purpose of managing a conservation plan this should be sufficient as long as the counted number of seals is a lower bound of the total population.

No evaluation has been made of the population effect of the existing seal sanctuaries in the Baltic. The benefit of avoiding disturbances at breeding and haul-out sites seems evident. The large size and variability in the range of foraging habitat makes it difficult to define essential foraging habitat. Specific migration constrictions could be defined as essential habitats.

For a re-colonisation of the southern Baltic coast by seals it is obviously that undisturbed haul-out sites are essential to provide a year-round basis for pupping, moulting and resting. Protection of suitable sites from disturbance seems to be the crucial factor, to be pursued equally in already existing nature conservation areas. Restoration of historically used and in the meantime probably degraded habitats could enhance the re-colonisation process. (Restoration of natural habitats and distribution ranges is also an objective of the COUNCIL DIRECTIVE 92/43/EEC).

The current population monitoring programs cover most of the distribution range. Population distribution has been expanding, therefore monitoring should be initiated in the more peripheral range of the south-eastern Baltic. By catch monitoring is essentially completely absent in the Baltic countries. A voluntary sampling scheme is used in Sweden, which seems to give reasonably reliable data, given the situation with an economic compensation for seal damages and the introduction of protective hunting.

2.6 Status of the freshwater seals of the Baltic region

2.6.1 Saimaa seal Phoca hispida saimensis

2.6.1.1 Population discreteness, distribution and migration

Population backcasting method produced a maximum population size of 1300 animals in the year 1893, representing a density of approximately 0.30 seals per km^2. At present (in 2000) the densest population (0.88 - 1.12 seals/ km^2) was found in the small (25 km^2) Kolovesi National Park. An extrapolation, based on area of Lake Kolovesi, gives a potential total population size in Lake Saimaa of about 3800 - 4900 seals (Table 2.3) (Sipilä & Koskela 2003).

The carrying capacity of Lake Saimaa, which is roughly estimated by mean productivity in the main food species vendace, corresponds to approx. 6300 seals in the lake (1.44/ km^2). Estimating the potential number of the shoreline lair sites and density of lair sites presently in use in the main Lake Saimaa breeding areas, provides at least 10700 lair sites, corresponding to 5350 seals (1.21/km^2) (Table 2.3) (Sipilä & Koskela 2003).

Table 2.3. Estimations of pristine seal population size in Lake Saimaa.

Estimating method	Population size
Backcasting	100 - 1300 (in 1893)
Density of seals, water sqkm	3800 – 4900
Nourishment (vendace)	approx. 6300
Density of shoreline lairsites.	approx. 5350 – 10700

Radiotelemetry studies have shown that adult Saimaa seals exhibit high site fidelity and movements, longer than about 20 km do not typically occur (Hyvärinen *et al.* 1995; Kunnasranta 2001; Koskela *et al.* 2002; Kunnasranta *et al.* 2002). However, movements of sub adult animals seem to be longer (Kunnasranta 2001). Wintertime disturbance is supposed to be one of the main threats to the seal population (Sipilä 2003). A pilot study is underway to measure behaviour during breeding period of the seals, and also aims to estimate effects on human caused disturbance on the seals (Rautio *et al.* 2005)

The minimum observed population of Saimaa ringed seal was approx. 190 seals in 1990 (Sipilä 2003). In 1990-2004, the mean annual population growth has been 2.6% per annum. The estimated changes in population size differ a lot between breeding areas (Table 2.4). It is very likely that the ringed seal will vanish from the northern parts of Lake Saimaa in the near future (Figure 2.2, Table 2.4), (Sipilä *et al.* 2005, WP 7).

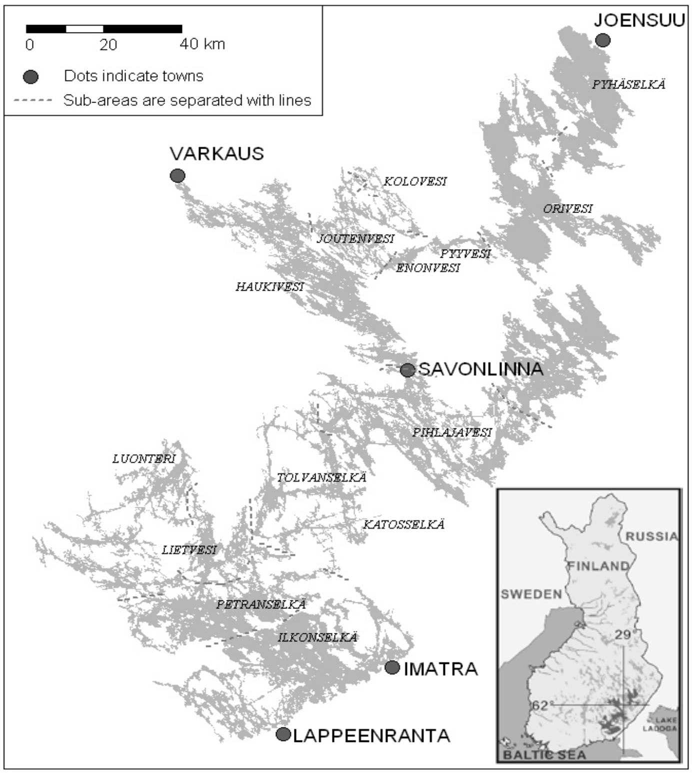

Figure 2.2. Sub-areas of Lake Saimaa. © Metsähallitus 2005, © Maanmittauslaitos 1/MYY/05

Table 2.4. Estimated number of Saimaa ringed seals in the early winter 1990, 1995, 2000, 2004 and mean annual growth rate in different sub-areas of Lake Saimaa. These figures do not include pups born in the estimation year.

Sub-area	Number of seals				Growth rate 1990 - 2004
	1990	1995	2000	2004	
Pyhäselkä	13	9	4	4	0.919
Orivesi	14	13	12	10	0.976
Pyy-Enonvesi	7	7	17	15	1.056
Kolovesi	15	15	25	25	1.037
Joutenvesi	16	16	25	30	1.046
Haukivesi	48	49	53	55	1.010
Pihlajavesi	38	43	60	80	1.055
Tolvan-Katosselkä	16	20	20	20	1.016
Lietvesi	15	10	9	10	0.971
Luonteri	2	2	2	2	1.000
Petranselkä	4	6	13	15	1.099
Ilkonselkä	4	4	3	3	0.980
Total amount	189	192	242	269	1.026

2.6.1.2 Effects of contaminants

Current levels of DDT and PCB concentrations are lower (Kostamo 2004), as compared to previous studies (Helle *et al.* 1985). The decrease of OCL compounds from 1981 to 2001 has averaged 75 %. However, the levels of organochlorine concentrations in Saimaa seals have never been as high as those in Baltic seals (Kostamo 2004). There is no updated information on possible effects of environmental contaminants on the Saimaa seals.

2.6.1.3 Health status

Post mortal studies of 66 Saimaa ringed seals in the years 1982-2000 were made by Finnish National Veterinary and Food Research Institute in Helsinki and Joensuu. The infestation rate of lungworms (*Parafilaroides sp.*) was low in Saimaa ringed seal populations, as presence is only three cases were recorded. Heartworm (*Dipetalonema spirocaude*) was found. The uterine state of 7 mature seals was determined, and they were macroscopically normal (Westerling *et al*. 2005)

Intestinal helminthes of the Saimaa seals, especially hookworms (*Corynosoma* sp.), have been studied lately. They do not seem to be harmful for the Saimaa seals (Sinisalo *et al.* 2003, 2004).

Magnetic resonance imaging (MRI) was used to study one drowned one year old Saimaa seal (40 kg). The main advantages of MRI method are the excellent capacity to distinguish tissues and tissue margins and the possibility to observe structures without intervention. Typical to drowned animals, the venous sinuses and vena cava posterior were full of coagulates and non-coagulated blood. Also, the right ventiricle and the chamber were extremely stretched compared to the left side of the heart. In this specimen all other structures were normal, without any indicates of diseases. The specimen was also healthy according normal pathological postmortem study. The MRI study continues for determine possible marks of possible sickness on Saimaa seals carcasses (Usenius *et al.* 2005).

2.6.1.4 Reproductive capacity

The most recent information was presented to ACE in 2003 (ICES 2003 CM/ACE:03 2003, Ref E,G.).

2.6.1.5 Current abundance and survey methodology

The most recent information was presented to ACE in 2003 (ICES 2003 CM/ACE:03 2003, Ref E,G.).

2.6.1.6 Current information on by-catches and human-induced mortality

During the period 1990 – 2004, a total of 209 seal carcasses were found, and 30% of them were too decomposed for post mortem analyses to be done. The cause of death was determined from 146 carcasses. The most common causes of death were drowning (or suffocation) in fishing tackle (52.1%) and mortality of lanugo-coated pups (41.8%). Only 6.2% had died a "natural" death (lanugo-coated pups excluded), e.g. due to infections. (FIGURE2.3)

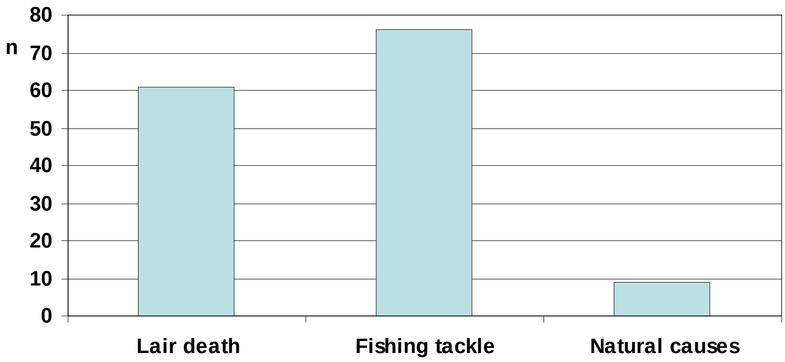

Figure 2.3. Main causes of death of the Saimaa ringed seal 1990-2004. "Lair death" includes prematures, still-borns and accidental death of lanugo coated pups, "fishing tackle" also includes deaths from suffocation without direct evidence of contact with fishing tackle. "Natural causes" does not include lanugo coated pups found dead.

2.6.1.7 Current population status

The present population size, winter 2004-05, was ca 280 seals in Lake Saimaa (Sipilä & Koskela, unpub.)

2.6.1.8 Current information on trophic interactions

The Saimaa ringed seal feed mostly small schooling fish species like perch, roach, vendace, ruff and smelt. These common fish species cover about 90 % of the diet. It has been estimated that Saimaa seal consume about 1000 kilos of fish per year (Kunnasranta *et al.* 1999). According to a present study, food availability is not limiting factor for growth of the seal population in Lake Saimaa (Auvinen *et al*. 2005, in press, WP 5).

2.6.2 Ladoga seal Phoca hispida ladogensis

2.6.2.1 Population discreteness, distribution and migration

A wintering habitat of the northern Lake Ladoga was discovered and seems to be important breeding area for the seals, although most of the population likely breeds on the southeastern part of the lake. On the northern part of the lake most of the lairs are situated in the snowdrifts of islands or islets, although also lairs within snowdrift in ridged ice areas are found (Kunnasranta *et al.* 2001).

The most recent information on population status was presented to ACE in 2003 (ICES 2003 CM/ACE:03 2003, Ref E,G.).

2.6.2.2 Effects of contaminants

The burdens of environmental toxins in the Ladoga seal are considerable, but not very critical (Kostamo 2004). According to Sipilä *et al.* (1996), the cadmium and lead concentrations in the tissues of the Ladoga seal did not increase in habitats that overlap areas of human activity. However, it is noted that mercury concentrations in the liver and kidneys of the Ladoga seal are elevated (Sipilä *et al*. 1996, Medvedev *et al.* 1997) (Table 2.5). Further, the mercury concentrations in the lanugo hair of Ladoga seals are unusually high (Kunnasranta 2001).

Table 2.5. Concentrations of mercury (μg g^{-1}, wet weight) in the tissues of Ladoga ringed seal: all age classes (Medvedev et al. 1997), in tissues of adults (Sipilä et al.1996) and in natal hair (Kunnasranta et al. unpublished).

Liver all age classes Mean ± S.E.	Kidney all age classes Mean ± S.E.	Muscle all age classes Mean ± S.E.	Liver adults Mean ± S.E.	Kidney adults Mean ± S.E	Muscle Adults Mean ± S.E.	Hair Pups Mean ± S.E.
35.39 ± 10.73 *n* 21	6.15 ± 1.23 *n* 11	3.20 ± 2.05 *n* 15	60.8 ± 25.5 *n* 6	15.1 ± 5.9 *n* 7	2.0 ± 0.40 *n* 8	20.51 ± 1.55 *n* 52

The PCB concentrations in the blubber of the Ladoga seal reported by Olsson *et al.* (1986) were low compared to those of the Baltic seal (e.g. Helle *et al.* 1985). In addition, according to Kostamo *et al.* (2000), concentrations of EOX, and DDT and PCB compounds in male seals seem to be lower than in Saimaa male seals, but higher than those of ringed male seals from the White Sea. It should be noted, however, that the highest analysed concentrations are not directly comparable because of differences in ages of the male seals analysed. Additional studies are required to obtain the current status of contaminants in Lake Ladoga seals. The future of the seals is strongly dependent on the amount of pollution emitted by industry and agriculture, and on the use of these compounds in Russia (Kostamo *et al.* 2000, Kostamo 2004).

2.6.2.3 Health status

Post mortal studies of 30 Ladoga ringed seals in the years 1982-2000 were carried out by Finnish National Veterinary and Food Research Institute in Sortavala veterinary station in Karelian Republic

The infestation rate of lungworms (*Parafilaroides sp.*) was high in Ladoga seal population and in two cases heartworms (*Dipetalonema spirocaude*) were recorded. The uterine state of mature females (*n* = 5) were determined, and they were macroscopically normal (Westerling *et al.* 2005).

2.6.2.4 Reproductive capacity

The reproductive capacity of the subspecies is not known.

2.6.2.5 Current abundance and survey methodology

The most recent information was presented to ACE in 2003. (ICES 2003 CM/ACE:03 2003, Ref E,G.).

2.6.2.6 Current information on by-catches and human-induced mortality

There are no reliable statistics on the number of seals caught in fishing tackle. Rough estimates suggest that in the Soviet era, during the 1980's, around 200-400 seals died due to fishing tackle (e.g. Sipilä *et al.* 1996).

We interviewed 36 fishing crew leaders, mainly fishing ship captains, from southern Lake Ladoga and 17 from northern Lake Ladoga in 2003. Present annual mortality due to fishing tackles in Lake Ladoga is relatively high. Approximately 10% of population drowns in Lake Ladoga fishing gear per annum (Table 2.6) (Verekin *et al.* 2005).

Table 2.6. Ladoga seals mortality to fishing tackle in 2003 according to interviews of fishermen.

Fishing plants	Seals caught
Shilsseburg	133
Novaiy Ladoga	152
Olonets- Vilitsa	50
Valaam	9
Pitkäranta	No data
Sortavala	No data
Lahdenpohja	No data
Priozerks	7
Total	351

Increased in fishing effort will increase interactions between seals and fisheries. Fishing probably will pose a serious threat to the seal population in the long term (Sipilä *et al.* 2002).

2.6.2.7 Current population status

The present population size has been estimated to be 3000 - 5000 seals (Verekin *et al.* 2005).

2.6.2.8 Current information on trophic interactions

In the Sortavala veterinary station autopsies were made on 27 Ladoga ringed seals, which had drowned in fishing gears in the northern part of the lake during period 2000-2003.

The most important fish species in the scanty material studied were the smelt (*Osmerus eperlanus*) and the vendance (*Coregonus albula*). In addition, eight other fish species were found in diet, among them ruff (*Gymnocephalus cernuus*) as the commonest. Typical for the fishes found was their small size. The bigger salmon fish species were seldom represented in the material. Crustaceans, particularly *Gammaracanthus lacustris*, were quite common (Stenman *et al*. 2005).

2.7 Recommendations

- Increase efforts in pathological investigations, particularly regarding intestinal ulcers origin and effect. The easiest way to do this is to ensure that a qualified scientist is present during seal hunts
- We re-iterate the need for by-catch monitoring as stated in the WGMME report in 2003; collection of biological material for scientific research with further study of health status, e.g. reproductive capacity and blubber thickness.
- Ringed seals in the Southern distribution range (South of the Bothnian Sea) need more research because the current knowledge about vital population parameters are missing.
- Improvement of grey seal monitoring, standardizing and intercalibration of survey methods is needed for population trend estimates, which is one of the bases for internationally coordinated population management.
- Owing to rapid increase in human activities in the Baltic, there is urgent need to address questions related to human impact on seals through infrastructure development (e.g. shipping, oil transit, fixed links and wind parks)
- Since it is five years since last full photo ID survey, there is a need for estimation of the true grey seal population size (e.g. using photo ID and remote sensing, pup counts, coastal tourism).

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3 European Commission request

Term of Reference: b) develop further the response to the European Commission standing request regarding fisheries that have a significant impact on small cetaceans and other marine mammals:

i) review any new information on population sizes, by catches or mitigation measures and suggest relevant advice,

ii) review the usefulness of available prey data to quantify marine mammal-prey interactions for multispecies modelling purposes, and provide recommendations for future sampling schemes for quantification of marine mammal-prey interactions;

3.1 Review new information

The singular review under ToR (b_i) pertained to common dolphins, which is reviewed under b_{iii}), below.

3.2 Prey data availability and needs

To model the effect of marine mammals on prey populations requires information on abundance, distribution, diet and consumption rates of marine mammal populations, and abundance, distribution and dynamics of prey populations. In the north Atlantic, marine mammal population sizes are monitored on a more or less regular basis, with periods varying from annual counts of seals to decadal sighting surveys for cetacean abundance estimation. As marine mammals are rather long-lived animals their population sizes are relatively invariable across years. However, marine mammal distributions, and thus the proportion and timing of populations that forage within certain areas/ecosystems, may vary significantly between seasons and years in response to environmental variability (Forney 2000) and greatly influence the impact of marine mammals on the various prey populations. Nevertheless, data on seasonal and annual variation in distribution are lacking for most marine mammal species.

Information on marine mammal diet is obtained from numerous sources. Stomach samples of stranded, by-caught or harvested animals and scat samples provide direct information on the species as well as size or age-classes eaten by the mammals, although the biases in this information may be substantial (e.g. Gannon *et al*. 1997, Santos *et al*. 2004). Stranded animals may not be properly functioning animals and their stomach content may poorly reflect the diet of the population. By-caught animals may be individuals scavenging from e.g. trawlers or foraging in trawls, while animals not attracted to fishing operations may be targeting different prey species. Several indirect methods for identifying marine mammal diet are also available. Lately methods comprising analyses of stable isotopes, fatty acid signatures and pollutants are used to trace links between prey species and marine mammals (Bustamante *et al*. 1998, Kirch *et al*. 1998, Hooker *et al*. 2001, Das *et al*. 2003). Such methods may allow identification of the most probable prey species, and, contrary to scat and stomach samples, integrate information on diet over time. Also, synoptic cruises, simultaneously collecting information on marine mammal distributions and distributions of potential prey species, may be used to make inferences about prey use (Fiedler *et al*. 1998, Mauritzen *et al*. 2005). As predators in general tend to aggregate where their preferred prey is available, positive spatial associations between predators and prey may indicate an ongoing trophic interaction (Hassel and May 1974, Fauchald and Erikstad 2002).

Prey consumption rates are generally scaled to body mass of the predator using the general relationship $R = aM^b$, where R is the consumption rate, M is body mass, with a and b estimated from a number of different data sources based on allometric relationships. However, also direct measurements of intake from behavioural studies, estimates of intake based on analysis of stomach contents, estimates of energy requirements based on utilisation of blubber stores, and daily consumption based on feeding rates may provide information on consumption rates (Anon. 2002). Prey consumption estimates are uncertain, and estimates may vary with a factor of 10 for larger whales (Anon. 2002). It is also important to bear in mind that the energy content of a prey species fluctuates between seasons. For instance, the relative amount of lipid in krill varies between 10% and 50% (dry mass) during the year, while the fat content of Barents Sea capelin varies between 3% and 19% (Martenson et al. 1996). Hence, the number or biomass of prey needed to support a marine mammal varies through the year.

The data available on diet and consumption rates can be used to model possible scenarios regarding marine mammal prey interactions. However, to increase the precision and the predictability of such models the models need to integrate changes in prey consumption relative to changes in prey availability, i.e. the marine mammals' functional responses (Anon. 2002). As most marine mammals in the north Atlantic are generalists, they are likely switching between prey species as described by the sigmoidal curves of type III functional response (Mackinson *et al*. 2003 and references therein). The formulation and estimation of functional responses may greatly impact the modelled effect of marine mammals on prey populations (e.g. Mackinson et al. 2003, Tjelmeland and Lindstrøm 2005). However, the estimation of functional responses require information on the number or biomass of each prey species consumed by individual predators over a range of prey abundances, and quantitative information on prey abundance or density in the area where the predator had been foraging. Such information is rarely available (Anon. 2002). Using a time series based on stomach analyses of harvested minke whales in the Barents Sea from 1992 – 2001 in combination with abundance estimates of herring, capelin, krill and cod, Tjelmeland and Lindstrøm (2005) were able to integrate functional responses in assessment models of herring.

3.3 Recommendations for future sampling schemes

Future sampling schemes should, whenever possible, take into account the need to combine marine mammal diet with measures of prey availability to enable modelling of functional responses. Time series of marine mammal distributions, diet and prey availability are inevitable in that respect. Hence, sampling schemes should include sampling repeated over time.

Information on seasonal and annual variation in marine mammal distribution could be obtained by having marine mammal observers on board scientific cruises. Such observations will not be sufficient for abundance estimation, but yield more frequent information on distribution of marine mammals that is available from dedicated sighting surveys. In addition, marine mammals should be equipped with satellite-linked transmitters or time-depth recorders to follow their movements through seasons.

Information on diet should be obtained from scats and stranded, by-caught or harvested individuals when available. These methods may be combined with stable isotopes/fatty acid analyses to evaluate possible biases. Also, for species where no direct sampling method is available, stable isotopes/fatty acid analyses may be among the few methods available for studying diet. Whereas information from stranded and by-caught animals is available at irregular basis, scat sampling and harvest can be/are conducted at more regular basis and thus easier to combine with information on prey availability. We recommend that whenever possible, diet sampling should coincide with sampling of potential prey species in the area, either by sampling marine mammal diet in periods when fish/zooplankton surveys are run or alternatively, run surveys during the periods of marine mammal harvests (e.g. Lindstrøm 2001). Having marine mammal observers on board during fish/zooplankton surveys should provide valuable information on marine mammal behaviour relative to prey availability, given that the most relevant prey species are sampled during the surveys. Also, seasonal variation in the caloric values of relevant prey species should be estimated.

Finally, more detailed behavioural studies following individual mammals over time may be needed to get information relevant for estimating feeding rates, such as potential threshold in prey density for efficient foraging (Piatt and Methven 1992) and search and handling times. Partly, such information can be obtained using time depth recorders, but direct observations are needed to include information on marine mammal foraging behaviour relative to fine-scaled prey distribution (Boyd 1996, Baumgartner and Mate 2003).

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4 Interaction of common dolphins and fisheries in the North East Atlantic

Term of Reference b_{iii}) This term of reference was added in early 2005 following a letter from the European Commission (DG for Fisheries and Maritime Affairs) requesting a review of all considerations concerning common dolphin conservation and fisheries. In particular ICES were asked to review all information available on:

i.) the NE Atlantic population(s) (or possibly sub-populations) including size, status and trends;

ii.) the bycatch in fisheries (by fishing fleet, gear type, overall amount of bycatch, rate of bycatch and overall fishing effort);

iii.) possible mitigation measures and advice, including level of priority.

The European Commission further requested that the above information could be disaggregated to area as appropriate, depending on the distribution of common dolphin population(s) and the dispersal of the fisheries.

4.1 Summary

This term of reference derived from a request from the European Commission concerning the status of common dolphins and the degree of risk posed to it by bycatch in fisheries. The European Commission’s concern had been raised by the large numbers of dead common dolphins arriving on the beaches of western Europe that have evidently been bycaught. In order to understand the population level effects of extra anthropogenic mortality, it is necessary to know the size and status of the common dolphin population and to know the total extra mortality on that population. This section of the report reviews these areas and then reviews what is known and needs to be known on measures to reduce or mitigate the bycatch.

Population

There have been a number of studies of the genetics of common dolphins off north-west Europe. Broadly these have found that a single genetic population is present within the range of common dolphins from north Scotland to the Straits of Gibraltar and maybe further, and at least as far west as 25°W. There is however evidence of reproductive isolation of female common dolphins off Portugal compared with dolphins from further north, and there is some movement between the separate Mediterranean population and that of the adjacent north-east Atlantic. Genetic evidence however only examines long-term population structure. There is evidence from long-lived heavy metal (cadmium) levels of a separation between animals feeding predominantly on the continental shelf and those feeding further offshore in deeper oceanic waters, thus a stock structure may exist among northeast Atlantic common dolphins at least as long as individual life span.

There have been a number of abundance estimates in sections of the range of common dolphins. These sections partly overlap and the surveys have not been simultaneous. Nevertheless, in the order of 500,000 common dolphins appear likely to be present. There is no information on trends in total abundance, but some information on movements, both in the long- and short-term within the range of the species. In the longer-term, it appears that common dolphins have become commoner off northern Scotland in recent years compared to two decades ago. In the short-term there is evidence of movement into areas of continental shelf such as the western English Channel in winter and there is inter-annual variation in the apparent scale of this movement. Observational evidence appears to indicate that these animals are moving from offshore oceanic waters, but this is not necessarily supported by evidence from heavy metal levels.

There is an incomplete range of information on the life history of common dolphins and evidence of differences in distribution between various age/gender groups. There are evidently some distinct differences in the distribution of the two genders and of age classes of common dolphin off north-west Europe and some apparent differences in susceptibility to being caught in fishing nets. This information may be important in deducing overall population effect of any extra anthropogenic mortality. A number of life history traits of common dolphins will contribute to the vulnerability of the species to extra anthropogenic mortality – these being a late maturity, a low pregnancy rate and an approximate lifetime reproductive output of four calves per female.

A part of this extra anthropogenic mortality can be observed by examining stranded animals. National schemes to record these strandings are present along the entire Atlantic seaboard, but some are more comprehensive than others are. In general over the past decade there have been increases in all areas of numbers of common dolphins strandings, but no consistent increase in proportion with evidence of bycatch. In many areas there is a mid to late winter peak in these strandings. There is considerable inter-annual variation in numbers stranded; peaks in numbers stranded appear commonly to occur during seasons of increased proportions of onshore winds. Whether this is due to higher mortality rates in these years, or just a higher proportion of the mortality arriving ashore is uncertain. There has been a lessening (probably to near zero) in the level of deliberate harpooning of common dolphins for food in the last 30-40 years.

Overall many fish and cephalopod species have been recorded in common dolphin diet studies, but it appears that the species when feeding in the inshore habitat focuses primarily on small pelagic fish species. Some of these species are fished commercially (e.g. sardines and blue whiting off the Iberian Peninsula), while other deeper water species (e.g. Lancet fish *Notoscopelus kroeyeri* and some squid) that are consumed in the offshore habitat are not commercially important. A greater understanding of the causes underlying prey distribution and its variability might aid in identifying areas where bycatch may become a problem.

Bycatch

Reports of bycatch of common dolphins in fisheries off north-west Europe stretch back over several decades, but it has only been since the 1990s that large numbers of dead dolphins that had evidently been bycaught have arrived on beaches. As has been previously made clear by ICES and others, the only reliable way of assessing bycatch rates in fisheries is to undertake studies using fisher-independent observers (or observation methods). This report reviews the available information from such schemes. Two types of fishery appear to present a risk to common dolphins – pelagic trawls and bottom-set nets (pelagic drift nets and the setting of pursed seine nets on dolphins used to pose a threat, but both are now prohibited). Some bycatch has been reported in other types of gear.

The pelagic trawl fisheries in the EU are complex and varied, with over 12 target species and six nations involved and at least three major gear types. A similar or probably greater complexity applies to the bottom-set net fisheries of the area. It is clear that some of these fisheries have relatively low or non-existent cetacean bycatch rates, while one or two others clearly have relatively high bycatch rates. For most of them, however, there is insufficient information to assess bycatch rates at present. Various observation schemes are under way, but it is not clear to the Working Group how comprehensive or representative these schemes will be. It will be necessary at some point to review all of the fisheries occurring in the range of the common dolphin off north-west Europe and examine whether or not observation efforts have covered all fisheries in a representative manner. This will mean examining both the time and space over which the fishery and observations have occurred. It appears essential that the VMS data for relevant fisheries will need to be examined, as well as national data for small vessel fisheries.

It is also apparent that there is considerable variability in bycatch rates in those fisheries for which there are several years of data. This implies that there are dangers in taking (or not taking) measures on the basis of only one or two years of data and that fisheries observations may need to be extended in several fisheries where programmes appear to have finished.

The Working Group was disappointed to note that some countries with fisheries liable to catch common dolphins had not submitted recent data to ICES/the Working Group in relation to those fisheries, despite a specific request from the European Commission to do so. The Working Group was concerned that lack of information might be interpreted as lack of bycatch, and that measures to reduce bycatch might be imposed only on those countries/fisheries submitting data. This would be entirely inappropriate. The Working Group considered that measures should conversely be imposed on those countries/fisheries not submitting data, these measures to be removed once data were forthcoming demonstrating low bycatch in the fishery concerned. This would be consistent with the precautionary principle and would also act in support of scientists and administrators attempting to gain this information.

Mitigation measures

The report summarises various possible measures to reduce bycatch. These include limiting the fisheries in areas where large numbers of dolphins are present, using grids and acoustic deterrents in pelagic trawls, using acoustic deterrents in bottom set nets and education of fishers (and encouragement to innovate both gear and handling methods to reduce bycatch). The Working Group considered that priority should be given to gaining further understanding of bycatch phenomena, improving gear/acoustic deterrents, controlling total fishing effort and understanding the drivers behind variation in dolphin distribution. Recommendations are also given for further research/information needs.

4.2 North-east Atlantic population of common dolphin

4.2.1 Evidence for population sub-division

There is a continuous distribution of common dolphins from Scotland to Galicia, and likely as far as the Canary Islands and the Azores. There are no obvious gaps in this observed distribution of the common dolphin in the eastern Atlantic. This does not though exclude the possibility of there being one or more sub-divisions.

Murphy (2004a, b) summarised a number of methods that could be used to investigate any sub-division of populations. In common dolphins, these include fatty acid analysis, stable isotope analysis, contaminant analysis, life histories, genetics and skull measurements. Murphy (2004a, b) analysed common dolphin samples taken from a number of locations between Scotland and Galicia. Differences were found in fatty acid profiles, contaminant levels, stable isotope analyses and some aspects of growth rates, but not in skull measurements or genetics. The first group of factors indicate differences in diets between areas, these differences persist if diets were changed for days to weeks for stable isotopes and from weeks to months for fatty acids; thus confirming that diets are different between areas.

Westgate *et al.* (2003) found no evidence of genetic subdivision using analysis of molecular variance between commons dolphins inhabiting the western North Atlantic (the majority of samples analysed were collected from Georges Bank and off the coast of Virginia and North Carolina). A significant variation in both haplotype frequencies and haplotype frequency/distance (F_{ST}=0.017, p=0.0003; ΦST=0.02, p =0.01) was found between samples analysed from the North-east (samples obtained from dolphins incidentally caught by the Irish tuna fishery off the southwest coast of Ireland) and North-west Atlantic. It was concluded that common dolphins the North-west Atlantic are composed of a single panmictic group, whereas gene flow between western and eastern north Atlantic is more limited.

Investigations into geographical variation in skull morphology of common dolphin indicate some population segregation within the north-east Atlantic (samples analysed from Ireland, Scotland, England, Wales and Spain), with common dolphins off Portugal differing with those from other areas (Murphy 2004b). Portuguese female common dolphins appear to be reproductively isolated, and may not interbreed with common dolphins from other areas in this study, even though Portuguese males appear to disperse northwards (Murphy 2004b). Natoli *et al*. (2003) found no significant variation between common dolphins in the western Mediterranean and the adjacent North Atlantic (Straits of Gibraltar, Portugal), which indicates gene flow between those areas. Both skull morphometric and genetic data suggest that females from the Mediterranean mix with common dolphins off Portugal.

Preliminary analysis by Natoli *et al*. (2003) showed a significant variation ($p<0.001$) between samples obtained from England and the western Mediterranean, but not between samples from Portugal, the Straits of Gibraltar and England. Although sample sizes were small, further analysis by Natoli *et al*. (in prep.) using microsatellite and *mt*DNA analysis, concluded that there is a low level of differentiation in common dolphins in the north-east Atlantic, supporting the concept of an single genetic population from Scotland to the Canary Islands. Samples analysed were from common dolphins inhabiting waters off the coast of Scotland, the Celtic Sea, Galicia and the Canaries/Madeira/Azores.

However, with a larger sample size it may be possible to detect more structure (A. Natoli pers. comm.). Preliminary analysis of a current genetic study suggests no significant variation using microsatellite markers between common dolphins stranded along the Irish coastline or bycaught in the Irish tuna driftnet fishery off the south-west coast of Ireland (L. Mirimin, pers. comm.).

Current studies, from which only preliminary conclusions may be drawn, come from the studies of A. Viricel and V. Lahaye (pers. comm.), using samples collected from common dolphins stranded along the French coast and from material collected in oceanic waters as bycatch in the tuna fishery. Two markers operating at different time scales are being used: genetic markers (mitochondrial and microsatellites) corresponding to a time window of generations and heavy metals corresponding to a time window of about 15 years (mainly cadmium).

Based on the genetic markers, no difference has been found among stranded animals from shelf habitats over a range extending from the western Channel to the southern Bay of Biscay, which is consistent with previous findings. Among these animals, genetic diversity was found to be high (Nh for control region = 0.99 ± 0.01), suggesting a large effective population size. A comparison with offshore individuals has yet to be completed.

Heavy metals, accumulating in top predators via food intake, offer another view of possible stock structure, some operating at time scales of years to decades according to the half life time of these elements in target organs. In the case of the common dolphin, cadmium is mostly transmitted via the consumption of cephalopods, and most particularly oceanic cephalopods. The analyses of cadmium concentration in the prey of common dolphins and the analyses of stomach samples (see below), show that the exposure of oceanic common dolphin to this element via food is about 10 times the exposure in continental shelf habitats (Figure 4.1).

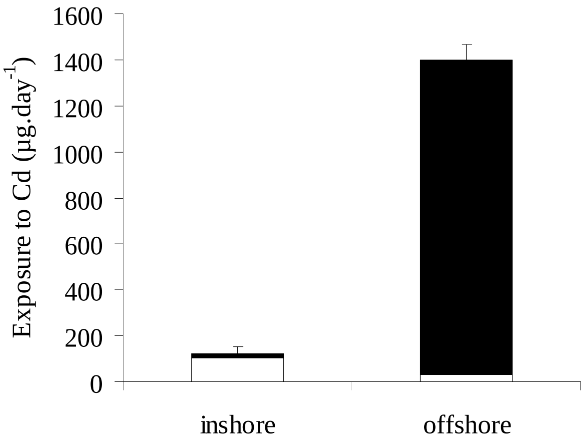

Figure 4.1: Common dolphin exposure to cadmium via food in continental shelf and oceanic habitats (black sections and bar indicating confidence limits)

If animals moved regularly between the two habitats at a time scale well below the half life time of this element in a given organ, this difference in exposure should be buffered and all animals should show the same rate of accumulation irrespective of the habitat where they were sampled. In contrast to this, common dolphins collected in oceanic habitat show a significantly higher rate of accumulation of cadmium in the kidney than individuals did from the continental shelf of the Bay of Biscay (Figure 4.2). This suggests that animals sampled in one of these habitats had essentially been foraging in this habitat for most of their life. Hence, some oceanic/continental shelf stock structure operates among north-east Atlantic common dolphins at least as long as individual life span.

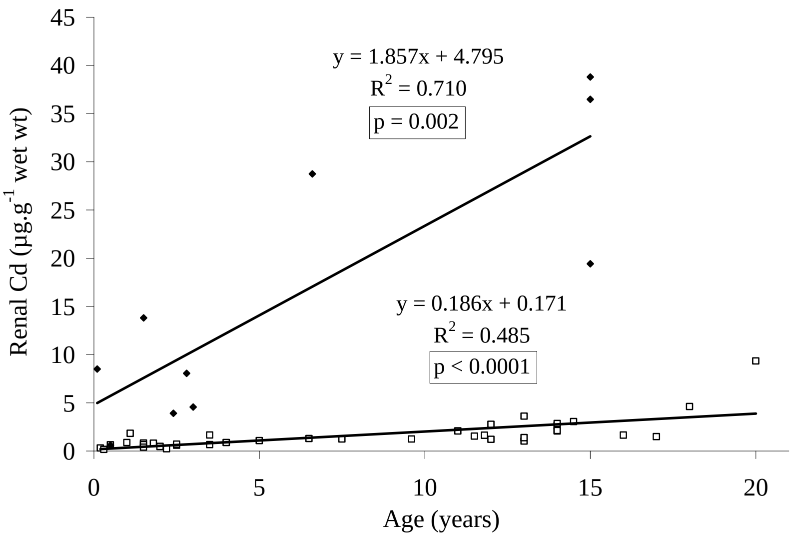

Figure 4.2: Accumulation rates of renal cadmium in oceanic (black diamonds) versus continental shelf (empty squares) common dolphins.

There is evidence from elsewhere that populations or stocks of common dolphins range over very large areas. Stock abundance estimates in the eastern tropical Pacific are for an area of 3.8 million km^2, while those in the western Atlantic are evaluated within an area of 573,000 km^2.

Further evidence supporting common dolphin populations encompassing large areas may be derived from the results of radio-telemetry – schools of common dolphins have been recorded ranging several hundreds of kilometres in a few days off the western USA (Evans 1982).

In summary, there are several independent bodies of evidence indicating that a single genetic population of common dolphins occurs off western Europe, with some evidence suggesting that offshore animals are separate from inshore animals over time scales at least as long as individual life times. There is some evidence of movement of common dolphins between the Mediterranean and nearby parts of the Atlantic.

4.2.2 Size of common dolphin population

4.2.2.1 Summer abundance surveys

There has been no single survey to assess the abundance of the common dolphin within its entire range of distribution off north-west Europe. Indeed, as noted above, the western boundary of the distribution is not well known, so designing a survey to cover the entire area of distribution would be difficult.

There were however three sightings surveys to estimate the abundance of cetaceans in the Celtic Sea and adjacent Atlantic waters in summer in the mid 1990s, one off Ireland in 2000 and one in the Bay of Biscay in 2002 (Table 4.1). A further estimate of abundance has been published by López *et al*. 2004 for waters off Galicia, based on observations made opportunistically from fishing vessels. The areas covered by these surveys are shown in Figure 4. 3.

The MICA survey was carried out by French scientists in 1993 and covered a large area from the continental shelf break westwards over deep water to the south and west of UK and Ireland (Goujon *et al*. 1993). The SCANS survey was carried out in 1994 by a multinational group of scientists and covered the Celtic Sea shelf area (Hammond *et al.* 2002) as well as the North Sea, overlapping with the MICA survey at the continental shelf break. Figure 4. 3 shows only the Celtic Sea section of the survey – the only area where there were sufficient observations of common dolphins for an abundance estimate to be calculated. No common dolphins were seen in the English Channel to the east of this area. The NASS survey was carried out by Faroese scientists in 1995 and covered two large areas (NASS east and NASS west) to the north and west of Ireland (Cañadas *et al.* in press). The 2000 survey (SIAR) was carried out over the shelf break north and west of Ireland in summer 2000 (Ó Cadhla *et al.* 2004). The ATLANCET survey covered 140,000 km^2 of the continental shelf and shelf break areas of the Bay of Biscay in summer 2002 with an overlap with the SCANS survey area in the southern Celtic Sea (Ridoux *et al*. 2003).

Table 4.1. Summary of encounter rate, density estimates and abundance estimates for common dolphins in five large-scale surveys in summer off north west Europe, 1993-1995, 2000, 2002 (from 1: Goujon *et al.* 1993; 2: Hammond *et al.* 2002; 3: Cañadas *et al.* in press; 4: Ó Cadhla *et al.* 2004; 5: Ridoux *et al*. 2003)

Survey Name	Year	Encounter rate individual (n/km)	Encounter rate group (n/km)	Density estimate (n.km^2)	Point (best) estimate (individuals)	Confidence limits (95%)
MICA[1]	1993	0.125	0.020	0.187	61,888	35,461 – 108,010
SCANS[2]	1994	0.101	0.009	0.374	75,449	22,900 – 248,900
NASS[3]	1995	0.198	0.024	0.555	350,696	210,958 – 539,926
SIAR[4]	2000	0.070	0.017	0.039	4,496	2,414 – 9,320
ATLANCET[5]	2002	0.237	0.016	0.126	17,639	11,253 – 27,652

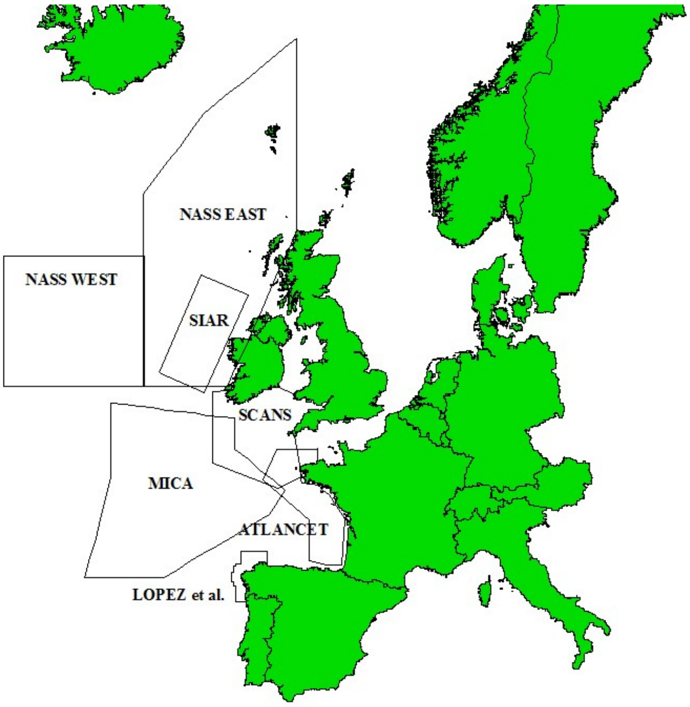

Figure 4.3. Areas of summer abundance surveys off north-west Europe, see text and Table 4.1 for details.

One estimate has been made for an area in the north-east Atlantic range away from those areas for which a formal abundance survey has been made. López *et al*. (2004) suggest that 7,000-10,000 common dolphins may be present in waters immediately off Galicia, but this figure was based on comparison of relatively sightings rates between areas of known density and rates from surveys made from fishing vessels. Line transect methods were not used.

4.2.2.2 Winter abundance surveys

Only two abundance estimates have been made in winter off north-west Europe (Figure 4. 4, Table 4.2). The shelf waters of the Bay of Biscay were surveyed aerially on a monthly basis from October 2001 to March 2002 as part of a monitoring programme (ROMER) aimed at assessing population size and habitats of wintering seabirds following the *Erika* oil spill (Bretagnolle *et al.* 2004). These flights were not initially designed to census cetaceans, but allowed the first series of distribution data to be collected in the area (G. Certain, pers. comm.).

WDCS/Greenpeace surveyed a relatively small area offshore of the Lizard and Start Point in Cornwall (de Boer *et al.* 2005) in winter 2004. The surveys of López *et al.* (2004) were made throughout the year and these authors do not make separate seasonal estimates of numbers.

Table 4.2. Summary of abundance estimates for common dolphins in three surveys off north-west Europe,

Survey Name	Year	Point (best) estimate (individuals)	Confidence limits (95%)	Source
Galicia	1998-99	7000- 10000		López *et al*. 2004
ROMER	Nov 2001 Feb 2002	3609 7953	1649 - 7896 4541 - 13928	Bretagnolle *et al.* 2004
WDCS/Greenpeace	Jan-March 2004	9708	4799 - 19639	de Boer *et al*. 2005

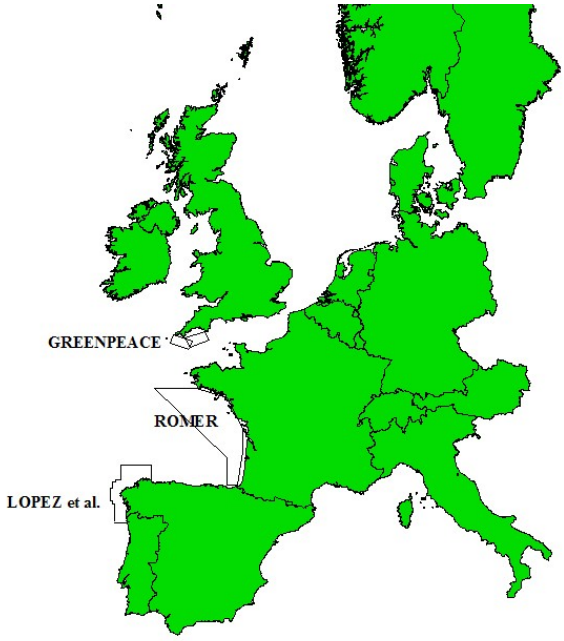

Figure 4.4. Areas of winter abundance surveys off north-west Europe, see text and Table 4.2 for details.

4.2.2.3 Correction for bias in abundance surveys

Each of the large-scale summer surveys derived an estimate of the abundance of common dolphins within their survey areas (Table 4.1). These figures are though potentially biased and have a degree of statistical uncertainty attached. The statistical uncertainty surrounding these figures for summer surveys indicated by confidence limits in Table 4.1. There is 95% certainty that the actual number of common dolphins present in the area should lie between the two figures given in Table 4.1. Bias is less easy to account for. Bias artificially increases (positive bias) or decreases (negative bias) the best estimate and its associated confidence limits.

Three main sources of bias have been encountered in abundance surveys.

i. a bias caused by not detecting all animals on the track line;

ii. a bias caused by non-random responsive movement of animals before they are detected on the survey;

iii. a bias caused by non-representative placement of survey lines within the area being studied.

An estimate based on surveys that miss some animals on the track line will be negatively biased. If animals move generally towards a survey vessel before being detected, the estimates will be positively biased, while if animals move away before being detected, then the estimate will be negatively biased. The bias caused by non-representative placement of survey will depend on whether disproportionately more survey effort was placed in preferred dolphin habitat (positive bias) or in areas not preferred by dolphins (negative bias).

The first form of bias can be estimated by techniques that examine how well observers are detecting animals – often this is carried out using two independent sets of observers on the survey vessel counting the same strip of sea. The second form of bias requires an attempt (usually using very high powered binoculars) to detect dolphins before they respond to the survey vessel; information on the direction that distant dolphins are swimming in can also be used. The third form of bias may be avoided by good design and execution of the survey or sometimes through post-hoc stratification when carrying out abundance estimation.

Efforts to detect and compensate for these biases varied among the three main surveys outlined above (Table 4.3). The WDCS/Greenpeace winter survey assessed all forms of bias but covered only a very small part of the range of the common dolphin population.

Table 4.3. Sources of bias in abundance estimates of common dolphins off north-west Europe in summer, and whether they have been allowed for in published abundance estimates.

Survey	Source of bias		
	Detection on line	Responsive movement	Survey representative
MICA	No	No	Yes
SCANS	No	No	Yes
NASS	Yes	Yes	?
SIAR	Yes	No	(Yes)
ATLANCET	No	No	Yes

Goujon (1996) calculated, after taking account of the overlap between survey areas, that the MICA and SCANS surveys combined abundance estimate was approximately 120,000 common dolphins. The abundance estimate for NASS can be added to this estimate to give a minimum number of common dolphins in the combined area of the surveys. Goujon (1996) for mathematical reasons did not present 95% uncertainty limits and it is not possible, nor mathematically sensible, to provide these figures for the combined area of the survey.

The three sources of bias need to be taken as far as possible into account in reaching a 'best estimate'. The first of the sources of bias listed above is compensated for in assessments of abundance by inflating the raw abundance estimate to allow for those animals likely to have been missed. Only one survey (NASS) has been able to allow for this bias in deriving an abundance estimate. This survey estimated that only 0.79 of the animals actually on the trackline were detected. This factor has been used in Table 4.4 to raise the uncorrected combined estimate for the MICA and SCANS surveys for illustrative purposes. How applicable this correction is to these surveys is unknown but it is the best and only estimate available. What is known is that the bias from this factor in these surveys will be negative.

Similarly, only the NASS survey has been able to estimate the bias caused by the movement of dolphins prior to their detection by the surveyors. Before correcting for the bias of positive movement, the uncorrected estimate was 3.7 times larger than the corrected NASS estimate. The scale of this bias is consistent with our knowledge that common dolphins are attracted towards ships and that bow-riding is a common and normal behaviour for this 'playful' species.

If common dolphins are attracted to survey vessels throughout their north-east Atlantic range, we would expect all abundance estimates for parts of this range to be biased upwards. The scale of this bias will depend on the behaviour of the animals in response to the particular survey vessels, which might be influenced by the density of all shipping in the area, and by data collection methods. There is no information to modify the NASS correction factor, which has therefore been used in Table 4.4 to suggest a correction of the numbers estimated by the MICA and SCANS surveys. Data collection methods in the SCANS survey at least (where some observers searched as far ahead as possible) were such that the scale of this bias is probably overestimated.

Table 4.4. Bias-corrected abundance estimates of common dolphins off north-west Europe from four surveys in summer. The MICA/SCANS figure is the combined figure estimated by Goujon (1996).

Survey	Point (best) estimate prior to taking account of other bias (individuals)	Allowing for bias caused by insufficient detection on line (x1.27)	Allowing for positive responsive movement (x0.27)

MICA/SCANS	120,000	152,400	41,148
NASS (corrected)	350,696	350,696	350,696
ATLANCET	17,639	22,401	6,048
Totals	470,696	525,497	397,892

4.2.3 Status and trends of common dolphin

It is not possible at present to determine any trends in overall common dolphin abundance in waters off north-west Europe. This may change partly following the forthcoming SCANS II survey of cetaceans in waters over the continental shelves of north-west Europe. This survey will not however cover the (large) parts of the range of common dolphins over deeper waters further offshore.

The status and trends in other aspects of common dolphin biology are reviewed in the next eight sections.

4.2.4 Current distribution and seasonal movements

Records of sightings of common dolphins made systematically have been gathered into UK's Joint Cetacean Database. These records include only those made with associated searching effort records, and while noting environmental conditions. Data held in this database may be analysed to give a representative picture of the distribution of cetaceans in waters around the UK and other nearby areas. Knowledge of the amount of effort (length of time) that has been spent in making observations is essential in order to avoid merely illustrating where observers happen to have looked hardest. The contents of the Joint Cetacean Database up until 1998 have been analysed and published (Reid *et al.* 2003). Only part of the data known to be available beyond 1998 has been added to the database and negotiations continue to gain access to the remaining data.

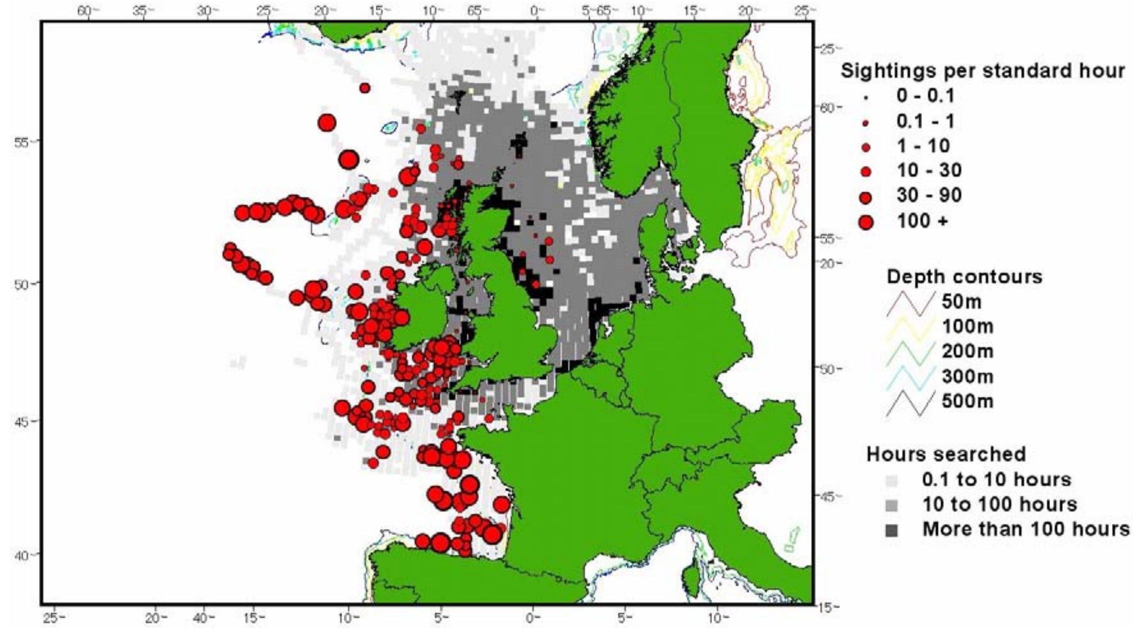

Figure 4.5. Sighting rates of common dolphins from May to October. Records from 1979-1998. Red circles are scaled in proportion to the number of animals observed per hour of observation. Sightings rates have been standardised for observations made under different sea conditions. They have not been corrected for the differing efficiencies of the various people and vessels used to collect the data. The shaded cells underlying the red circles provide a crude indication of how much observation 'effort' went into each cell in each of the two seasons (Data from Joint Cetacean Database).

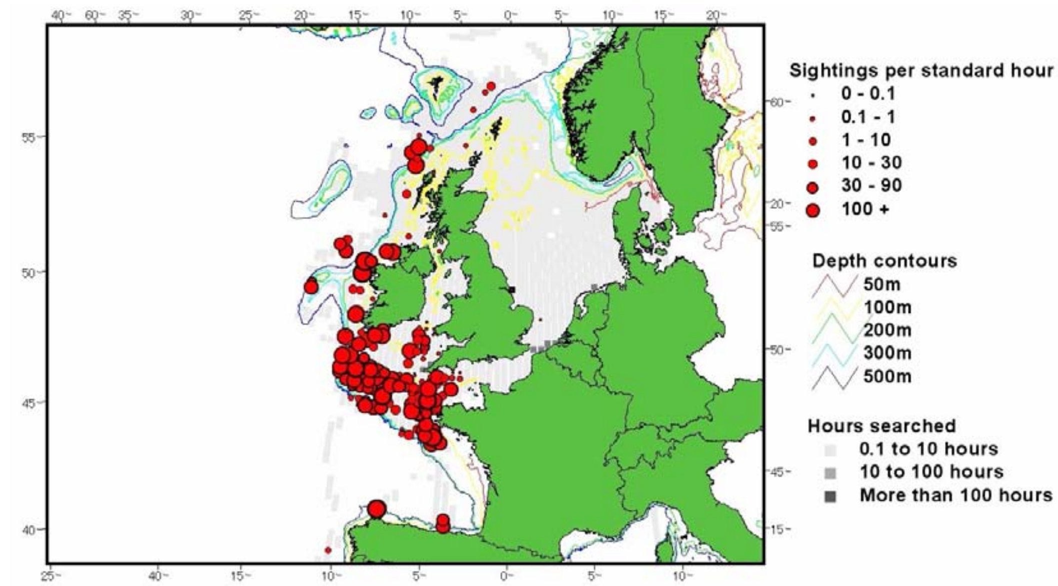

Figure 4.6. Sightings rate of common dolphins from November to April –see caption to Figure 4. 5 for further explanation.

There is some evidence of seasonal movement of common dolphins, with dolphins being more widely spread, especially in offshore deeper waters in summer (Figure 4.5) than in winter (Figure 4.6), when there is a pronounced concentration in the shelf waters of the Western English Channel and further offshore parts of the Celtic Sea.

Seasonal changes in the distribution of common dolphins in western European waters have been documented (Figures 4.5 and 4.6). During July to September, an increase in the relative density of common dolphins has been reported in the Irish Atlantic margin (Ó Cadhla *et al.* 2004). As described in Section 1.2.2, the NASS-95 survey estimates an abundance of 350,696 common dolphins. Although the majority of animals were encountered in survey areas NASS West, with no animals sighted north of 56º41'N. The West survey area of NASS is not only the most offshore area to be surveyed in the north-east Atlantic, but it has also has reported the highest density of common dolphins compared to any other area surveyed in the north-east Atlantic. The survey was carried out in July and August, and common dolphins were reported as far offshore as approx. 25ºW. Also during the summer months (May to September) common dolphins were regularly caught in drift nets for albacore tuna operating offshore, ranging from approx. 40º-52ºN. Throughout the same period a substantial aggregation of common dolphins was also recorded in the northern Biscay shelf slope, with a peak in sighting reported in July (Brereton *et al.* in press).

Following this, during the autumn/winter period, inshore movements of common dolphins take place, and they inhabit waters off the west and south coasts of Ireland, the south-west coast of the UK, with large aggregations reported in the Celtic deep and in the western English Channel (Evans 1992; Pollock *et al.* 1997; O'Cadhla *et al.* 2004; Figure 4.6). Pollock *et al.* (1997) reported that between November and December densities of common dolphins were most concentrated, with the majority of common dolphins inhabiting waters between 100m isobath and the shelf break (200m). Brereton *et al.* (in press) also reported that the shallow Brittany coast and western English Channel supports large numbers of common dolphins during the winter months (December to February), with a peak in sightings in December. Northridge *et al.* (in prep.) reported that the winter encounter rates in VIIe are about six times those recorded in the summer for the same area. Further evidence of the movement of common dolphins from offshore waters is available from the Seawatch Foundation's cetacean database, which shows a distinct peak in off shelf records during late summer (Figure 4.7).

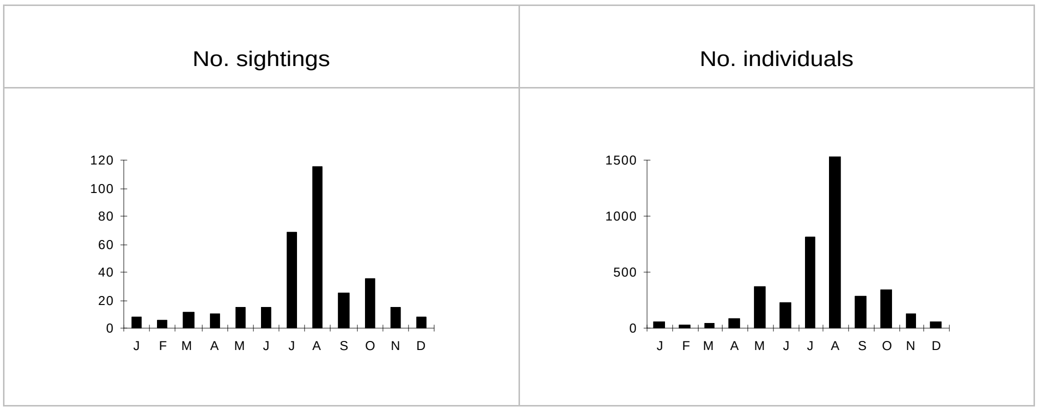

Figure 4.7. Number of sightings/individuals recorded in 'off shelf' waters from Seawatch Foundation database. These records will be biased by some variation in effort, but perhaps less than from other sources as many derive from observations made by weather ships, where observers are likely to be more systematic throughout the year.

Common dolphins are extremely mobile and radiotelemetric data have recorded swimming speeds of 0.77 to 3.20 nautical miles per hour (Evans 1982). The only study that tracked the movements of common dolphins used radio transmitters, and was carried out in the eastern Pacific Ocean. Results showed that ten days after a female common dolphin was fitted with a transmitter, the dolphin had travelled about 270 nautical miles from her point of release (Evans 1982). However, in the north-east Atlantic, data on the habitat range of individual common dolphins is unknown, but distributional data suggests large-scale seasonal movements.

4.2.5 Longer-term changes in distribution

There is some evidence of medium to long term changes in the distribution and range of common dolphins. Figures 4.5 – 4.7 represent an average over a number of years, coverage is not uniform between years and there is variation in common dolphin distribution between individual years. The Joint Cetacean database was examined to compare two periods, 1980–1993 (Figure 4.8) and 1994–1998 (Figure 4.9) (Northridge *et al.* in prep). Comparing the later period indicates more recent denser concentrations of animals around Brittany, with fewer animals further west on the continental shelf. This suggests an eastward movement between these time periods. There is also the suggestion that common dolphins are more frequent in winter months further east in the Channel in the 1994–1998 period, compared with earlier years. This pattern can also be seen if sightings are binned by longitude in the western Channel and the two periods are compared (Figure 4.10). The Joint Cetacean Database has not been fully updated with post 1998 data, and the WDCS/Greenpeace survey (de Boer *et al.* 2005) only covered a relatively small part of the English Channel at a high enough survey intensity and thus is not useful in indicating whether or not this pattern has persisted into more recent years. The WDCS/Greenpeace survey did not visit the area off Brittany that held concentrations in 1994–98.

Groups of common dolphins have been sighted in recent years in the central English Channel (just east of 2º45'W), with group sizes of ca. 2000 individuals recorded on occasions, although most groups comprised of less than 10 individuals (Brereton *et al.* in press). Since 1990, there has been a notable increase in the number of common dolphins that have been reported stranded along the southwest of the UK, especially during the winter period (Northridge *et al.* in prep and see Section 1.2.6.1).

This increase in strandings has been attributed to a number of factors including

1. an increase levels of reporting of dead animals;
2. stronger onshore winds during the wintertime depositing more animals along the coastline; and
3. an increase in the numbers of common dolphins that are incidentally caught in fishing nets and washed ashore, since the introduction of pelagic trawling for several species of finfish in the late 1980s (Gosselin 2001; Northridge *et al.* in prep).

However, the increase in strandings could also be due to an increase in the relative numbers of common dolphins in this region during the winter period. Between 1990 and 2000, the SST increased in the western English Channel by approx. 1ºC, which exceeded any other SST change in the area over the last 100 years (Hawkins *et al.* 2003). The possible increase in the relative density of common dolphins in the western English Channel during the winter period could be related to increasing local water temperatures, and what effects this has on the prey species of the common dolphin, but this all needs to be investigated further.

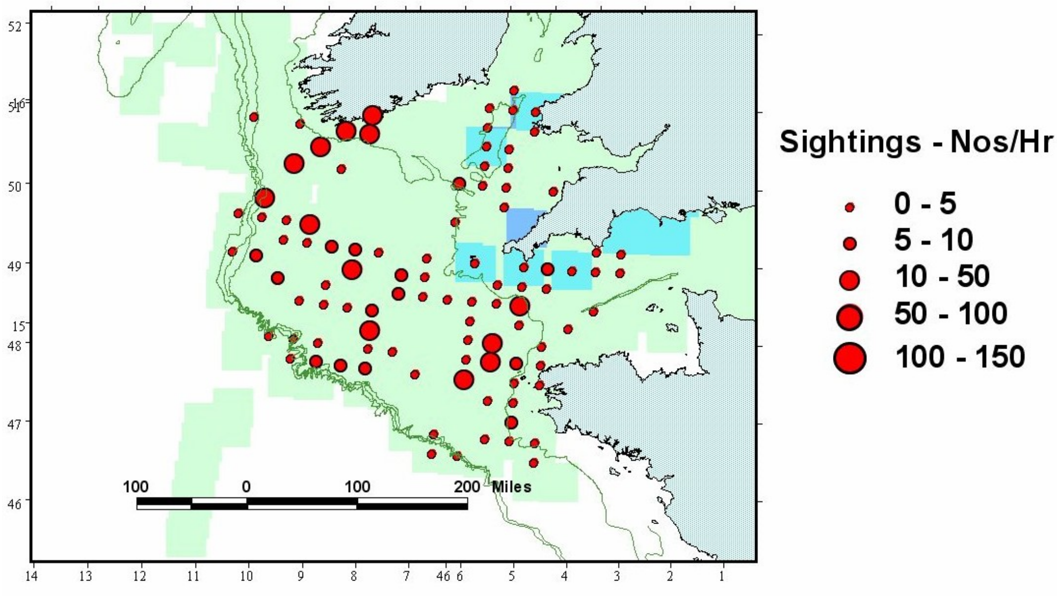

Figure 4.8. Winter (November – April) distribution of common dolphins on Celtic Shelf, 1979–1993 (data from Joint Cetacean Database)

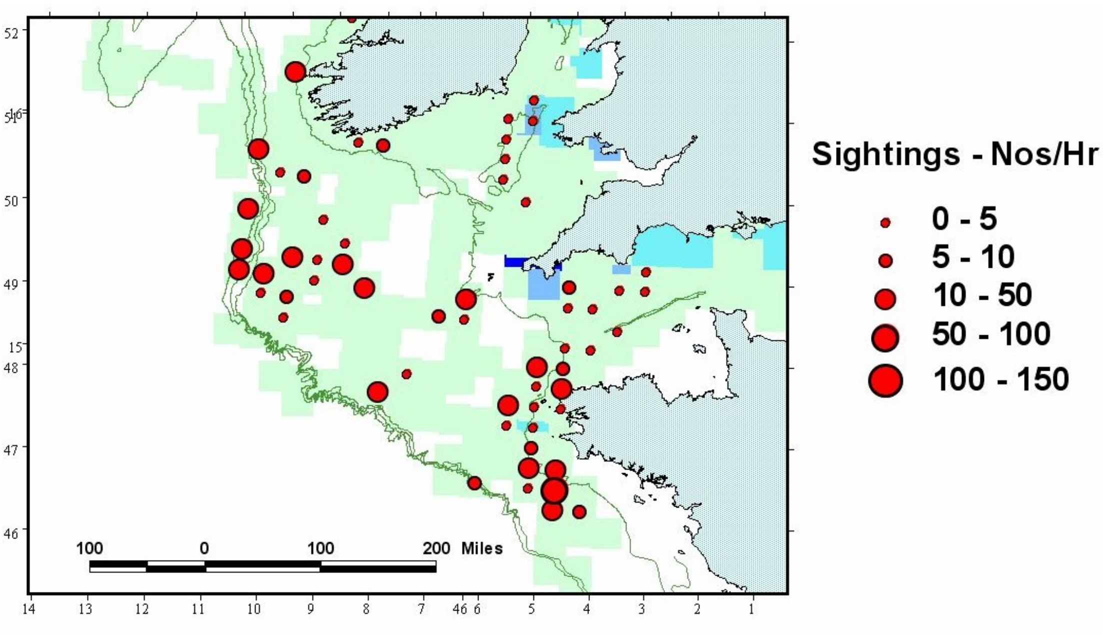

Figure 4.9. Winter (November – April) distribution of common dolphins on Celtic Shelf, 1994–1998 (data from Joint Cetacean Database)

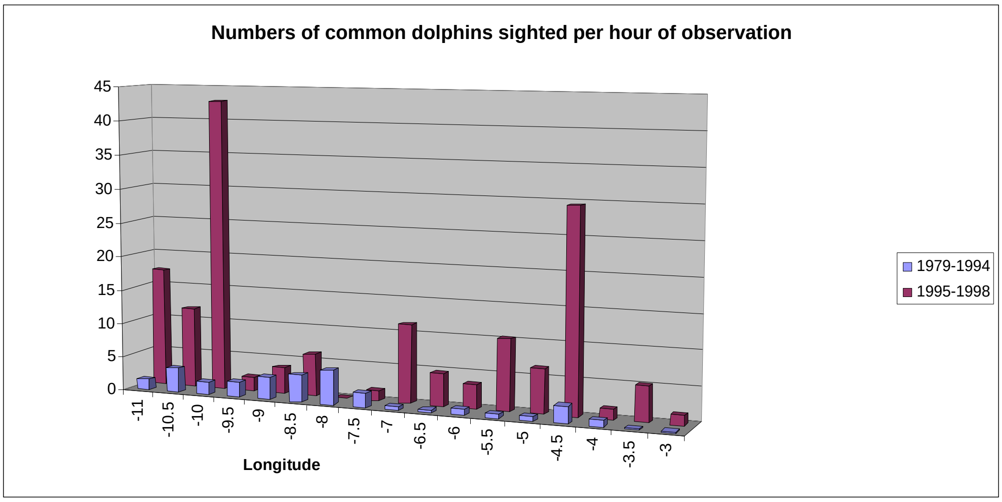

Figure 4.10. Comparison of the number of common dolphins sighted per hour of observation in the western English Channel for two time periods (1979-1994 and 1995-1998). Sighting rates were standardised to account for sea state.

Food availability plays a very important role in the distribution of the common dolphin, and a combination of climatic fluctuations and over-fishing in certain regions may lead to local reductions in common dolphin abundance or major distributional shifts. This also has implications for conservation and management as it makes the identification and protection of suitable habitats, or critical habitats very difficult (Murphy 2004a).

Changes in the distributional range of common dolphins in the North-east Atlantic have occurred over the last 100 years. As mentioned before, throughout the NASS-95 survey common dolphins were not sighted north of 56º41'N, whereas the SCANS survey carried out in 1994, only reported one sighting of this species above 52ºN, around approx. 58º30'N (north of Scotland), despite large observational effort up to 62ºN (Cañadas *et al.* in press). However, in recent years, the relative abundance of common dolphins has increased in waters off the north-west coast of Scotland. Sightings surveys conducted in May–September 2002 and 2003 showed that the relative occurrence of common dolphins has increased off the north-west Scottish coast, in comparison to previous studies, attributed to increasing water temperatures (MacLeod *et al.* 2005). There has also been an increase in the number of common dolphin strandings in this northern area since 1992.

In recent years, sightings of common dolphins were infrequent in the North Sea, with the majority of sighting reported in the northern North Sea, from June to September (Reid *et al.* 2003). Sightings were also rare in the eastern section of the English Channel and the southern North Sea. However, between the 1920s and 1960s common dolphins migrated into the southern and eastern North Sea, and there was an increase in reported strandings of common dolphins along the Dutch (a modal peak occurred 1945-1950, n= 25; Bakker and Smeenk 1987) and the Danish coastlines (1937-1952; total number of strandings, n = 10; Kinze 1995).

This increase in strandings along the Dutch and Danish coastlines coincided with a decline in strandings (1930s to 1970s) along the Irish and the English coasts, which strongly suggests a shift in the distribution of this species in the western European waters at this time (Evans and Scanlan 1989, and references therein; Murphy 2004a). The decrease in strandings along the English coastline coincided with changes in fish stocks off the southwest coast of England. During a warming period in sea surface water temperatures (SST) between the 1920s and the 1960s, available nutrients, herring *Clupea harengus* and whiting *Merlanguis merlangus* (along with other fish species) decreased in abundance (Southward 1963; Evans and Scanlan 1989). During the increased SST in the 1930s, fish species shifted their distribution northwards, and it is believed that common dolphins followed (Fraser 1934; Evans and Scanlan 1989). Since

1965, a change in plankton abundance has occurred off the southwest coast of England, and many of the conditions prevailing in the 1920s returned, along with an increase in strandings of common dolphins along the southwest coast of England (Evans and Scanlan 1989) and the southern and western coasts of Ireland (Murphy 2004a).

It is not known whether the common dolphin entered the southern and eastern North Sea via the English Channel or from the Northern North Sea during the 1920s-60s, although both possibilities could have occurred. However, it appears that common dolphins are again moving into the eastern North Sea, with sightings and strandings of common dolphins in Danish waters, and along the Danish coastline. Six common dolphins have stranded along the Danish coastline between 2001 and 2003 and smaller schools containing up to 10 individuals have been sighted (C. Kinze, pers. comm.). Prior to this, the last reported stranding of a common dolphin was in 1978 (Kinze 1995), whilst sightings of the species have been reported for the years 1979, 1982, 1990 and 1996 (C. Kinze, pers. comm.). Also, between 1993 and 2001 there have been sightings and strandings of common dolphins in Swedish, Norwegian, German, Polish and Finnish waters (C. Kinze, pers. comm.).

4.2.6 Life history

Murphy (2004a) used Gompertz equations to describe growth in male and female common dolphins. Asymptotic values obtained for total body length were 211.6 cm for males and 197.4 cm for females (Figure 4.11). Asymptotic lengths were attained at approximately 11 years of age in males and 9 years in females. Table 4.5 outlines the biological parameters of common dolphins in the north-east Atlantic. Murphy (2004a) found that the average age attained at sexual maturity (ASM) in male common dolphins was 11.86 years (SE = 0.62). However, in the age classes 8 to 13 years both immature and mature male common dolphins were found in the sample. The ASM in females could not be obtained due to a lack of young mature females ranging from 9-11 years, although it was proposed that sexual maturity in female common dolphins is possibly attained in some individuals between 9 and 10 years (Murphy 2004a). Similar findings were found when common dolphins off the French coast were analysed. By determining age from tooth section and female reproductive status from gonadal macroscopic examination, it was possible to identify the main characteristics of the female reproductive development. All animals under 5 years old are immature. From ages 5 to 8 years, an increasing proportion of individuals reach puberty, as indicated by the presence of mature follicles. Past ovulation, shown by the presence of *corpora albicancia*, and pregnancy, indicated by the presence of either a *corpus luteus* or a foetus, can be observed from 9-21 years old. Age at sexual maturity is thus 9 years old. Given the low number of females studied in each age class it is not yet possible to identify periods of higher fecundity within reproductive life span of the females.

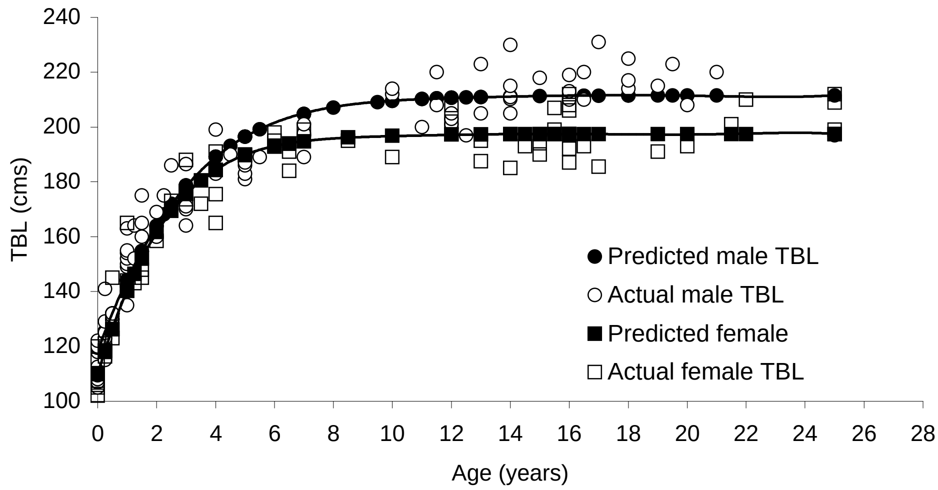

Figure 4.11. Gompertz growth curves superimposed on length-at-age data for female common dolphins (n= 59) and male common dolphins (n=92).

Table 4.5. Biological information on common dolphins inhabiting waters off the Irish coast. TBL = total body length; * = asymptotic values obtained using Gompertz growth curves; **including samples from French bycatch and stranding common dolphins, n.a. = not available/applicable (Murphy (2004a).

	Male (n)	Female(n)
Min TBL	105	93 cm
Max TBL	231 (233**) cm (137)	231 cm (100)
Min Age	Neonate	Neonate
Max Age	25 (28**) (104)	25 (72)
TBL range of sexually immature individuals	**105-220 (159)	93-206 cm (35)
TBL range of sexually mature individuals	**195-233 (45)	183-216 cm (37)
Age range of sexually immature individuals	0-13 (148)	0-10 yrs (32)
Age range of sexually mature individuals	8-28 (38)	12-25 yrs (24)
Average age attained at sexual maturity (ASM)	11.86**	n.a.
Age at physical maturity*	11 (12**) yrs (92)	10 yr (59)
TBL at physical maturity*	211.6 cm (92)	197.4 cm (59)
Mean length in physical mature individuals	212 cm	199.1 cm
Combined testes weight range in sexually mature individual	415.9-5000 g	n.a.
Gestation period	n.a.	11.5 months
Annual Pregnancy Rate (APR)	n.a.	28.2%
Calving Interval (CI)	n.a.	42.5 months (3.5 yrs)
Lactation period	n.a.	10.35 months (0.86 yrs)
Resting Period	n.a.	20.7 months (1.7 yrs)

Murphy (2004a) reported that in both male and female common dolphins large individual variation was observed in the age at which sexual maturity was attained. In cetaceans, there may be many factors determining when an individual attains sexual maturity, such as the general health of the animal, but also hierarchical position, genetic factors, quality and quantity of food, and consumption of food high in contaminant levels, (especially endocrine disrupting chemicals) may all have a synergistic effect (Murphy 2004a). Therefore, it is important when assessing population reproductive parameters to calculate the average age attained at sexual maturity, and not just produce an age or body length range for sexual maturity. Unfortunately, Collet (1981) had not calculated the ASM of her sample, and therefore we cannot compare our data with the earlier French study in the 1970s.

The range obtained for age at sexual maturity by Murphy (2004a) does appears higher than the previous values obtained by Collet (1981) of 5 to 7 years (Murphy 2004a). However, in Collet's study age at sexual maturity was estimated using age and body length graphs, as the age was not determined for animals whose gonads were analysed. A more recent study on common dolphins bycaught off the French Atlantic coast by Goujon *et al*. (1993), found that sexually immature and sexually mature individuals range from ranged 0-11 years (n = 37), and 10-23 years (n = 10), respectively. Pierce *et al*. (2004) estimated the overall ASM in female common dolphins in the North-east Atlantic was 8 years (SE = 0.69), however most of the mature individuals in the sample were older than 10 years of age (89%). The ASM was estimated at 10 years (SE = 0.47) for the French sample, and at 8.5 years (SE = 0.5) was estimated for the Spanish sample (Pierce *et al*. 2004).

A study carried out by Danil (2004) in the eastern tropical Pacific on reproductive parameters in the female common dolphins incidentally killed in the tuna purse seine fishery, calculated the ASM at 8.1 years, but on average females gave birth after 10 years of age. Due to a lack of juveniles in the sample, the ASM was recalculated, and an estimate 10.2 years was obtained (Danil 2004). The maximum age obtained was 25 years and asymptotic length was attained at 196.5cm.

Analysis of male gonadal material was undertaken by Murphy *et al*. (in press) and samples examined were from common dolphins stranded along the Irish and French coastlines, and bycaught in the Irish and French tuna driftnet fisheries (see Figure 4.12). By comparing slope of the regression lines (total body length on age) between samples, a significant variation was found between the growth coefficients of the Irish stranding sample and the French stranding sample at $P<0.05$ but not at $P<0.01$ ($t = 2.365$, 65 d.f.). This suggests possibly differences in growth rates between common dolphins inhabiting waters in the inner Bay of Biscay, and off the south and west coast of Ireland.

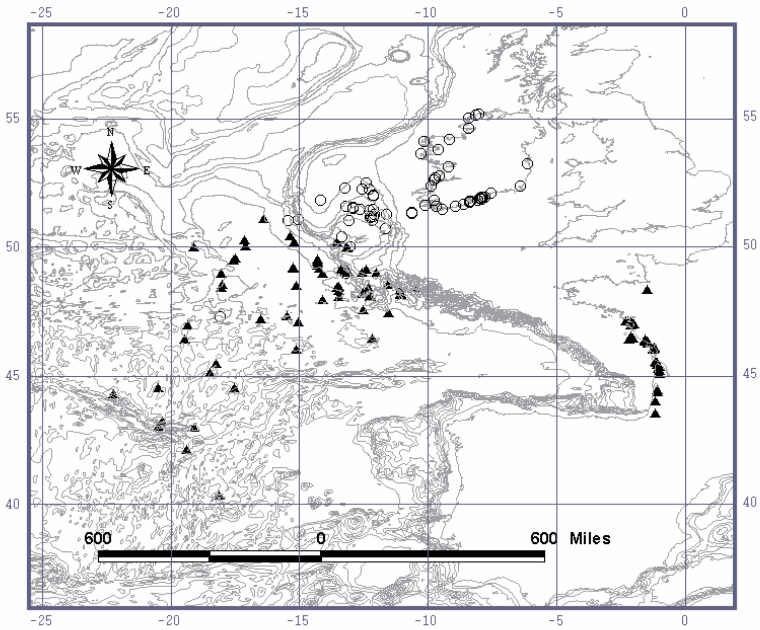

Figure 4.12. Distribution of sampling locations in the North-east Atlantic, of male common dolphin whose gonads were analysed, samples were obtained from the Irish and French strandings projects and bycatch observer programmes with tuna drift net fisheries (closed triangles = French data, open circles = Irish data. (Murphy et al. in press).

Reproductive seasonality was found to occur in the common dolphins in the North-east Atlantic, and the mating and calving periods were estimated to take place during the months May to September, evidenced by marked seasonal changes in both the mass and cellular activity of testes (males), and the presence of ovulating and recently pregnant females during that time (Murphy 2004a). Sightings of common dolphin groups containing newborn calves occur off western Ireland during the months July to September (Ó Cadhla *et al.* 2004), and initial sightings of calves in the Bay of Biscay have been reported in May, with peak sightings of in July (Brereton *et al*. in press). Furthermore, in July very large groups were noted in the Northern Biscay shelf slope (statistically significant higher groups sizes than the rest of the year; Brereton *et al*. in press), which coincides with the mating period of the common dolphin (May to September).

Overall Murphy (2004a) estimated the pregnancy rate in common dolphins off the Irish coast was 28.2%. The sample was composed of common dolphins that stranded along the Irish coastline, and also animals that were incidentally caught in the Irish tuna fishery. Similar results were calculated by Pierce *et al.* (2004), as the overall pregnancy rate for the North-east Atlantic (samples analysed obtained 2001-2003, n = 95 mature female common dolphins) was estimate to 30%. In common dolphins off the Irish coast, a calving interval of 42.6 months, a lactation period of 10.4 months, a resting period of 20.7 months, and a gestation period of 11.5 months was also estimated (Murphy 2004a). Danil (2004) calculated a calving interval of approx. 2.5 years, and an annual pregnancy rate of 47% for common dolphins in the eastern tropical Pacific. Unlike in the north-east Atlantic the calving period extends all year round, and some females were simultaneously pregnant and lactating (26.8%), therefore shortening their calving interval. In the eastern tropical Pacific there are a number of reasons why common dolphins are able to reproduce all year round, but all are mainly due to upwelling modified areas that the dolphins inhabit in this region. These areas provide an environment that is more stable throughout the year in terms of environmental parameters, food availability, and predation risk, all factors that typically affect movement patterns. Furthermore, females could exploit this stability and meet the energetic demands of pregnancy and lactation year-round (Danil 2004). The lower pregnancy rates in the north-east Atlantic compared to the eastern tropical Pacific may be due to the contaminant load of common dolphins in European waters (see Section 1.2.10).

In the north-east Atlantic, if sexual maturity is attained between at 9-10 years of age and common dolphins live for about 25 years, the maximum lifetime female reproductive output would be approximately four calves (Murphy 2004a). Large variations in female common dolphin reproductive output probably occur, which could be a consequence of:

(1) variation in age at attainment of sexual maturity between individuals;

(2) health status of the female, as some females may not be capable of conceiving or carrying foetuses to full term due to disease, infection, or other pathological reasons;

(3) length of time before a young mature female attains the status of breeding "cow" within the social structure (thus attaining social maturity);

(4) the possibility of breeding and non-breeding mature females within a group; and

(5) fisheries killing large numbers of neonates and calves, which would result in females shortening their calving interval and mating during the following oestrus/mating period

or a combination of these factors (Murphy 2004a).

4.2.7 Strandings of common dolphin

In the North-east Atlantic, strandings have shown a consistent spatial and seasonal pattern for common dolphins with pronounced winter peaks identified in strandings records for the UK and Ireland, and the Atlantic coasts of France, Spain and Portugal (Tregenza and Collet 1998; López *et al.* 2002; Sabin *et al.* 2002; Silva and Sequeira, 2003; Murphy 2004a). A large number of the common dolphins that strand during these winter peaks exhibit signs of bycatch (Kuiken *et al.* 1994; Tregenza and Collet 1998; Murphy 2004a). However, post-mortem examinations of known bycaught animals has shown that some fishing gears (e.g. pelagic trawl nets) may not leave any characteristic external markings of bycatch on dolphin carcases (Kuiken *et al.* 1994). Therefore for stranded dolphins where cause of death could not be determined, the possibility of bycatch cannot be ruled out. It is not possible at present, on the basis of evidence from stranded animals, to draw up a full list of fisheries that cause common dolphin bycatch, or to rank any gear in terms of risk to common dolphin.

4.2.7.1 Strandings of common dolphins on UK coasts

In the UK, the majority of common dolphin strandings occur on the coastlines of Cornwall, Devon and Dorset (Gosselin 2001). Since 1990, there has been an increase in the number of common dolphins reported stranded on these south-west coasts of the UK, with increased levels of strandings in 1992, 2001, 2002, 2003 and 2004 (Figures 4.13, 4.14). Entanglement in fishing gear was the most common cause of death accounting for 57.9% (n = 179) of the total number of strandings (n = 302) on the English coastline from 1990 to 2003. 77.8% of dolphins diagnosed as bycatch stranded during the period January to March and 97.7% of these strandings occurred along the Southwest coast of the UK (Deaville *et al.* 2003). The proportion of common dolphins diagnosed as bycatch in 2004 has not been calculated yet. Peak years of bycatch (>40%) between 1990 and 2000 occurred in 1991, 1992, 1993, 1996, 1997 and 2000 (Figure 4.15, Gosselin 2001).

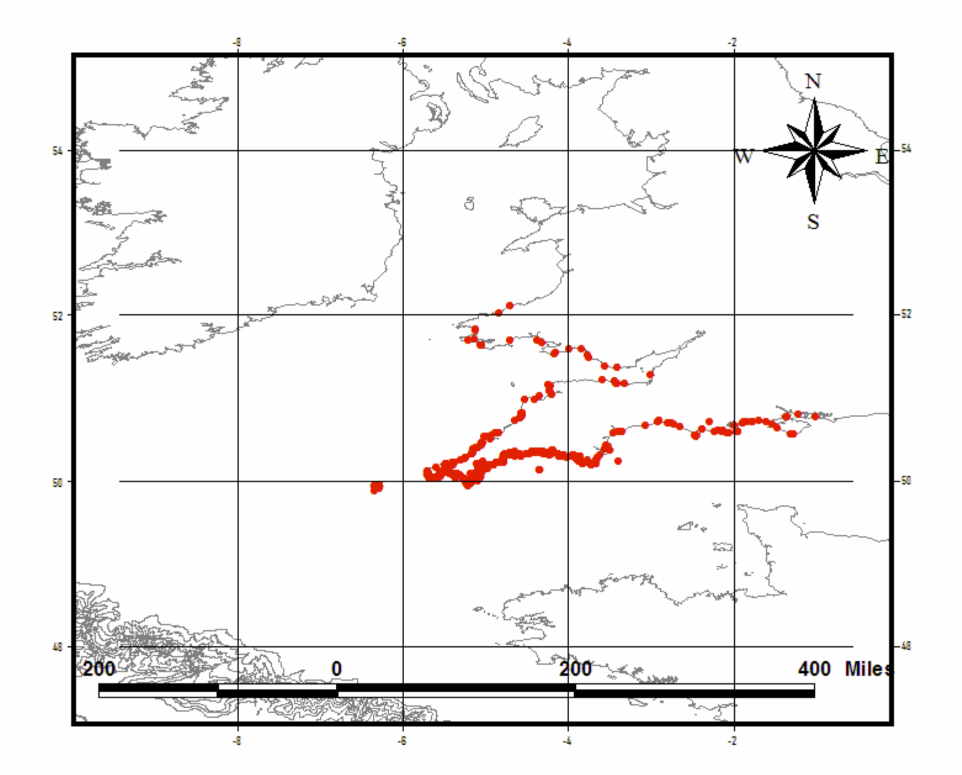

Figure 4.13. Distribution of common dolphins stranded in winter (November to April) on the south-west coasts of the United Kingdom, 1990-2004.

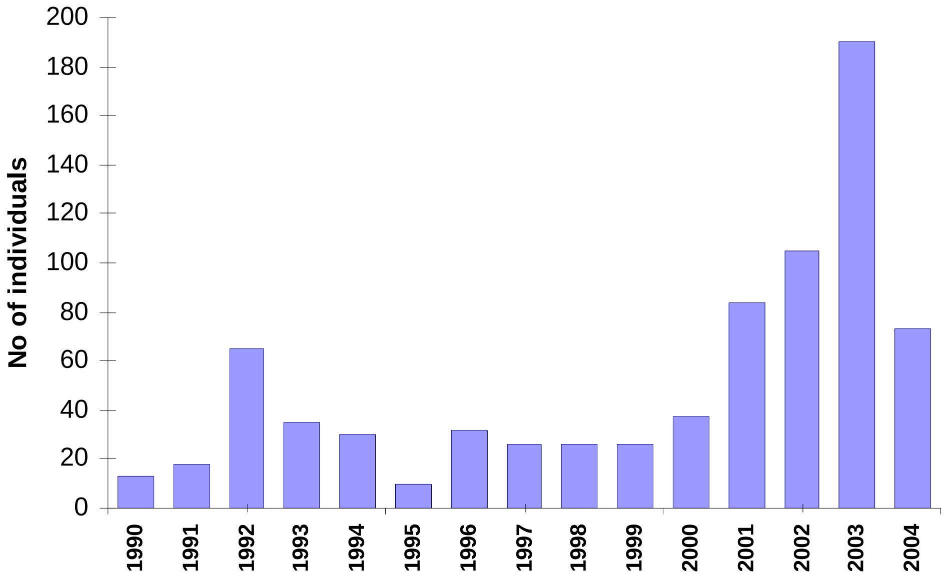

Figure 4.14. Number of common dolphins stranded in winter (November to April) on the south-west coasts of the United Kingdom, 1990-2004.

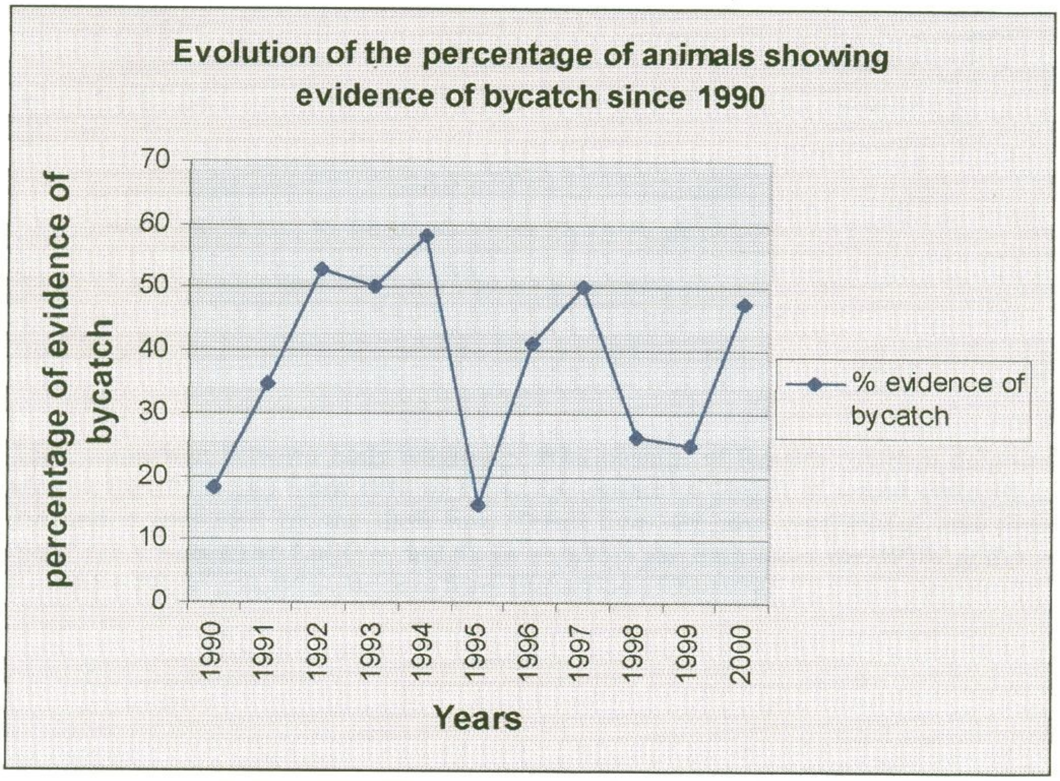

Figure 4.15. Percentage per year of stranded common dolphins in the UK that showed evidence of incidental capture in fishing gears (Gosselin 2001).

In 1992, 118 dolphin carcasses were washed ashore along the south-west coast of the UK, 54 of which were identified as common dolphins. The majority of the carcasses showed evidence of incidentally being caught in fishing nets. Furthermore, they also had food remains of partly digested or totally undigested prey in their alimentary system, which indicated that these animals died a short time after feeding (Kuiken *et al.* 1994).

The reason for the increase in the number of strandings between 2001 and 2004 has not been established. It has been suggested that the increase in strandings in recent years was due to the operation of pelagic trawls for bass in the western English Channel. However, there is no clear trend in the proportion of stranded dolphins with evidence of bycatch (Figure 4.15), implying that the increase may relate more to an increase in number of common dolphins present off the south-west coasts of UK (see Section 1.2.5). In addition, the difference in the age (body length) and gender composition of common dolphins caught in the UK bass fishery and those stranded, and the seasonal pattern and location in the fishing effort since 2001, suggests that the pelagic trawl fishery not responsible for most of the strandings (Northridge *et al*. in prep). Logically, this suggests that other fisheries operating in the same area that do not have cetacean bycatch monitoring schemes may be responsible.

4.2.7.2 Strandings of common dolphins on Irish coasts

The majority of common dolphin strandings occurred along the Irish western and southern coastlines, corresponding with areas of highest sightings in recent years (Murphy 2004a). It was suggested, that the lack of stranding data along the east coast corresponds with low sighting levels in the Irish Sea (Pollock *et al.* 1997; Berrow *et al.* 2002) but may also be a consequence of water circulation and prevailing wind direction depositing dead dolphins from the Irish Sea onto the Welsh coastline (Rogan *et al.* 2001).

Peak strandings years were noted in 1992, 1998, 2001, 2002 and 2003. The peak during 1992 may have resulted from interactions between common dolphins and fisheries (Berrow and Rogan 1997), with 41% of the strandings reported during January-March, along the southern coastline (Murphy 2004a). At the same time, as described in Section 1.2.7.1, a peak in common dolphin strandings was reported along the UK coastline, and the majority of dolphins were diagnosed as bycatch. The increase in strandings during 1998 resulted from a high number of live strandings (35%). Live strandings also contributed to an increase in strandings during both 2001 and 2002, whereas the peak in strandings in 2003 can be attributed to interactions with fisheries. Although only three months (January-March) were analysed in 2003, 35% of common dolphins that stranded during this period (n = 20) were diagnosed as bycatch (Murphy 2004a).

Murphy (2004a) reported that 49% of all common dolphin strandings since 1990 occurred during the period January to March, with 25% of strandings during this period diagnosed as bycatch. The majority of strandings during the first quarter occurred along the western (52%) and southern (41.3%) coasts of Ireland. Common dolphins had a total body length range of 116 to 223 cm and were between 0.5 to 25 years in age (Murphy 2004a). Fish was the main prey item and examination of stomach contents revealed species such as hake *Merluccius merluccius*, sprat *Sprattus sprattus*, horse mackerel *Trachurus trachurus*, herring *Clupea harengus*, whiting *Merlangus merlangus*, blue whiting *Micromesistius poutassou*, mackerel *Scomber scombrus* and *Trisopterus* species (Murphy, unpublished data). Blue whiting, mackerel and horse mackerel are some of the main fish species targeted by pelagic trawls operating in the Celtic Sea and off the west coast of Ireland between January and March (Couperus 1997a; Morizur *et al.* 1999; Murphy 2004a).

4.2.7.3 Strandings of common dolphins on French coasts

Long-term information on common dolphin relative abundance, distribution, causes of death and biological traits is available in data deriving from a stranding reporting and recording scheme co-ordinated from La Rochelle since the year 1972 (e.g. van Canneyt *et al*. 2004). The structure of this stranding scheme and the reporting effort has not changed since the early 1980s. About 12,000 stranding events have been recorded to date on the coasts of France, the great majority of them from the Atlantic coast.

The most remarkable feature of the time series was a drastic increase in small delphinid stranding level from 1989 onwards, with extensive year-to-year variations (Figure 4. 16). Years of high stranding level corresponded to events of multiple strandings (or stranding peaks), during which the stranding rate was 20 to 50 times average seasonal figure. These events typically last for 2-3 weeks, are composed of 94% common dolphins, of which about 2/3 show either amputation of tail flukes, pectoral flippers or dorsal fin, broken rostrum or opening of the abdominal cavity, features typical of bycaught animals that have been returned to the sea by fishers..

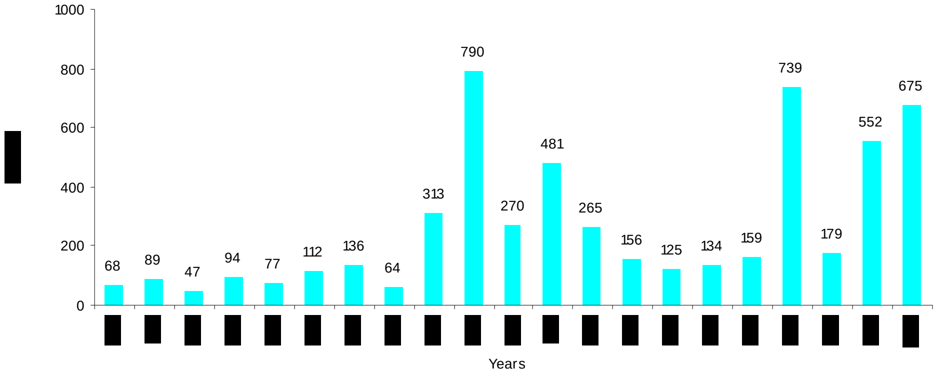

Figure 4.16. Time series of common dolphin strandings along the Atlantic coast of France.

Five peaks in strandings (1989, 1991, 1997, 1999 and 2000) with a total of 1,210 dolphins were compared with stranding levels in the winters immediately prior to and after the peak (353 dolphins) and with non-peak years 1993, 1994, 1995, 1996 and 1998 (282 dolphins). Peaks of multiple strandings had five characteristics:

1. more strandings in the southern bay of Biscay (64% *vs* 24-35%),
2. having a higher proportion of common dolphins, (96% *vs* 72-83%),
3. displaying a higher sex ratio in favour of males (2.2 *vs* 1.4-1.7),
4. showing more external evidence of incidental catches (45% *vs* 22-37%) and
5. with the smaller length classes being less represented.

These patterns suggest that the peaks are largely associated to events of increased incidental mortality, but that incidental mortality also occurs out of these peaks. The fact that males of common dolphins are more heavily exposed than females may be a result of differential habitat and food utilisation and may have important consequences as to the impact of this additional mortality on social structure.

4.2.7.4 Strandings of common dolphins on Spanish coasts

The common dolphin is the most frequently (47%) recorded cetacean species that strands on Galician coasts (López *et al*. 2002). In the period 1990-1999 there was a generally increasing trend in the annual total number of animals reported in the Galician Strandings Reporting Scheme, but the most striking increase was in common dolphin strandings. There is a late winter peak of strandings (March) that coincides broadly with the time of the year when the upwelling index is at a peak, with winds from the west predominating. The number of strandings was low in autumn, when easterly winds prevail. There were fewest strandings in summer, when upwelling occurs due to the strong westwards component of the winds. Strandings of males were generally more frequent than those of females; in all four quarters males significantly outnumbered females. The bias towards males was higher at the end of the year. The total lengths of male dolphins ranged from 100 to 240 cm, while lengths of females ranged from 100 to 220 cm. There were significant seasonal differences in the average size, smallest average stranding length was in the first quarter and the largest in the third quarter of the year.

4.2.7.5 Strandings of common dolphins on Portuguese coasts

Silva and Sequeira (2003) analysed common dolphin stranding information from the Portuguese coastline. Between 1975 and 1998, approximately 23% of stranded dolphins along the Portuguese coast exhibited signs of bycatch. Highest numbers of strandings occurred from February to April, with a peak in March; this was attributed to severe weather conditions during the winter period, differences in the distribution and/or abundance of animals, oceanographic conditions, and/or topographic conditions of the region. Furthermore, higher numbers of strandings were reported along the northern and central coasts, than along the southern coast. The pattern and abundance of the main prey species (sardines and blue whiting) of the common dolphin off the Portuguese coast matches the stranding pattern. Significantly more immature individuals were found stranded, and the sex ratio was biased towards males. The male-biased mortality was significantly more prominent between January and March.

4.2.8 Trends in sources of mortality

Some anecdotal information on the effects of deliberate killing of common dolphins is available. In the 1970s and earlier, bow-riding dolphins were deliberately harpooned for food, a practice which has now mostly disappeared. For example, in the Irish Sea and Celtic Sea, the practise of harpooning was observed during a trip in the *Nephrops* fishery in the 1970s (Y. Morizur pers. comm.). A rough extrapolation to that French fleet fishing there throughout the year suggests (using 2. dolphins per month per boat) a total of 3000 killed dolphins. In the tuna lining fishery a retired fisherman has suggested a possible 2000 dolphins killed every year by the French fleet. The Spanish tuna fleet probably had the same practises (Antoine 1990). Note that these figures are of all dolphins, not just common dolphins. Silva and Sequeira (2003) note that 6% of common dolphins stranded on Portuguese coasts between 1975 and 1998 had been hunted, but that this practice had been banned in 1981.

In the 1970s, set nets were not mechanised and were relatively short and pelagic pair trawling had not developed, thus it seems likely that any effects of fishing would have been less than following the introduction of these practices.

One way to get the trend in the fishing impact status of common dolphin could be to make a study based on interviews with retired fishermen covering several metiers and countries concerned with areas VII and VIII. Precision in the estimates could be obtained by calculation of variance in the interview data. The interview could also provide information on past variations in local abundance of dolphin populations between seasons and years.

At the present moment, in the North-east Atlantic common dolphins have quite a high age at sexual maturity, a low pregnancy rate, and a long calving interval. These results suggest that the population is possibly near its carrying capacity. If fisheries are catching too many common dolphins in the North-east Atlantic, this will be detectable by analysing their reproductive parameters. If common dolphins are being exploited, over time, an increase in the pregnancy rate, a shortening of the calving interval, and increase in the growth rate, and lowering in the age attained at sexual maturity may arise, as a result of increased resource availability from the decrease in population abundance (Murphy 2004a).

4.2.9 Gender segregation

Group segregation has been noted to occur in common dolphins in the Pacific (Ferrero and Walker 1995), and in the Eastern Tropical Pacific (Danil 2004). Off the New Zealand coast, the presence of nursery groups and male bachelor groups has been reported (Neumann *et al.* 2002). However, large mixed groups composed of juveniles, mature males, mature females and their calves were also observed (Neumann *et al.* 2002).

In the north-east Atlantic, Murphy (2004a) has shown a high mortality rate of neonates and calves in the Irish tuna fishery, and similar results have been reported in the French tuna fishery (Goujon *et al*. 1994, MICA study area see Figure 4.1). A total of 88 common dolphins incidentally caught in the Irish tuna fishery were aged, and the sample was composed of 44 individuals within the age classes 0-2 years, 8 within the age classes 3-8 years, and 33 individuals were within the age classes 10-25 years (Murphy, unpublished data). This suggests a number of points (1) the tuna drift-net fisheries were operating in the calving area/maternal feeding grounds or breeding area of common dolphins (Murphy 2004a); (2) age and sex segregation during the summer period, as a lack of sub-adults was noted in the bycaught common dolphin sample; and (3) strong age-dependant bycatch selectivity.

Additional evidence of segregated juvenile social groups was documented when a group of seven common dolphins live stranded in 2002 along the west coast of Ireland. Four individuals died, all were sexually immature, ranged in body length from 150 to 195 cm, and from one to eight and a half years in age (Murphy 2004a). During the winter period, bycatches in pelagic trawl fisheries operating off the French Atlantic coast have shown a high incidence of juvenile common dolphins. Van Canneyt *et al*. (2003) found during peak winter strandings along the French Atlantic coast, weaned immature individuals (2-6) were more exposed during these peaks, whereas older adults 10+ and nursed calves 0-2 were less exposed. Of the common dolphins that were diagnosed as bycatch, the proportion of weaned juveniles was twice as high as that of older adults. During the same period however, the UK strandings data shows a high proportion of larger-sized common dolphins stranded along the south-west coast (Figure 4.17). Where data was available, 61% of the female sample was larger than 190 cm (n = 189), and 54% of the male sample larger than 200 cm (Natural History Museum, unpublished data).

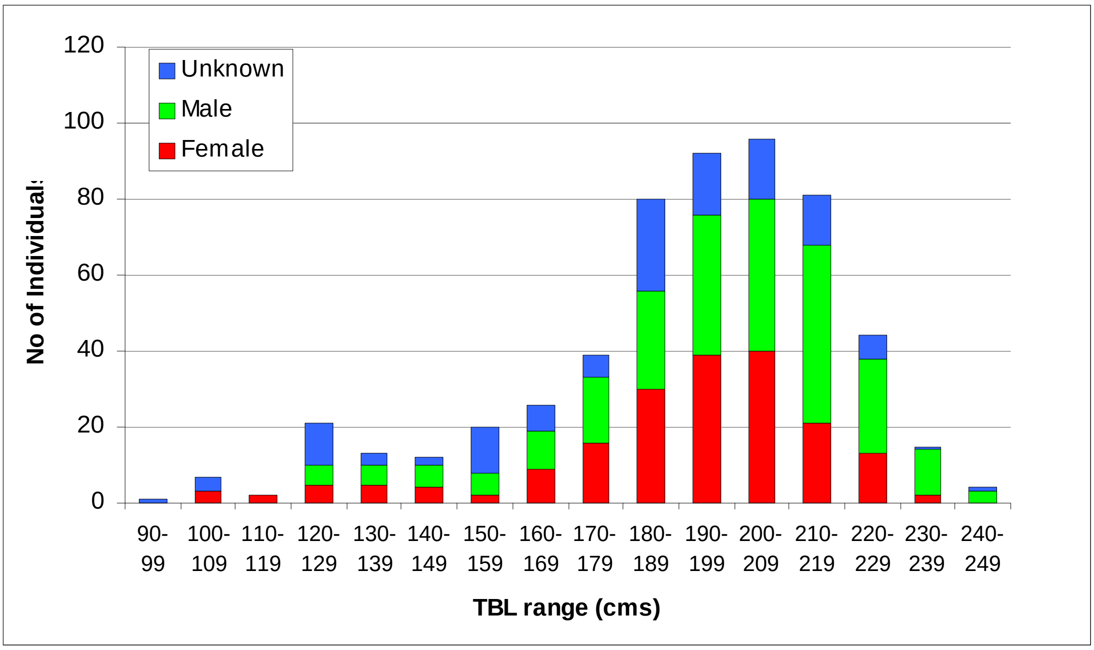

Figure 4.17. Total body length (TBL) range of common dolphins stranded in winter (November to April) on the south-west coasts of England, March 1990-March 2004.

75% of the common dolphins caught in the bass trawl fishery operating in the western English Channel were males (Northridge *et al*. in prep); where data were available, these ranged in body length from 150 to 230cm, with 69% less than 195 cm (Northridge, unpublished data).

This suggests further sexual segregation at this time, with mainly sub-adult males associating with the bass trawl fishery. Age and/or sex segregation was also investigated in multiple individual bycatch events during the spring and summer months off the Portuguese coast. Results showed that sexually mature females only associated with young calves, sexually immature males either formed separate groups or joined mature male groups, but there was a complete absence of a sexually immature female group (Silva and Sequeira 2003).

Evidence of age and sex composition can also be observed in multiple live-stranding events, with mass strandings of nursery groups (mature females with calves) reported along the Irish and French coasts (Dabin *et al.* 2003; Murphy 2004a). In February 2001, 15 common dolphins live stranded on the Mullet Peninsula, west Ireland. Eleven dolphins were refloated and five died; one male yearling, three sexually mature females aged between 14 and 17 years and one pregnant 17-year-old female (Murphy 2004a). Further evidence of nursery groups within common dolphins in the north-east Atlantic arose when a live mass-stranding event occurred along the French coastline in 2002. The group consisted of one male calf and 52 females aged between 6 to 21 years (Dabin *et al.* 2003). This implies that weaned immatures (2 to 6 years old) (van Canneyt *et al.* 2003; Dabin *et al.* 2003), sub-adult males and mature males segregate from nursery groups.

4.2.10 Health status of common dolphins

Investigations into contaminant levels in common dolphins have been undertaken in the North-east Atlantic, which included analysis of both heavy metals and organochlorines. Zhou *et al*. (2001) found an increase in mercury with body length, and variations in mercury levels were noted in the North-east Atlantic. Mercury levels were low in the Portuguese sample compared to sample from the French Atlantic coast (Holsbeek *et al*. 1998).

Pierce *et al*. (2004) undertook a preliminary analysis of persistent organic pollutants (POPs) and toxic metals in female common dolphins, in order to examine bioaccumulation of pollutants and investigate transport pathways and impacts on reproduction. Samples from Scotland, Ireland, Holland, Belgium, France and Galicia, Spain were analysed. The analysis of POP burdens in relation to different putative explanatory factors showed that in common dolphins, diet (as exemplified by the major gradient in the fatty acid variables), country (France v elsewhere) and the number of *corpora albicantia* were the significant explanatory variables. Although sample sizes were small, the median levels of S18-PCBs and CB153 in common dolphins from the Western Channel and the Bay of Biscay were above the threshold levels for effects on reproduction as reported in the literature for the bottlenose dolphin (Schwacke *et al.* 2002) and the harbour seal (Reijnders 1986; Boon *et al.* 1987). Hence, country (France) was a main explanatory variable for explaining POP burdens. Further analysis found that longitude was the second most important variable for explaining PCB burdens, i.e. there is a north-south gradient pattern (with higher PCB values in southern animals). Other important explanatory variables were diet, reproductive status, season and length (G. Pierce pers. comm.).

The number of c*orpora albicantia* was another significant explanatory variable for POPs. Further analysis needs to be undertaken to see if effects of the number of *corpora albicantia* relates to the variations in POP levels between pregnant and resting mature individuals (Pierce *et al.* 2004). The number of scars recorded on common dolphin ovaries was also correlated with mercury levels (Pierce *et al.* 2004). Other results showed that cadmium concentrations appeared to be closely linked to diet.

Danil (2004) also found that pregnancy rates decrease with increasing age. If on the whole, older individuals were analysed in the north-east Atlantic by both Murphy (2004a) and Pierce *et al.* (2004), this may explain the lower pregnancy rate in the north-east Atlantic compared with Danil’s (2004) study in the eastern tropical Pacific. Another possible explanation for the lower pregnancy rate in the north-east Atlantic compared to the eastern tropical Pacific may be due to the contaminant load of common dolphins in European waters. As mentioned before Pierce *et al.* (2004) found that the number of *corpora albicantia* scars is a significant explanatory variable for individual POP levels. This then suggests that some resting mature individuals are possibly not/rarely reproducing, and are therefore just ovulating and accumulating higher POP levels. In contrast, individuals with lower numbers of these cars are reproducing normally and are passing their contaminant load onto their offspring. However, Danil (2004) suggested that if foetal mortality occurs, as this has been documents in other dolphins from the eastern tropical Pacific (Perrin *et al*. 2003) and was not accounted for in the initial calculated 47% pregnancy rate, the adjusted calculated calving interval for common dolphins could be on average 1.7 times longer.

4.2.11 Diet of common dolphins

The most likely cause of the seasonal (and possibly inter-annual) movements observed in common dolphins is changes in diet, foraging behaviour and/or changes in the distribution of food species. A number of studies have analysed the dietary remains of common dolphins in the North-east Atlantic. On the whole, during the winter period, common dolphins appear to prey upon small pelagic fish in inshore areas (Brophy *et al.* 2005), whereas common dolphins caught in tuna driftnets in the summer beyond the continental shelf edge had been feeding predominantly on squids and meso-pelagic fishes (Hassani *et al.* 1997, Brophy 2003). Diet studies by countrty are summarised below; note that the Irish, French and Galician data used here are derived from on-going studies, are preliminary and have not yet been published.

4.2.11.1 Diet studies of common dolphins from UK waters

In UK waters, four different studies were carried out analysing stomach contents of common dolphins. Common dolphins were found to eat a wide range of fish over the continental shelf in winter. In recent years diet has included especially horse mackerel and sardines (Table 4.6).

Table 4.6. Diet of common dolphin in the English Channel: numbers are percentages of stomachs containing each fish type (Northridge et al. in prep.).

	Pascoe (1985)	Kuiken *et al*. (1994)	NHM (1995)	Gosselin (2001)	Gosselin (2001)
Sample size	2	36	76	18	n=18, 63.85kg
Prey species	**% of stomachs in which present**			**% of diet by weight**	
Clupeids	100		12		
Gadoids			4		
Garfish			1		
Gurnard			4		
Horse mackerel		6	16	76	45
Mackerel	100	59	41	30	12
Norway pout		6	1	35	<1
Pleuronectiformes			1		
Pollock			1		
Sandeel			3		
Sardine		38	32	30	43
Solenette			1		
Squid sp	50		13		
Whiting			8		
Witch flounder			1		

In Scottish waters, stomach contents of nine common dolphins that stranded between 2000 and 2003 were analysed by Pierce *et al.* (2004). Fourteen fish taxa and two cephalopod taxa were identified from the stomachs of common dolphins, and mackerel followed by whiting were the main prey consumed, together making up more than 40% of the estimated prey weight.

4.2.11.2 Diet studies of common dolphins from French waters

The food of the common dolphin has been investigated separately for inshore (continental shelf) animals sampled from stranded specimens, and offshore (oceanic) animals sampled from those bycaught in the former albacore drift-net fishery. Stomach content analysis allowed the diet to be quantified as frequency of occurrence, relative abundance, reconstituted mass and prey size distributions (L. Meynier, J. Spitz and C. Pusineri, unpublished data).

The diet of the common dolphin in the two oceanographic domains differs completely in terms of species composition. A very diverse fauna of vertically migrating mesopelagic fish and squids dominated the diet in the oceanic domain, together with some epipelagic fish (Table 4.7). A characteristic of the oceanic domain was the fairly equal proportion of fish and squid in the diet, even if the squid contribution might be overestimated due to longer retention time of squid beaks in the stomachs compared to fish diagnostic remains. Major species were the lantern fish *Notoscopelus kroeyeri* and, secondarily the squids *Teuthowenia megalops* and *Gonatus steenstrupi* and the epipelagic Atlantic saury *Scomberesox saurus*.

Table 4.7: Important prey in the stomachs of French oceanic common dolphins (sampling from bycaught animals common dolphins the importance of prey species identified from 64 non-empty stomach samples, 32588 prey items) and important prey in the stomachs of stranded common dolphins (n = 71 non-empty stomach samples and 11421 prey items from the Bay of Biscay, 48 non empty stomach samples from the western Channel). Importance is expressed as % frequency of occurrence, % number and % weight)

Prey	% Occurrence		% Number		% Mass	
Oceanic/offshore common dolphins						
Fish						
Lancet fish *Notoscopelus kroeyeri*	87.3		67.0		36.6	
Atlantic saury *Scomberesox saurus*	30.2		0.8		3.3	
Cephalopods						
Atlantic cranch squid *Teuthowenia megalops*	69.8		3.5		18.7	
Atlantic armhook squid *Gonatus steenstrupi*	63.5		0.9		12.2	
Inshore common dolphins	BB	WC	BB	WC	BB	WC
Fish						
European anchovy *Engraulis encrasicolus*	56.3	2	16.9	0.1	11.7	0
Horse mackerel *Trachurus trachurus*	70.4		15.5		18.3	
Cephalopods						
Loligo spp	16.9	38.8	0.2	5.3	2.8	28.7

On the continental shelf, dietary data are from single stranded animals collected in all seasons in the Bay of Biscay and from a mass stranding in winter 2002 in the Western Channel. A few pelagic fish largely dominate the diet of the common dolphin: sardine and, to a lesser extent, horse mackerel and anchovy (Table 4.7). *Loligo* squid formed a high percent mass in the western Channel but it is unclear if this represents a permanent difference or is due to the particular nature of the material analysed.

4.2.11.3 Diet studies of common dolphins from Irish waters

Irish studies reported similar findings to the French study. Diets of common dolphins that stranded along the Irish coastline and bycaught in the Irish tuna fishery were analysed. The sample composed of 133 non-empty stomachs, and a total of 49 prey species were recorded, which included 35 fish species, 13 cephalopods and 1 crustacean. In the offshore sample, fish were numerically the most important prey group (95% of prey by number), with cephalopods and crustaceans comprising 5.4% and 0.1% respectively. Fish representing at least five families and 16 species were identified, but myctophids dominated the fish component accounting for 13,155 (90.2%) of the fish recovered. *Diaphus sp.*, *Myctophum punctatum* and *Notoscopelus kroeyerii* were the three most important species representing 89% of fish prey (Brophy *et al.* 2004). Fish also formed the dominant portion of the stomach contents of stranded individuals (97%), with cephalopods making up 3%. However, gadidae comprised 59% of the fish component, with *Trisopterus spp.* (45%) being the most commonly occurring fish sampled.

Brophy *et al.* (2005) found that in both groups, the foraging strategy appears to involve targeting relatively small-sized schooling fish. Offshore dolphins feed nocturnally when the migrating deep-scattering layer approaches the surface. Whereas, in inshore sampling area, it appears that aggregations of small pelagic fish are preyed upon. No significant difference for these variables was found between different sex or maturity groups in the bycatch sample, however significant variations were found between sexually mature male and females common dolphins that stranded along the Irish coastline (J. Brophy pers. comm.). The majority of strandings take place along the Irish coastline during the winter period (Murphy 2004a), and variations in dietary data between mature males and females suggest different feeding habitats and different preferred prey during this time. As mentioned before, age segregation occurs during the wintertime, with formation of nursery groups of mature females and calves, and also possibly groups of bachelor males; and the dietary data also suggests this.

Brophy *et al.* (2004) also recorded eight individuals with milk in their stomachs (aged 0 – 3 months), while three (aged 3 – 6 months) had both milk and solid food suggesting that weaning occurs between 3 and 6 months.

4.2.11.4 Diet studies of common dolphins from Spanish (Galician) waters

Between 1991 and 2003, 414 non-empty stomachs of stranded common dolphins were analysed (Santos *et al*. 2004). Although all common dolphins were stranded animals, one-third showed evidence of entanglement in fishing gear, while for a number of other individuals, cause of death could not be determined and the possibility of bycatch could not be ruled out (Santos *et al*. 2004). Similar to other studies on stranding patterns, the majority of common dolphins were found stranded during the winter period, with a large proportion of individuals sexed as male (López *et al*. 2002).

Santos *et al*. (2004) documented 25 fish taxa and 15 cephalopod taxa in dietary remains from stomachs. The majority of the prey were pelagic species (e.g. blue whiting, sardine and horse mackerel), although commercial and inshore species (e.g. sandeels, scaldfish, sole, gobies, garfish, *Atherina* sp. etc.) were also present. Blue whiting and sardine were the most important prey categories by reconstructed weight, together making up more than 56% of the weight. The main cephalopod species eaten by the common dolphin were *Loligo* sp. (*L. vulgaris* and *L. forbesi*), which comprised the main prey by reconstructed weight.

It was concluded that common dolphins show signs of being opportunistic feeders, for example, more sardines were consumed in years of higher sardine abundance and lower recruitment of blue whiting. However, their diet was restricted as 56% (%weight) of the diet was composed of sardine and blue whiting. Santos *et al*. (2004) also reported clear indications for direct competition between common dolphins and Galician fisheries for these fish.

4.2.11.5 Diet studies of common dolphins from Portuguese waters

Silva (1999) analysed 50 common dolphin stomachs from animals stranded on Portuguese coasts and incidentally caught in fishing gear off the Portugal between 1987 and 1997. Even though 27 different species of fish and eight cephalopod species were identified in their diet, the majority of the prey was composed of four main pelagic species. Six different fish species (sardine, blue whiting, *Atherina* sp., *Trachurus* and scombrid species), composed 84% of the total estimated weight, but overall sardines were the most important prey item, occurring in 81% of the stomachs, and represented 27% of the total number of prey taken and 43% of the estimated weight (Silva 1999).

Common dolphins stranded along the Portuguese coasts appear to have a higher proportion of sardines in their diet than animals stranded along the Galician coastline (Santos *et al*. in prep). However, it appears that the estimated number of sardines from spring acoustic surveys carried out by both countries since 1986, showed that this species is more common in Portuguese waters than off Galicia (Carrera and Porteiro 2003; Santos *et al.* in prep).

4.2.11.6 Fatty acid analysis

During the BIOCET project, blubber samples from common dolphins (Scotland (mainly west coast), Ireland (south and west coast), France and Galicia, NW Spain) were analysed (Pierce *et al*. 2004). The inner blubber layer was analysed as higher levels of fatty acids derived primarily from the diet are found in this layer (Pierce *et al*. 2004).

Principal component analysis of the sample suggested segregation of Ireland and Scotland away form France and Spain. Overall, no significant variations in fatty acids were found between the Scottish and Irish samples (Pierce *et al*. 2004). Due to variations in diet of mature male and female common dolphins, and as mature female common dolphins were the largest group analysed for their fatty acid profiles, this group was further investigated using canonical discriminant analysis (CDA). CDA revealed a significant geographical variation in fatty acid profiles in mature female common dolphins in the North-east Atlantic ($p = 0.000$). However, the CDA plot shows an overlap in fatty acid profiles between Ireland and Scotland, whereas France and Spain appear to be separated from all other areas (Figure 4.18). Canonical discriminate functions 1 and 2 accounted for 53.9% and 37.0% of the variation. Overall, fourteen out of fifteen fatty acids analysed varied between areas (Ireland, Scotland, France and Spain).

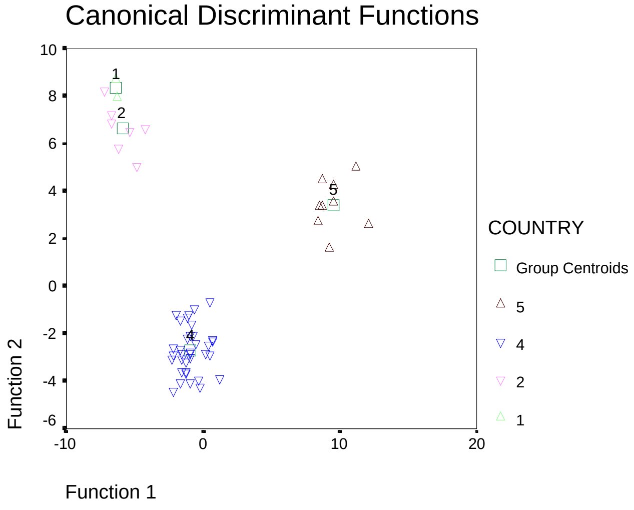

Figure 4.18. CDA plot of mature female common dolphin fatty acid profiles for each country. Country codes 1 = Scotland, 2 = Ireland, 4 = France, 5 = Spain.

4.2.11.7 Diet summary

Common dolphins appear to vary their diet depending on the relative abundance of their prey and the region they inhabit in the north-east Atlantic. However, even though they appear generalists and opportunistic feeders, the majority of their diet is selective for only a few main species. The precise species varies from region to region in the north-east Atlantic.

4.3 Bycatch of common dolphin in fisheries

The following section summarises recent information available to the working group on common dolphin bycatch in extant fisheries. The Working Group were disappointed to note that some countries with fisheries liable to catch common dolphins had not submitted data to ICES/the Working Group, despite a specific request from the European Commission to do so (letter from John Farnell, DG for Fisheries and Maritime Affairs dated 20 April 2005). The Working Group was concerned that lack of information might be interpreted as lack of bycatch, and that measures to reduce bycatch might be imposed only on those countries/fisheries submitting data. This would be entirely inappropriate. The Working Group considered that measures should conversely be imposed on those countries/fisheries NOT submitting data, these measures to be removed once data were forthcoming demonstrating low bycatch in the fishery concerned. This would be consistent with the precautionary principle and would also act in support of scientists and administrators attempting to gain this information.

4.3.1 Information on bycatch of common dolphins in UK fisheries

4.3.1.1 UK pelagic pair trawl for bass

The Natural Environment Research Council's Sea Mammal Research Unit (SMRU) has been funded for some years to monitor bycatch in selected fisheries around the UK. The pair trawl fishery for bass in the western Channel has been monitored since 2000, with trials also being undertaken in an effort to reduce (and preferably minimise) the bycatch rate in the fishery. Bycatch in this fishery was reported in Northridge *et al.* (2003) and is updated in Table 4.8.

Table 4.8. Summary of observations of dolphin bycatch in the UK offshore bass pair trawl fishery during 2000-2005. See text and Table 4.9 for explanation of 2003-2004 season. Figures for 2004-2005 are very preliminary, and have not been stratified.

Season	Observed hauls	Dolphin mortalities	Bycatch rate (dolphins/observed haul)	Unobserved hauls*	Estimated total mortality
2000-2001	91	52	0.57	241	189
2001-2002	91	9	0.10	295	39
2002-2003	113	26	0.23	382	114
2003-2004	133	169	1.27	263	429**
2004-2005	149	99	0.66	71	145
Totals	577	331	0.57	1252	916

* These data require checking and may not be accurate; this in turn may affect estimated total mortality

** An estimate of 439 was presented in SMRU (2004) based on SMRU's observations with the addition of 20 hauls and 17 dolphin mortalities that were reported by fishermen. The figures given here exclude any such additional data provided by fishermen to ensure consistency between years.

Preliminary estimates of bycatch in the UK offshore bass pair trawl fishery for the winter fishing seasons are shown in Table 4.8. The rate of bycatch varies between years, and varies within year (seasonally) as well. Table 4.8 shows that 2003-2004 was an anomalous year in terms of bycatch rate – rates in this year were higher than the three previous years combined. This year also showed an unusual seasonal pattern (Figure 4.19), with many more bycatches than usual early in the season. In years 2000-2001, 2001-2002 and 2002-2003 bycatch rates broadly reflected fishery effort, with most effort and highest bycatch rate both being in March. On the assumption that total fishing effort (number of hauls) is known accurately, it is reasonable to estimate total bycatch from observed rates of bycatch in these years. In the 2003-2004 season this pattern was not maintained, with highest rates early in the season, when fishing effort is low. If the mean annual rate is multiplied by all hauls (see caveat above), the resulting estimate of total bycatch will be biased upwards. This bias can be addressed by dividing the season into two – a higher bycatch period (all season to the end of February) and a lower bycatch period (March and April), and estimating bycatches before and after this time period. The less biased total bycatch for 2003-2004 is 429 animals (Table 4.9), which compares to an uncorrected figure of 503 mortalities.

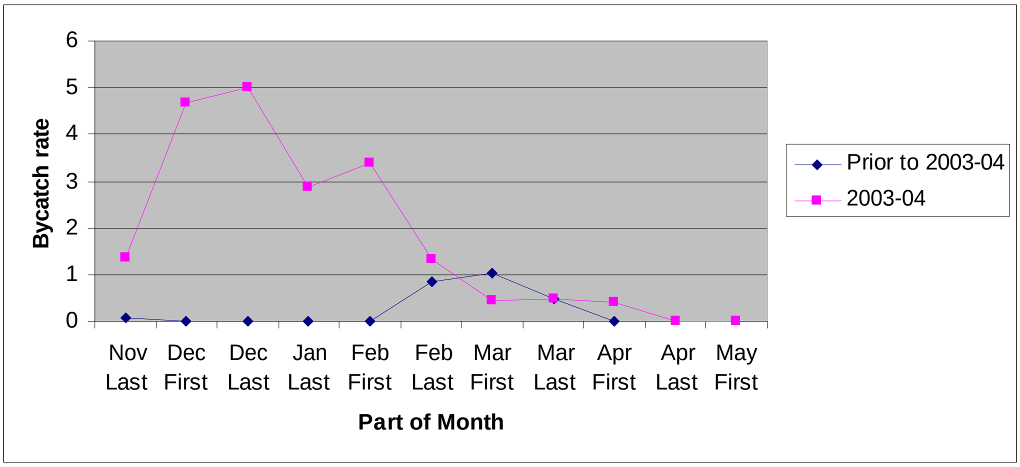

Figure 4.19. Bycatch rates by half month, November to May in two periods. Note that no observations have been made in early January.

Table 4.9. Bycatch rates in two parts of the 2003-2004 season to give less biased estimate of total bycatch.

Months	Observed hauls	Dolphin mortalities	Bycatch rate (dolphins/observed haul)	Unobserved hauls	Estimated total mortality
Mid Nov-Feb	64	144	2.25	87	340
Mar-April	69	25	0.36	176	89
Season total	133	169		263	429

The reason for the change in pattern between years is not known. During the latest fishing season (2004-2005) common dolphin bycatch rates were considerably lower than in the 2003-2004 season, but still higher than in three earlier seasons. A preliminary and as yet unverified analysis suggests that there was a total of around 220 fishing operations in this fishery over the 2004-5 winter season, of which 149 were observed (67% coverage), with a total of 99 observed common dolphin mortalities. The observed bycatch rate per operation (0.66) is about half that of the previous season's rate. A simple and unstratified ratio estimate suggests that the total mortality in this season's fishery would therefore have been around 150 common dolphins. Neither the fishing area nor the techniques have changed between 2001-2002 and 2003-2004, so it seems likely that the observed increase in bycatch rate in 2003-2004 was due to changing patterns of seasonal movements among the dolphins. The fishing area will have changed slightly in 2004-2005 due to the closure within UK territorial limits to UK vessels.

The mean annual mortality was 183 animals over the five years 2000-2005. There was a bias in the bycatch in this fishery towards sub-adult males.

4.3.1.2 UK offshore gillnet for hake/pollock

The only fishery other than the bass fishery in which common dolphin bycatch has been recorded and measured in this area is the offshore hake/pollock gillnet fishery, a fishery prosecuted by both Irish and English vessels usually of 15m or more in length. Cetacean bycatch rates were measured in this fishery in 1992-1994. Dolphin bycatch was estimated to be around 200 animals per year in this fishery (Tregenza *et al.* 1997, Tregenza and Collet 1998). Subsequent management action and a decline in the overall effort in this fishery has probably led to lesser bycatch levels than these at present.

In 1999-2000 a second study was undertaken at the request of the Cornish Fish Producers' Organisation and the National Federation of Fishermen's Organisations to test possible mitigation measures to minimise porpoise bycatch. The study also recorded bycatches of common dolphins and at a slightly higher rate than in the 1994 (Table 4.10). All of the dolphins were taken between October and March. The hake gillnet fishery that was monitored in these two studies is distributed well offshore, with boats making trips of several days. Fishing effort has also declined in this gillnet sector since 1995.

Table 4.10. Quantified common dolphin bycatch observation in static net fisheries in Cornwall

Fishery	Years	No of hauls observed	Km.hours	No of common dolphins
Hake gillnets	1992-1994	949	52050	3
Tangle nets	1992-1994	7	1050	0
Hake gillnets	1999-2000	237	18000	2*
Hake gillnets with pingers	1999-2000	181	12400	1

* 1 common dolphin and 1 unidentified species of dolphin

A review of existing accounts from before the 1990s reveals that common dolphins were recorded bycaught in several different fisheries in the area (not all of them with fatal consequences). A summary of this information is shown in Table 4.11. Gillnet, tangle net and demersal trawl fisheries are included, and this is not an exhaustive list.

Table 4.11. List of common dolphin bycatch incidents in southwest England drawn from past literature. Note that some 'dolphin' incidents may involve species other than common dolphins. Direct observations are of Type 'O', anecdotal records from fishermen 'A' and approximate rates given by fishermen are of type 'E'.

SPECIES	NUMBER	DATE	PLACE	FISHERY	REFERENCE	TYPE
Common dolphin	1	1923	St Ives	Mackerel drift net	Harmer 1925	O
Common dolphin	1	1925	West of Milford Haven	Trawler	Harmer 1927	O
Common dolphin	1	1926	Off Start Point	'Fishing smack'	Harmer 1927	O
'Dolphin'	2	1979	Off Sussex	Midwater trawl	Northridge 1988	O
Common dolphin	5*	1982	Plymouth	Purse seine	Pascoe 1986	O
Common dolphin	1	~1983	Channel	Trawl	Northridge 1988	A
'Dolphin'	1	1980s	Devon	Set gill net	Northridge 1988	A
'Dolphins'	4	1986	Mevagissey	Wreck nets	Northridge 1988	A
Common dolphin	6	1986	Off Cornwall	Midwater trawl	Northridge 1988	O
Dolphin	1	1987	Brighton	Gillnet	Northridge 1988	A
Dolphins	2-3 pa	1980s	Mevagissey	Gillnets	Northridge 1988	E
Dolphins	3/yr/boat	1980s	Mevagissey	Wreck nets	Northridge 1988	E
Dolphins	2-3/yr/boat	1980s	Mevagissey	Wreck nets	Northridge 1988	E
Dolphins	'occasional'	1980s	Wales	Tangle nets	Northridge 1988	E

The UK's Sea Mammal Research Unit has also made observations on board some of the other UK pelagic trawl fisheries in the area, and no cetacean bycatch has been recorded in these. The number of operations monitored in each is shown in Table 4.12. Such observations, if they reveal no bycatches, do not imply that none exists, but the more sampling that is done the more sure one can be of the true underlying bycatch rate, or how close to zero it really is.

Table 4.12. Observations in other pelagic trawl fisheries in the Channel since 2000.

Fishery	No of hauls observed	Expected annual no of hauls
Anchovy	4	8
Herring	2	157
Horse mackerel	0	104
Mackerel	9	225
Pilchard	7	126
Sprat	1	405

For all the other fisheries in the area there has been no cetacean bycatch monitoring and we therefore do not know the extent to which bycatches might occur in any of the other fisheries. However, it is clear that some other fisheries must have cetacean bycatches, because neither the approximately 180 dolphins per year in the UK bass trawl fishery, nor the similar number of dolphins per year in the hake/pollock fishery are likely to explain the numbers of dolphins washed ashore.

In 2004 four animals were retrieved from the sea in an area where bass trawl fisheries were operating. These animals were necropsied. This animal had a series of lacerations and net and rope marks about the head. A cast was made of two parallel rope marks which proved to be consistent with a pair of 6-8mm polypropylene ropes. Net mesh marks were measured and were consistent with a 270mm stretched mesh size. The incisions on the animal's beak were tested with a series of twines, and a 0.6mm monofilament twine exactly fitted the incisions that were about 2 or 3mm deep. These net characteristics describe a 10.5 inch monofilament net with a twin polypropylene headline – a type of tangle net that is typically used to catch rays and monkfish. None of the twine marks were consistent with pelagic trawl twines, which are much thicker (S. Northridge pers. comm.). De Boer *et al.* (2005, Annex vi) reported that two of the remaining animals exhibited very similar markings, none consistent with bycatch in pelagic trawl. The results from the fourth dolphin were not reported.

This supports the idea that a range of fisheries are involved with common dolphin bycatch in the western Channel/Celtic Sea. Furthermore, two common dolphins caught by a bass trawler in December 2003 had monofilament netting in their stomachs, suggesting that they may have been feeding out of gill or tangle nets.

4.3.2 Information on bycatch of common dolphins in Irish fisheries

So far, the only Irish fishery that has been monitored in detail for cetacean bycatch is the Irish tuna fishery. In 1994 and 1995, monitoring programmes were undertaken in the Irish herring fishery operating in the Celtic Sea, however no cetaceans were recorded as bycatch in this fishery at that time (Berrow *et al*. 1998). Currently, work is being undertaken on monitoring cetacean bycatch in the Irish mackerel and blue whiting fisheries, but so far this year, no cetaceans have been reported as bycatch in these fisheries (BIM, pers. comm.).

Initial investigations into the Irish fishery for albacore tuna were carried out in 1996 and 1998 and it was estimated that 345 and 2,552 common dolphins, respectively, were incidentally by the whole fishery (Harwood *et al*. 1999). During 1998 and 1999, An Bord Iascaigh Mhara (BIM) and the Marine Institute undertook a major two-year study into developing alternative tuna fishing techniques (BIM 2004). In 1999, tests on experimental trawls were carried out off western Ireland and the southern Bay of Biscay, and 313 hauls over 160 days were observed. Results showed that a total of 145 animals, which include four species of cetacean, were incidentally caught (Table 4.13; BIM 2005). Ninety percent of hauls had no cetacean bycatch, but 125 common dolphins were caught, however this occurred in just four pair trawls (BIM 2000). This highly clustered pattern of bycatch is not unusual in pelagic trawls, and may be as a result of the cohesive nature of dolphin social groups (BIM 2000). In 1999, a number of other species were incidentally caught including striped dolphins, Atlantic white-sided dolphins and long-finned pilot whales. In more recent years only common dolphins have been reported as bycatch.

Table 4.13. Cetacean bycatch from the Irish pair pelagic trawl fishery for albacore tuna. Unpublished data obtained from BIM.

	Year				
	1998	**1999**	**2002**	**2003**	**2004**
No. of observed hauls	144	330	113	55	35
No. of cetacean bycatch incidents					
Common dolphins	12	23	5	1	1
Striped dolphin		4			
Atlantic white-sided dolphin		1			
Long-finned pilot whale		4			
Total	12	32	5	1	1
Mean no. of incidents per haul	0.08	0.10	0.04	0.02	0.03
Sum of cetacean bycatch	**Number of bycaught animals**				
Common dolphins	44	125	16	1	2
Striped dolphin		10			
Atlantic white-sided dolphin		2			
Long-finned pilot whale		8			
Total	44	145	16	1	2
Mean no. cetaceans per haul	0.31	0.44	0.14	0.02	0.06

A noticeable decrease in incidental capture of cetacean in the Irish tuna fishery was been recorded between 2002 and 2004 (Table 4.13). BIM (2004) suggested that the decrease in cetacean bycatch may have resulted from a number of improvements in avoidance techniques by the Irish fleet, many of whom have been involved in the pair pelagic fishery since it commenced in 1998, which are (1) avoiding fishing operations when cetaceans are active in the area (2) carrying out a number of practices such as extinguishing stern lights while towing and (3) dropping the headline to several metres below the surface. These practices are simple to adopt and do not adversely affect fishing for albacore (BIM 2004). The use of deterrent devices in the pelagic trawls during the fishing seasons in 2002 and 2003 may further have reduced cetacean bycatch. The first of these practices seems likely to be the most effective and has been widely adopted following initial results that suggested that if cetaceans are caught in an area, there will not be large catches of albacore, and hence vessels should avoid areas of high cetacean activity (BIM 2000). It should also be noted that, since the start of the use of pelagic trawls for albacore tuna, the Irish fishery has changed fishing area. The Irish driftnet fishery only fished for tuna far off the southwest coast of Ireland (see Figure 4.12), whereas the pelagic trawl fishery is in the inner Bay of Biscay along the 1000m contour line (Figure 4.20). The main reason for the change in fishing location is attributed to the larger concentrations of tuna that are found close to the continental shelf, making it easier to fish using pelagic trawls in this location.

As can been seen from Figures 20 and 21, the majority of cetacean bycatch in the Irish tuna fishery in 1998-2003 occurred off the southwest coast of Ireland. Initial investigations into observations of incidental bycatch in the fishery found that cetacean bycatch was strongly correlated to depth of water, in that catches of cetaceans were low when fishing depth exceeded 500 metres (BIM 2000). However, recent analysis has found the relationship between cetacean bycatch and wind strength to be significant, and the relationship between water depth and dolphin bycatch is less clear (BIM 2005b). By looking at combined data from

1998, 1999, 2002 and 2003, results showed a significant difference in the likelihood of cetacean bycatch occurring in water deeper than 1000 m than water shallower than 1000m (BIM 2005b). Waters off the southwest coast of Ireland are highly productive as a number of frontal systems occur here. Albacore tuna migrate along these fronts from the Bay of Biscay to the Porcupine Bight region in spring and they return in autumn (BIM 2000), but the majority of common dolphin bycatch occurs in the Porcupine Bank (Figure 4.21). This suggests that this area has the highest concentrations of common dolphins in the area/period of the fishery. This suggestion is though very tentative as the majority of the data in Figure 4.21 was obtained in only two years (1998 and 1999) and low overall bycatch levels were observed in 2002 and 2003.

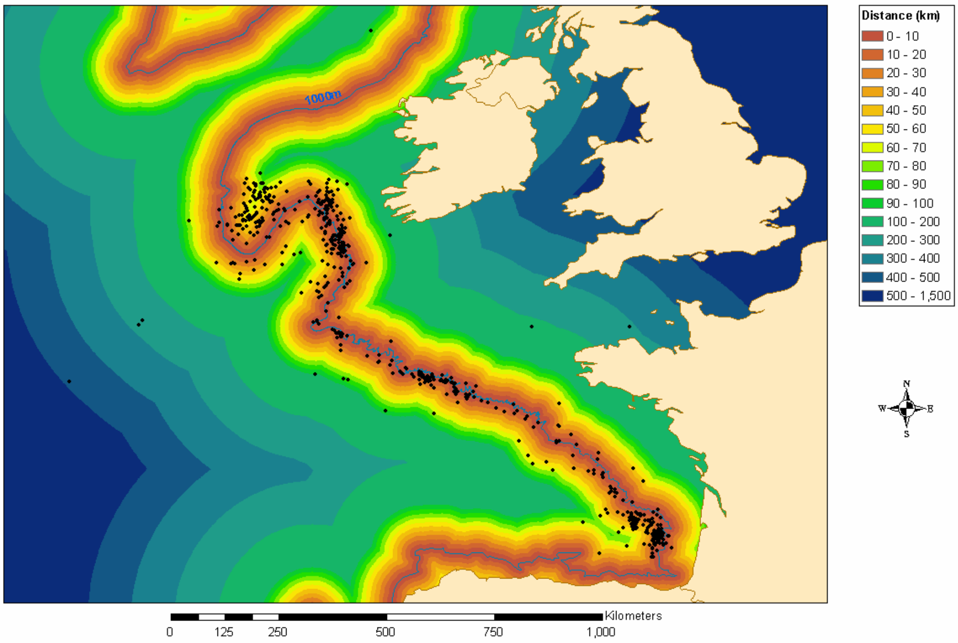

Figure 4.20. The observed position of mid water paired trawl albacore fishing positions in late summer (July to October) 1998, 1999, 2002 and 2003 (BIM 2005b).

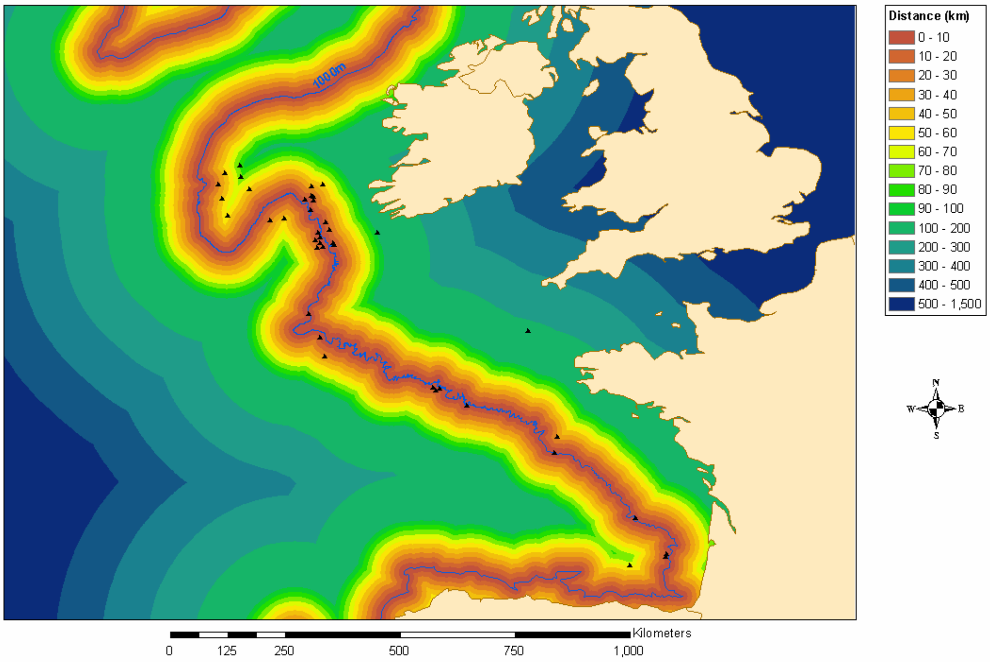

Figure 4.21. The observed positions of dolphin bycatch in relation to the 1000m depth contour in late summer (July to October) 1998, 1999, 2002 and 2003 (BIM 2005b).

4.3.3 Information on bycatch of common dolphins in French fisheries

4.3.3.1 French fixed net fisheries

Fishing effort

Nets fisheries occur inshore in areas VII and VIII, with a fleet of boats mostly less than 12 metres in length on day trips.

In English Channel and North Sea, 172 vessels have set nets as their main gears. Seventy-five per cent of the boats are less than 13 metres long. Most of the vessels in Area VII use small mesh nets less than 2 metres in height for sole. The immersion time is usually less than 2 days and net length per boat is around 5 to 10 km. Other vessels in area VIIe use large mesh nets for demersal or benthic species such as monkfish, rays and turbot. The immersion time is usually 3 days. These boats may have up to 35-40 km of nets in the sea.

In Bay of Biscay (area VIII) there are around 250 boats using set-nets as their main activity. Seventy-five per cent of the fleet have their length less than 16 meters. The main target species is sole. These nets are set for 12 or 24 hours and the length of nets at sea varies with the size of the vessels.

Bycatch

No bycatch of common dolphin has been reported in French set net fisheries. Observations have been made in several fisheries in French waters. The effort observed in looking for bycatch is summarised in Table 4.14.

Table 4.14. Summary of effort observed in French set net fisheries. No bycaught cetaceans were observed during this effort.

ICES AREA	YEARS	TARGET SPECIES / NETTING TYPE	EFFORT OBSERVED	REFERENCE
VII e, h	1992-93	Monkfish, turbot, brill, crayfish	519 km net	Morizur *et al*. 1996a, b; J. Sacchi in EC project AIR 2 N° CT 93-1122
VII d	1995-96	Sole	44 km net	Brabant *et al*. 1994
VII d	1997	Sole	22 km net	Minet 1997
VIII	1995	Hake	27 km	Brabant *et al*. 1994
VII	2003-04	Sole, turbot, monkfish	33 trips	Ifremer (pers. comm.)
VIII	2003-04	Sole, hake, spider crab	27 trips	Ifremer (pers. comm.)

4.3.3.2 French pelagic trawl fisheries

Effort

Fleet description

France has 3 large industrial boats (50-80 metres long) working on pelagic species. Several other vessels (length 16-24 metres; engine power 400 kW) use pelagic trawls. A hundred vessels work individually and combine pelagic trawling with bottom trawling during their annual activity. Seventy boats work nearly full time with pelagic trawls towed by pairs of boats.

Pair trawling activities

The pair pelagic trawling fleet targets several fish species during the year. Not all the pairs are involved every year in each seasonal fishery. The fishing effort involved in a fishery may vary between years due to the availability and/or the size of fish (e.g. anchovy) and the market value of the fish at the time. There is also a regulation for the access to anchovy (since 1992, in agreement with Spain, there is a seasonal fishing ban between 20 March and 1 June). The target fish could be anchovy, sardine, mackerel, horse mackerel, tuna, bass, black bream or hake. The areas of activity are the ICES Areas VII and VIII. Fishing effort can be expressed in several units (pairs, boats, fishing hours). Special care needs to be taken to avoid any mistake with boat and pair units.

Effort was calculated for some fisheries by using the Ifremer database on activity on a monthly basis. One boat may have several metiers in one month. The number of boat months by fishery was computed by allocating a fraction of month to each metier according to the number of metiers used in a month. The minimum (the number of boat months) and the maximum were also determined. The database was started in 2000 but data for 2004 is not yet available. Effort cannot be quantified with sufficient precision for short term fisheries. An example of the output from this database is provided below for the bass fishery. Knowledge of effort is required specifically for extrapolation of the results of bycatch observation on samples of the fleet.

The French log book database can provide more detailed information on fishing effort. However the logbook database may not contain information on all trips. Moreover the quality of the data available needs to be examined specifically for pair trawling.

Bass in Area VII

The fishing effort for bass in area VII was examined in a monthly basis (starting in January 2000, Figure 4. 22). Fishing in 2001-2002 seems to have started earlier than for the previous years (mainly 2000-2001). The maximum in one month for the medium estimate does not exceed 22 boats during the period of the study.

As the fishing season overlap between two consecutive years, a new expression of the season fishing effort was done by dividing the year in August. The results indicate that most of the seasonal fishing effort is later in the season (Figure 4.23).

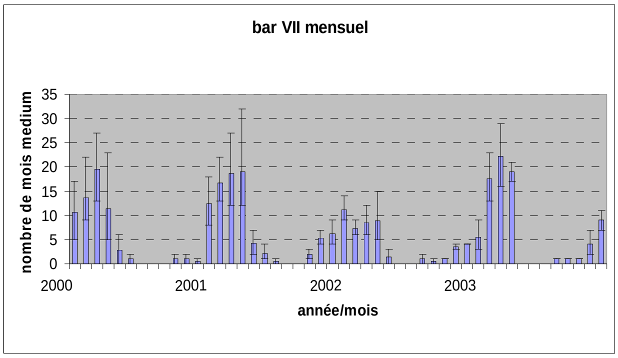

Figure 4.22. Fishing effort (boat months) by month in the French bass pair-trawl fishery in Area VII for years 2000-2003.

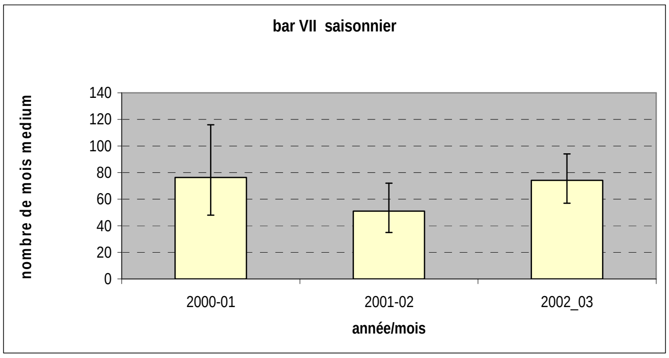

Figure 4.23. Fishing effort (boat months) by season in the French bass pair-trawl fishery in Area VII for years 2000-2003.

Bycatch observations

In 1995, observations at sea were conducted in several French seasonal pelagic trawl fisheries in areas VII and VIII (Morizur *et al*. 1999). Bycatch of common dolphin was observed in the hake, tuna and bass fisheries. However the bycatch rate was difficult to assess because of there were insufficient observations. No cetacean bycatch was observed in fisheries for anchovy, sardine, horse mackerel and seabream.

In July 2004 the European Project PETRACET started a one year project to collect data in several fisheries in areas VII and VIII. French fleets included in PETRACET are tuna, bass in area VII, bass in area VIII and the spring and autumn anchovy fisheries. Half of the time dedicated to observation at sea is dedicated to the French fisheries. An objective is to focus on the month of maximum effort in each fishery. The collaboration of the fishing industry has resulted in a many pairs of trawlers contributing to the observation programme. The observations for the tuna fishery and the autumn fishery were completed at the end of 2004, but the rest of the project is not yet finished and therefore the results presented here are intermediate and provisional. Definitive results should be available in autumn 2005. Common dolphin is the main species recorded in incidental bycatch in the fisheries observed so far (Tables 4.15, 4.16 and 4.17).

Tuna fishery

A total of 98 hauls were observed from August to October 2004; only three of them had common dolphin bycatch. A similar bycatch rate was found by Irish observers in the Irish fleet in this season.

Autumn anchovy fishery

A total of 221 tows were observed from July to November 2004 with no cetacean bycatch.

Bass, Area VIII fishery

Ninety hauls were observed between January and March 2004. Eighty of these had no bycatch of cetaceans. Most of the bycatch was of common dolphin and was concentrated on one pair that worked in a very localised area where other pairs were not observed operating at the time.

Bass, Area VII fishery

So far 59 tows on bass have been observed from 3 trips in area VII in February and March 2005. Only two hauls contained cetacean bycatch. The available data are probably less than 5% of the French fishing effort in that fishery.

Table 4.15. Provisional and incomplete results from French observation effort in the Petracet project, by fishery.

Fishery	Tuna	Autumn anchovy [1]	Bass VII [1]	Bass VIII [1]	Spring anchovy [1]	Horse mackerel [1]
Number of pairs observed	5	13	4	8	4	4
Number of trips	6	46	3(+2) [2]	12(+2) [2]	8	5
Number of days at sea	91	109	21(+11) [2]	50(+11) [2]	24	13
Number of hauls observed	98	221	59	90	43	20
Number of hauls without bycatch	95	221	57	80	43	20
Common dolphin bycatch	6	0	2	68 [3]	0	0
Risso's dolphin bycatch Striped dolphin				1 3		

1. including combined fish targets
2. including combined areas
3. forty-four cetaceans were caught on one trip with 5 incidental hauls in its fifteen observed hauls.

Table 4.16. Number of hauls with incidental bycatch and (total observed hauls) by month and in each seasonal fishery based on provisional and incomplete results from French observation effort in the Petracet project (2004-05).

Fishery	Jan	Feb	Mar	Apr	May	June	July	Aug	Sept	Oct	Nov	Dec
Tuna								3 (39)	0 (55)	0 (4)		
Bass VII		1 (12)	1 (47)									
Bass VIII	0 (17)	8 (53)	2 (20)									
Anchovy	0 (42)	0 (1)					0 (33)	0(126)	0 (26)	0 (18)	0 (18)	
Horse mackerel			0 (13)					0 (7)				

Table 4.17. Provisional and incomplete results from French observation effort in the Petracet project, frequency of hauls as a function of incidental bycatch number by fishery

Incidental bycatch number	0	1	2	3	4	5	6	7	8	9	10	10+
Tuna	95		3									
Bass VII	57	2										
Bass VIII	80		2	3	1	1	1	1				1
Anchovy	264											
Horse mackerel	20											

Early indications from the Petracet project indicate that sixty percent of the common dolphins that were bycaught were females, in contrast with previous observations in the bass area VII fishery (Northridge 2003). The length distribution showed a mode at size class 170-190 cm, suggesting that most animals caught in the fishery were immature and young adult. Much of the bycatch also occurs at night (Figure 4.24), which may help in deriving mitigation strategies.

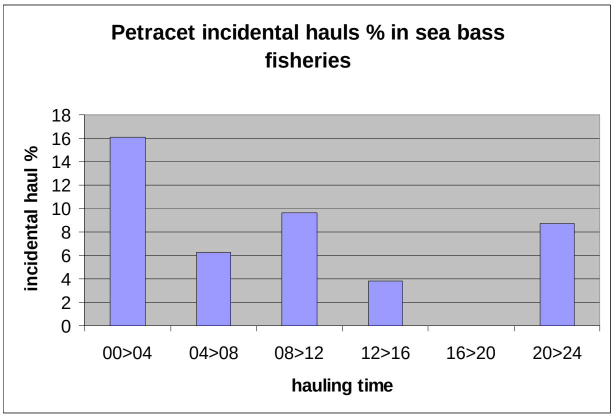

Figure 4.24. Percentage of hauls having bycatch incidents in relation with hauling time (hours) from Petracet observations (preliminary and provisional data).

At this point in the Petracet project, no check has been made for the quality of the deployed observation effort in relation to the activity of the fleet in time and space. This check requires VMS data to examine the representatives of samples and to find the best method for extrapolation. The French observations so far indicate that the greater part of the total bycatch is due to solely one pair fishing in a specific area at a specific time.

4.3.4 Information on bycatch of common dolphins in Spanish fisheries

All the following data on bycatch has been collected by AZTI (Fisheries and Food Technological Institute) observers on board commercial fishing vessels based in the Basque Country (Spain). The main objective of the observers was to study commercial aspects of the fisheries (e.g. to estimate discards and retentions, to obtain biological material, to study the catching pattern of a particular gear, for tagging). However, information on cetacean bycatches was collected on all fishing trips in which observers where on board.

Information concerning cetacean bycatch for the different fishing gears (fixed gears, purse seine and trawling in general) made in vessels based in the Basque Country harbours was reported in ICES (2003a, b).

As there is only one fishing gear in which cetacean bycatch has been observed (bottom pair trawl using VHVO nets), this review will be focused on it. Moreover, the available information in AZTI for trawlers is more complete and includes landings of the different species, number of fishing operations and spatial distribution of the operations.

There are three main fisheries in which the Basque trawlers are involved in European waters, each one of them can be considered as a metier:

- The ‘Baka’ bottom trawl fishery in ICES Subareas VI, VII and VIII targeting mixed species.
- The bottom pair trawl operating with very high vertical opening (VHVO) nets in ICES Divisions VII h, j and VIII a, b, d targeting hake.
- The bottom pair trawl operating with VHVO nets in ICES Division VIIIc targeting blue whiting.

4.3.4.1 ‘Baka’ bottom trawl

The ‘Baka’ bottom trawl metier is used by 25 vessels (2002) with fishing effort occurring in ICES Subreas VI, VII and VIII. The number of vessels involved in this fishery decreases from 1994 to 1999 (from 35 to 19vessels) and increases to reach 25 vessels in 2002. The fishery in which these vessels are involved is a mixed species fishery with a wide range of target species.

As there are no great differences between seasons, the annual spatial distribution of effort for this fleet in 2000 and 2001 is presented in the Figure 4.25. The observation effort for 2000 in number of hauls per ICES statistical rectangle is shown for only 2000, as an example, in Figure 4. 26.

No cetacean bycatch has been recorded by AZTI observers on board of vessels of this fleet.

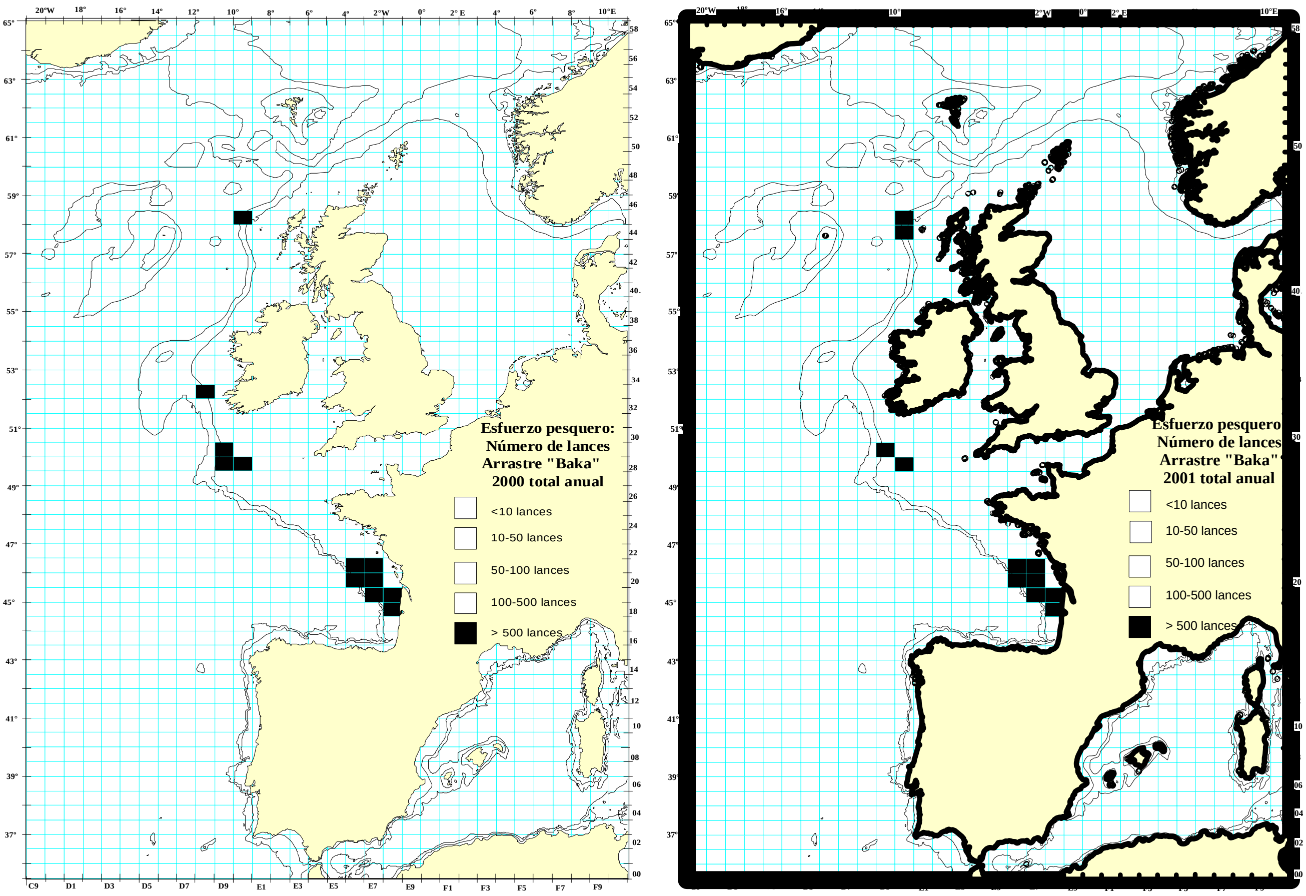

Figure 4.25. Annual distribution of effort of the "Baka" bottom trawl in 2000 and 2001 (number of hauls per ICES statistical rectangle).

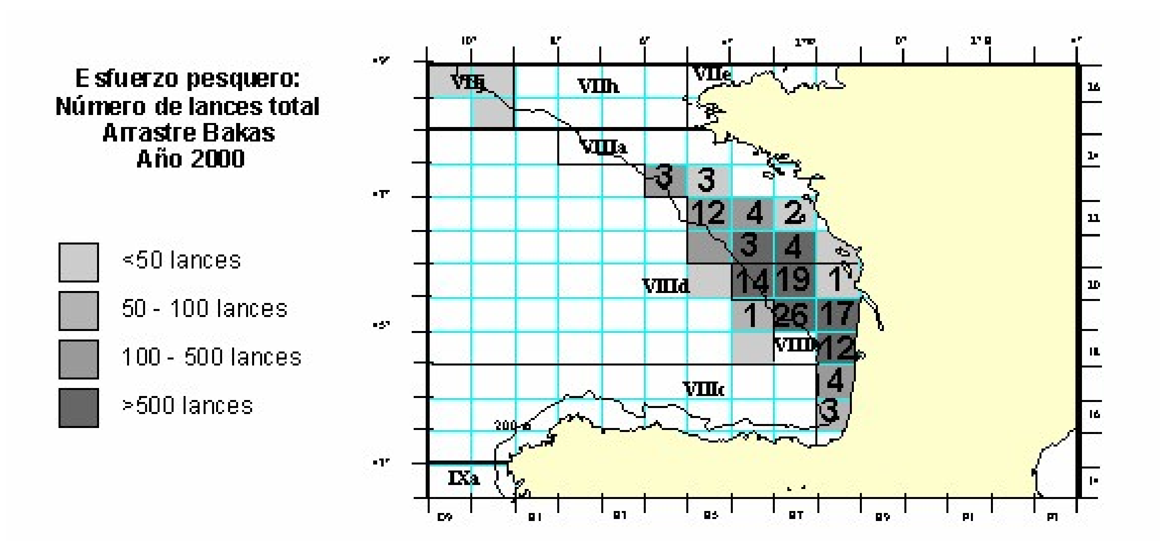

Figure 4.26. Distribution of fishing effort (coloured areas) and effort level sampling (number of hauls observed) for the "Baka" bottom trawl in the Bay of Biscay in year 2000.

4.3.4.2 VHVO bottom pair trawl

The VHVO bottom pair trawl fishing metier for hake started in 1993 with one pair and increased to 14 pairs in 1999 and then decreased to 9 pairs in 2002. This reduction in the number of vessels has resulted in a reduction of the total fishing effort for this metier targeting hake. The distribution of effort for this metier is less than of the ‘Baka’ metier in 2000 and 2001 and it is concentrated on ICES Divisions VII h, j and VIII a, b and d. The approximate total effort for years 2000 and year 2001 were respectively 7,341 and 4,920 tows in VIII a, b and d and 870 and 759 in VII h and j.

The VHVO bottom pair trawl metier is the only one in which cetacean bycatch has been observed. These bycatches have only been observed in ICES Divisions VIII a, b and d. The spatial distribution of the bycatches is shown for the years 2000 and 2001 (Figure 4.27). During this period a total of 199 hauls was observed: in 194 hauls no bycatch was observed and in five hauls bycatch was observed. More hauls were observed in year 2001 than in 2000.

For year 2000, a total of 81 hauls were observed in ICES Divisions VIII a, b and d (1.1% of the total effort in these Divisions), four hauls with cetacean bycatch (all common dolphins) were detected in these ICES Divisions in February (1), October (1) and November (2) (Table 4.18). The statistical rectangles with bycatch were different in each month, 23E5, 19E6, 20E6 and 19E7 respectively (Figure 4.28). In 2000, there were no observers on board this fleet in Divisions VII h and j.

Table 4.18. Monthly distribution of the number of days at sea, hauls observed and bycatch in the VHVO bottom trawl in ICES Divisions VIII a, b and d for year 2000.

	Jan	Feb	Mar	Apr	May	June	July	Aug	Sept	Oct	Nov	Dec	Total
Days at sea	0	18	7	6	8	0	0	0	6	4	19	3	71
Hauls observed	0	20	9	7	9	0	0	0	7	5	21	3	81
Hauls with bycatch		1	0	0	0				0	1	2	0	4

In year 2001, 118 hauls were observed (2.4% of the total effort in ICES Divisions VIII a, b and d) and only one haul had a cetacean bycatch in February in statistical rectangle 23E5 (Figure 4.28). Common dolphin was the cetacean species involved in bycatch. For this year, 38 hauls were observed in ICES Divisions VII h and j (5% of the total estimated effort in these Divisions) and no cetacean bycatch was recorded in them.

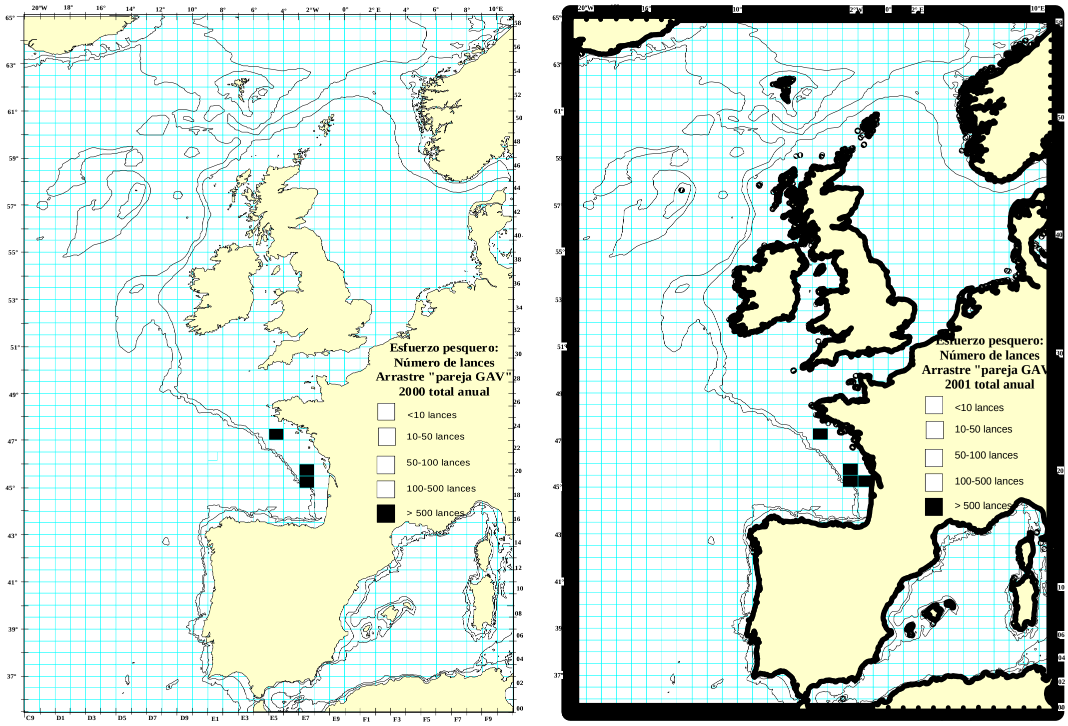

Figure 4.27. Annual distribution of effort of the VHVO bottom trawl for 2000 and 2001 (number of hauls by ICES statistical rectangle).

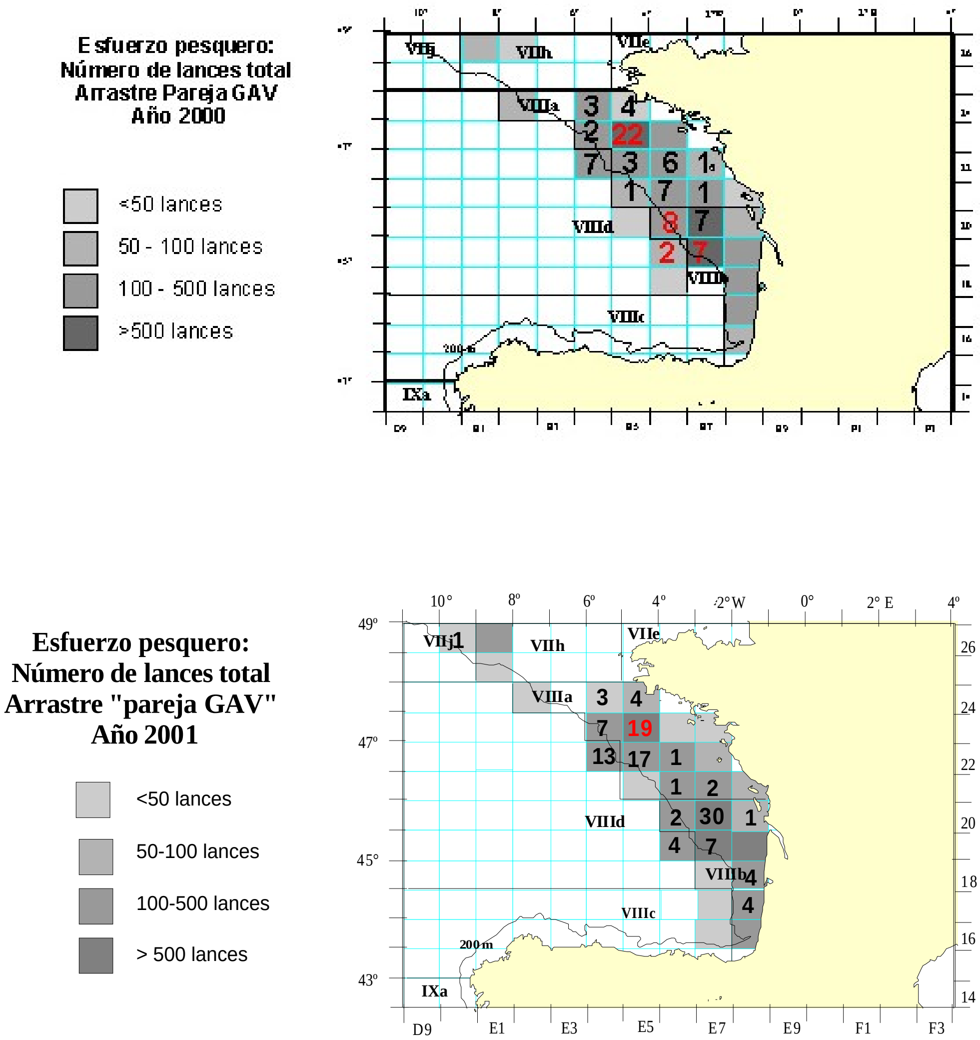

Figure 4.28. Spatial distribution of observations of the VHVO bottom trawl in years 2000 and 2001 (number of hauls observed per ICES statistical rectangle in ICES Divisions VIII a, b and d). The numbers in red indicate those rectangles in which cetacean bycatch was observed.

In conclusion, from the investigated metiers based in the Basque Country, incidental cetacean bycatch was observed in autumn and winter in only one fleet, bottom pair trawl operating with VHVO and targeting hake in ICES Divisions VIII a,b,d. In this fishery, a reduction of the fleet size and total effort has been observed since 1999 and only 9 bottom pair trawlers were concerned in 2002. No cetacean bycatch was detected in VII h,j. For the two other trawl metiers present in the Basque fishing fleet ("Baka" bottom trawl operating in Sub-areas VI and VII and Div. VIII a,b,d, and bottom pair trawl operating with VHVO nets in VIIIc) no cetacean bycatch has been observed in all the cruises with AZTI observers.

No extrapolation seems possible as the observations represent in general less than 3 % of the fishing effort in the VHVO bottom trawl fleet in Div. VIII a,b,d.

4.3.5 Information on bycatch of common dolphins in Portuguese fisheries

There has been no dedicated observer scheme to record bycatch in Portuguese fisheries, but Silva and Sequeira (2003) report on 124 bycaught animals noted from 39 separate fishing events between 1975 and 1998. Six different fisheries operating off the Portuguese coastline caught dolphins. Gill nets were responsible for the largest number of occurrences (n = 23, 59%) and had captured more than 67% (n = 84) of the dolphins. Beach seine nets and trawling operations killed respectively 11% and 9% of the individuals, with the former being only involved in four of the 39 by-catch events.

4.3.6 Information on bycatch of common dolphins in Danish fisheries

No information on bycatch in Danish fisheries was available to the Working Group, although we understand that some is to be supplied to the European Commission later in 2005.

4.3.7 Information on bycatch of common dolphins in Netherlands fisheries

Couperus (1997a and b) describes the incidental catch of cetaceans in Dutch pelagic trawls as found from an independent observer programme that covered about 5% of the annual effort of this fishery between 1992 and 1994. In parallel with this independent observer scheme, a self-reporting scheme was set up that covered the same fishery during the last two years of the study. With the addition of some further records from 1989–1991, a total of 71 bycatch incidents was recorded involving a minimum of 312 individual dolphins. Forty-one of these incidents (172 individuals) occurred in one year (1994). Approximately 90% of the incidents occurred in the late winter/early spring in the mackerel and horse mackerel fisheries that, at this time of year, both operate southwest of Ireland. Atlantic white-sided dolphins were the main bycaught species (83% of identified individuals), with long-finned pilot whales, common dolphins and bottlenose dolphins being caught in this area. Elsewhere (mostly in the western North Sea and the western English Channel), very few white-sided dolphins were caught and common dolphins, long-finned pilot whales, and white-beaked dolphins were present in the bycatch. About 40 % of dolphins were not identified to species level.

A new bycatch observation programme started in 2005 and the results of the first four trips on pelagic trawls (targeting either mackerel/horse mackerel or blue whiting) were made available to the Working Group. On only one of these trips was there any bycatch, when three common dolphins were caught. It is too early to comment on these data.

4.3.8 Discussion

4.3.8.1 Variation in bycatch rates

Bycatch of cetaceans involves the interaction of a human activity (fishing) with the activity of cetaceans. Both of these activities are variable in time and space, for a variety of reasons.

The variation amount of deployed effort has been described temporally above, in both the scale of months and years (seasons). At present we are not able to examine the spatial variance in this effort – this requires access to other data, such as that held in logbooks or produced by the VMS systems. Variation in fishing effort is driven by factors such as distribution and abundance of target species, market values (both absolute, and relative to other potential target species) and weather.

The variation in distribution of common dolphins has been described partially, but in contrast to variation in fishing effort the only way of improving this knowledge would be through a series of costly surveys. Another possibility would be to understand the causes of variation in common dolphin distribution. This seems most likely to be related to the relative abundance and availability of various preferred fish species. Even if we understood this cause better, there is still a lack of information on the distribution and abundance of many of the preferred prey types of common dolphins (e.g. lantern fish, squid).

The combination of these two variations (in fishing effort and dolphin distribution) inevitably means that there will be considerable variation in bycatch rates. This may be illustrated by the time series of cetacean bycatch available for the Irish tuna fishery (BIM and Ifremer data) (Table 4.19) and for UK bass fishery (Table 4.8). This means that it would be difficult to estimate an average impact by taking only one year. The recent EC regulation should provide in the future more additional observations at sea. The Working Group notes that several years of data are thus required before any 'average' bycatch rate can be estimated and then extrapolated to whole fleets. The exact number of years of data required will differ between fisheries, dependent on the degree of variation in bycatch rates.

Table 4.19. Variation in the bycatch rate of common dolphin bycatch in the tuna pelagic trawl fishery (BIM and Ifremer data).

Year	1998	1999	2002	2003	2004
Origin of sampled vessels	IR	IR	IR	IR	IR and FR
Observed hauls	144	330	113	55	133
Cetacean bycatch incidents	12	23	5	1	4
Mean number of common dolphin bycatch incidents per haul	0.08	0.07	0.04	0.02	0.03
Mean number of common dolphins per haul	0.31	0.38	0.14	0.02	0.06
Sum of cetacean bycatch	44	125	16	1	8

4.3.9 Summary of bycatch observations

Table 4.20 summarises this section. Owing to the lack of comprehensive monitoring, it is not possible at present to say how important each of these fisheries is in relation to overall cetacean bycatch. It is noticeable that many schemes either have not described the proportion of the fishery being monitored, or that proportion is low and may not therefore be representative. It is not possible to assess the statistical validity of scaling up these observed bycatch rates to a full fishery without knowing whether the sample observed is likely to be representative. Note also that representativeness is necessary in both time and space, in other words the sample needs to represent the fishery in its seasonality and geographic distribution, as well as in quantity.

If total bycatch of common dolphins in all fisheries is to be calculated, then it is probably necessary to set observation scheme sampling standards and to examine the totality of fishing effort off north-west Europe in order to determine the scale of observations required. This would be a large piece of work, but if such work is not carried out and comprehensive schemes implemented, there is the risk of not be able to determine if the total common dolphin bycatch exceeds that which would cause a unsustainable impact on the population.

Table 4.20. Summary of bycatch observation schemes reported to ICES including years of observation scheme, main target species of fishery (or metier) and percent of fishery effort observed.

Nation	Pair trawl	Gill net	Other trawl	Non-systematic
UK	2000-2005 (bass) (32%)	1999-2000 (hake) (?)		
Ireland	2002-2004 (tuna) (?)			
France	2004-2005 (several) (?)	2003-2004 (sole) (?)		
Spain (Basque country)	2000-2001 (hake) (1.3%)	1998 – 2001 (mixed) (?)	2000-2001 'Baka' (1.7%)	
Portugal				1975-1998
Netherlands	1992-1994 (mackerel/horse mackerel) (5%)			
Denmark				

4.4 Mitigation measures

4.4.1 Introduction

The common dolphin population off north-west can be roughly estimated at around 500,000 individuals by merging all the surveyed areas and by taking into account studies of genetics. This figure may be modified following the SCANS II project in summer 2005, but more likely cannot be modified until a survey is carried out in waters west of the continental shelf break. ICES (2001) advised that a bycatch of 1.7% of the harbour porpoise population per year would place that population at risk. If this figure was applied to common dolphins, that would equate to a total removal of 8,500 common dolphins per year. The total impact of fisheries needs to be calculated. At this time of knowledge any impact of such level has not been observed or suspected in a single fishery, but could occur across all fisheries combined.

Regardless of the total level of bycatch, measures to mitigate or prevent bycatch would be consistent with a precautionary approach and public expectation. This section briefly reviews possible measures.

4.4.2 Spatio-temporal measures

Measures to limit fisheries when dolphins are present (both spatially and temporally) may seem a logical and simple way to reduce bycatch in some fisheries. However, for such measures to work there needs to be a good understanding of the bycatch phenomena and of the factors inducing its variation. Without having such understanding, care should be taken in changing fishing effort from one area to another because the results may prove to be the opposite of what it was intended.

For example in the French pelagic trawling, several target species exist and hake has been shown by Morizur *et al*. (1996b) as a target fish available during the whole year but having no strong influence on the seasonal activity of the fleet. In other words, the French pair trawling activity on hake appears to be a displacement activity. It has been proven that bycatch of cetaceans exist also in that fishery. Consequently any regulation of the fisheries associated with catching hake might displace effort into catching more hake and thus potentially increasing the bycatch in that fishery.

The same dangers may exist inside a single fishery. Northridge *et al.* (in prep.) reported that a ban of an inshore area could induce higher level of fishing effort offshore, in an area where common dolphins are more abundant.

This approach to reducing the impact of fishing was discussed in depth by STECF/SGFEN in June 2002 (SGFEN 2002 a, b). WGMME agrees with the conclusions of that report and agrees that at present there are no obvious areas in the European Union where fishery closures should be proposed.

4.4.3 Pelagic trawl

A review was carried out in 2002 of mitigation measures by STECF/SGFEN (SGFEN 2002 a, b). This section focuses on current projects.

The European project NECESSITY (March 2004-March 2007), coordinated by RIVO, contains a sub-project dealing with interactions between pelagic trawl fisheries and incidental bycatch of cetaceans. The objective is to explore additional information for better understanding of incidents, to study technical measures to limit the impact on cetaceans and to assess their biological and economical effects. Different systems are to be tested, including mechanical solutions (ropes, panels, grids) and/or acoustic deterrents. Different or combined systems may need to be used depending of the target fish species.

At this stage of the project, the grid system tested by the Sea Mammal Research Unit (SMRU) of the UK and Ifremer seems promising, as it has been proven to work in the bass fisheries. Ifremer showed that the grid was useable to catch bass and SMRU has shown by video that it can exclude common dolphins (Northridge *et al*. 2005). Ifremer has carried out a study to define the acoustic parameters of an acoustic deterrent that may be used to prevent any entry into the trawl (Le Gall *et al.* 2004). The system will be tested soon at sea and later on a commercial trawler, with the objective of incorporating the repellent system inside the net sounder or any other acoustic fishing equipment attached to the gear. BIM has developed with Aquatech an interactive acoustic deterrent (BIM 2004) and a modified version is under testing by DIFRES. Next autumn AZTI will start some cruises on commercial vessels with VHVO bottom nets to test at sea a cetacean excluder device based on ropes and floats (rope barrier). There are also other national contracts providing some funds for research experiments with the same objective.

Due to its higher bycatch event rate, any mitigation research on bass fishery should help develop mitigation in other pelagic trawl fisheries troubled with a bycatch problem.

4.4.4 Bottom set nets

In fixed net fisheries, pingers have proven their efficiency on porpoises. However for dolphins there are some contrary results on the efficiency of acoustic repellents. During some tests, the deterrents appear to work, while in others they fail. The reasons for this are not fully known but it is suspected that the effect will depend on

- (i) the cetacean species involved in the interaction with fisheries,
- (ii) the fish species (prey or not) present in the nets,
- (iii) the acoustics characteristics of signals used,
- (iv) the physical characteristics and quality of pingers, linked for some of pingers with the operational conditions in use for fishing (shocks depending of the shooting and hauling speed, of the shape of vessels and the hauling equipment).

A recent study (not yet completed) made by the Seafish Authority has shown that some pingers were not working. Moreover the ‘dinner bell’ effect in medium term may not be eliminated if the cetaceans are attracted by the fish caught in the nets. This is a good reason to assess the efficiency of pingers through time as required of Member States by the recent EU regulation. WGMME assume that these studies are (or will be) occurring at a national level and note that a coordination of such studies would benefit both scientific efforts in field experiments and avoid duplication. Such coordination could be done through a workshop under ICES WGMME. There have also been concerns that the repellent effect may exclude cetaceans from areas of important habitat. This is not proven, but if it was there may be a difficult choice limiting bycatch mortality and excluding cetaceans from their natural habitats.

4.4.5 Acoustic scaring from the vicinity of fishing vessels

The sonar equipment on use on fishing vessel and their effect on bycatch has to be studied. For example, panoramic sonar (220 db) has be seen to modify the behaviour of small cetaceans in the vicinity of boats. The starting up of such sonar has been observed to cause bow-riding dolphins to flee (S. Hassani, pers. com.) This means that such sonar equipment may influence any bycatch rate by commercial vessels. The acoustic equipment in use by pelagic fleet will be collated and analysed in relation to bycatch as part of the NECESSITY project.

4.4.6 Fishing tactics in the VHVO bottom trawl

As part of the EU project NECESSITY, interviews were carried out with skippers of VHVO bottom pair trawlers, AZTI has found that some of them use tactics to avoid cetacean bycatch with its consequent loss of catch of target species and damage to the fishing gear. These tactics focus on the phases of shooting and hauling, when the gear is near the surface or near dolphins. The manoeuvre consists of keeping the boats very close together during shooting and hauling operations in order to get the mouth of the gear closed until it is in contact with the bottom. The efficiency of any modification of fishing tactics may be difficult to test experimentally as it may require a considerable number of trials. However the presence of observers on board should encourage the fishermen to adapt their fishing tactics in order to limit the impact on cetacean populations.

4.4.7 Education of fishers

It is highly probable that observations at sea modify the behaviour of some fishermen and modify their fishing tactics. This is the reason why we have to check for the representativeness of the sampled boats. However it is hoped that in medium term the presence of observers on board contributes to the teaching of fishermen, who have always shown to be interested by the biology of species. Educational programmes are also in the scope of international organisations for the conservation of cetaceans (such as ASCOBANS). In itself, education can be a simple way of mitigation.

4.4.8 Priority for mitigation measures

The determination of priority for mitigation is a delicate task as it cannot be based solely on science. However WGMME agree on the following priorities ranked by their level of importance as perceived today:

1. Encourage a better scientific understanding of the phenomena by the use of on board observers.

This could be achieved by taking advantage of the good collaboration obtained with fishermen through actual observation programmes and the application of the new EC regulation. It is presumed that fishermen know the phenomena better than scientists do. A better understanding by merging efforts of scientists and fishermen should help to assess the true impact in all fisheries and to find solutions (including modified fishing tactics) to limit any kind of impact. Moreover, biological samples from bycaught cetaceans can help in the study of cetacean populations.

2. Improve gears and/or adapt acoustic system to limit bycatch

In trawling, mechanical solutions appear more promising than acoustic solutions, but in some fisheries the two approaches may be usefully combined. For netting the efficiency of pingers have to be determined on common dolphins, with field experiments in several different fisheries.

3. Control total fishing effort

It is insufficient to focus on seasonal fisheries having a high by catch rate. Low bycatch rate combined with high fishing effort may induce significant impact on cetacean populations. Fishing effort should be controlled carefully in fisheries where the overall impact is not known.

4. Limit fishing access in seasonal fisheries

It seems difficult to assess the true effect of any temporal and/or space regulation of the fishing effort. The seasonal distribution of common dolphin is subject to variation between years and the reasons for such variation are not well known making any forecast difficult. The behaviour of fishermen is also difficult to forecast in a seasonal fishery as it may be influenced by several factors. The level of priority to be accorded to this mitigation may depend also on the state of exploitation of the target species of the fishery.

5. Increase understanding in variation in common dolphin distribution

If there was greater understanding of why common dolphins were present in certain areas and understanding of the differences in distribution between genders, then it might be possible to better target some of the mitigation techniques. Telemetry may be a particularly useful technique.

4.5 Further information requirements

4.5.1 Representativeness of observations

We have no means at present of checking the representativeness of the bycatch observation in relation to the fleet deployment data. Extrapolation cannot be carried out without having studied this representativeness. The variability observed in bycatch events means also that it is critical to take account of variation in space and time when extrapolating figures.

This point is illustrated well in the UK data in the sea bass VII fishery (see Section 1.3.1.1). In years 2000-2001, 2001-2002 and 2002-2003 bycatch rates broadly reflected fishery effort, with most effort and highest bycatch rate both being in March. It is reasonable to estimate total bycatch from observed rates of bycatch in these years. In the 2003-2004 season, this pattern was not maintained, with highest rates early in the season, when fishing effort is low.

In general, VMS data should be used to check the representativeness of the observations or to find the best way to exploit them for extrapolation. It is essential that such data be made available to scientists.

4.5.2 Improved population estimates of common dolphin

As can be seen in Section 1.2, knowledge of abundance of common dolphins off north-west Europe is somewhat fragmentary. The 2005 survey of continental shelf waters (SCANS II) will provide a new abundance estimate for these waters. However, it is obvious that large numbers of common dolphins occur also in deeper offshore waters. If bycatches are to be placed in a true population context, then it is important that a further abundance survey in these offshore waters be conducted in the near future. Very preliminary plans for this survey are under discussion among relevant scientists, and the UK has committed to provide at least some initial funding. Considerable further funding will be required if this survey is to be carried out.

4.5.3 Greater understanding of diet of common dolphins

A greater understanding of the diet of common dolphins might help in predicting their occurrence in areas and therefore better targeting any mitigation. There appears to be some large scale differences in diet within the female common dolphin population off north-west Europe. It is not known if these differences also apply to males. It would be useful to compare the seasonal migratory patterns of common dolphins with those of their main prey.

4.5.4 Studies on the efficiency of acoustic repellents (pingers)

Such studies are (or should be) occurring at a national level. It would be useful to coordinate these efforts and WGMME recommends that it begins preparation for a workshop (possibly associated with WGMME meeting) on field experiments to follow the efficiency of acoustics repellents (pingers) through time.

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5 REGNS request for marine mammal data

Term of Reference c) for each marine mammal species affected by fishing, compile data (in excel spreadsheet format) which quantifies the seasonal distribution and abundance at spatial scales, where possible, that correspond to ICES rectangles for the North Sea. The data will be submitted to REGNS secure website in preparation for the REGNS integrated assessment workshop from 9-13 May 2005. These data should, where possible, be for the period 1984-2004 to assess trends. Also where possible, provide information on diet and variation/change of this for all species described;

5.1 Introduction

The WG was not able to provide REGNS with a data set that matched their request. WGMME did provide an updated and enlarged version of the relevant chapter of the 2004 WGMME report titled “Summary of size, distribution and status of marine mammal populations in the North Sea for 2000-2004.” The Term of Reference for the 2004 meeting was: f) *start preparation to summarise the size, distribution, and status of marine mammal populations in the North Sea for the period 2000–2004, and any trends over recent decades in these populations. Where possible, the causes of these trends should be outlined for input to the Regional Ecosystem Study Group for the North Sea in 2006.*

A small database entry from Norway was also submitted to the REGNS website.

5.2 North Sea Marine Mammals

Seven marine mammal species occur regularly and frequently in the North Sea, others occur in low numbers or in small parts of the area (e.g., killer whale, Risso’s dolphin, sperm whale). The cetacean species that occur regularly are: harbour porpoise (Phocoena phocoena), white-beaked dolphin (Lagenorhynchus albirostris), Atlantic white-sided dolphin (Lagenorhynchus acutus), bottlenose dolphin (Tursiops truncates), and minke whale (Balaenoptera acutorostrata). The seal species are the harbour seal (Phoca vitulina) and the grey seal (Halichoerus grypus). A summary of current knowledge for each of the seven species is included below; WGMME requests comments from REGNS as to whether this level of information and format is suitable.

5.3 Harbour porpoise

5.3.1 Population size

The only abundance estimate in the North Sea for harbour porpoises is 262,540 individuals. This estimate was made in 1994 (Hammond *et al*., 2002) and included the whole North Sea and the Channel. The Kattegat and part of the Skagerrak had an additional estimate of 36,046 harbour porpoises.

Aerial surveys were conducted in the German waters of the North Sea in 2002 and 2003. Abundance estimates were calculated for the mean summer population in the German territorial waters and EEZ in the North Sea (size of area 41,045 km²). Mean summer abundance (May to August) was estimated to be 16,643 animals (Scheidat *et al*., 2004). A further abundance survey for the North Sea and adjacent waters is planned for 2005 if funding is forthcoming.

5.3.2 Population distribution

A newly published cetacean atlas (Reid *et al*., 2003) shows the distribution of harbour porpoises in the North Sea at the scale of 1/4 ICES rectangles. The atlas is based on the Joint Cetacean Database, contributed to by the European Seabirds at Sea database (ESAS), Sea Watch Foundation (SWF) and Sea Mammal Research Unit (SMRU). It used most, but not all, effort-related cetacean data for North-west European water for the years 1979 to 1998 and over all seasons combined. The highest sighting rates for harbour porpoises were found in the northern central North Sea (Figure 5.1). The lowest sighting rates in the North Sea were in the southeastern part, close to the German, Dutch, and Belgian coasts and in the Channel.

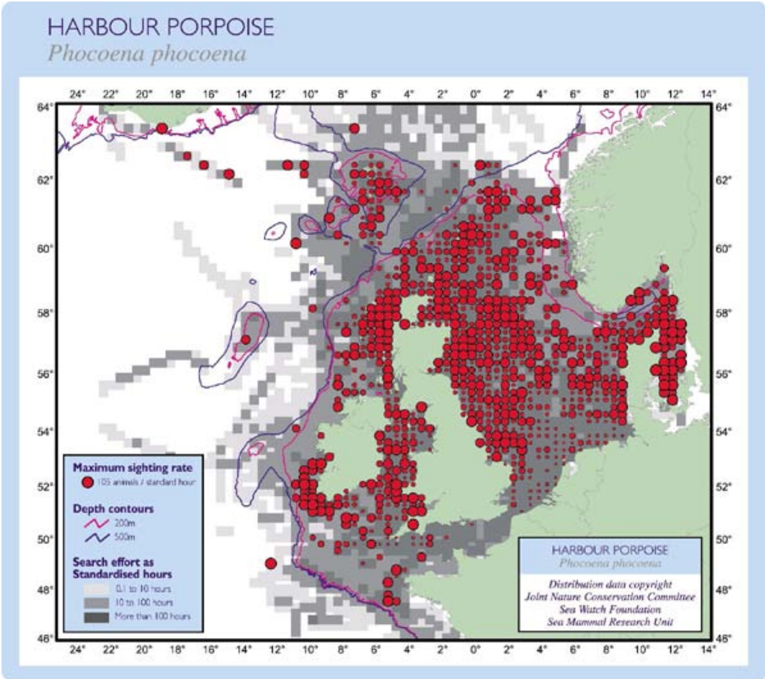

Figure 5.1. Distribution of harbour porpoises (Reid et al. 2003).

S. Hedley has re-analysed (map available at http://www.ruwpa.st-and.ac.uk/px/dens_hp.jpg) the ship-based data collected during 1994 abundance survey, modelling the expected encounter rate as a function of spatial covariates. The model showed the highest expected density of harbour porpoises was in the central and northern North Sea.

In German waters of the North Sea, harbour porpoises were not distributed uniformly in the summer months (May to August). The highest density was found in the northern part close to the Danish border (Scheidat *et al*., 2003, 2004) (Figure 5.2).

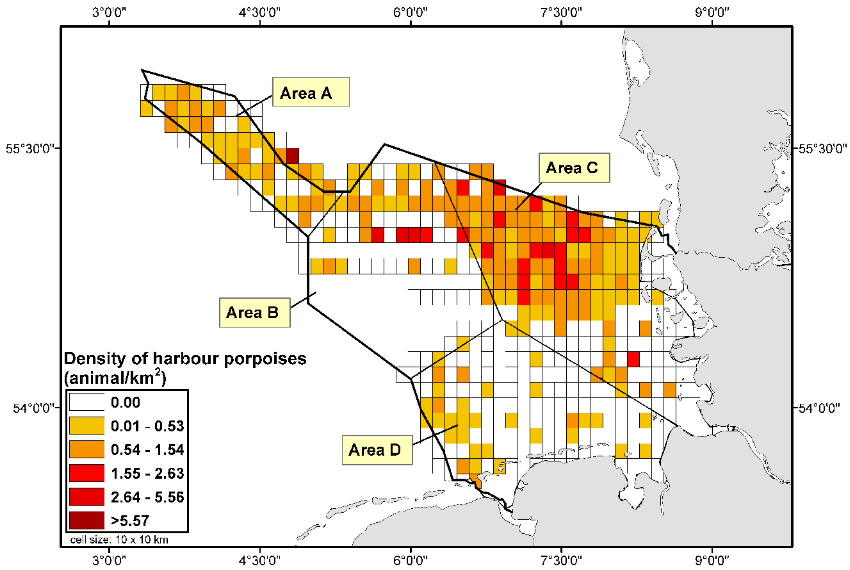

Figure 5.2. Map showing the distribution of harbour porpoises in the German North Sea for May to August 2002 and 2003. Density is shown as animals per km² per cell (10×10km²). Only flights conducted in good or moderate conditions were included (from Scheidat et al., 2004).

Seasonal occurrence has been investigated off parts of the British coast and off the Dutch coast. In Dutch waters, harbour porpoises are seen mostly from December to April (Figure 5.3).

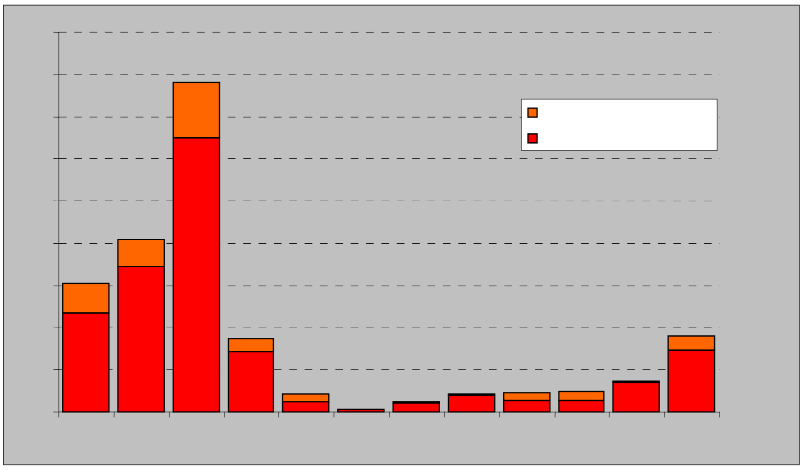

Figure 5.3. Seasonal pattern of harbour porpoises reported from coastal sites in the Netherlands since 1970 (Marine Mammal Database, updated 3/1/2004, http://home.planet.nl/~camphuys/Bruinvis.html)

5.3.3 Status

WGMME was unclear of the meaning of "status" in this context. As only one point estimate is available of abundance, no overall population trend is available. Trends in occurrence off the coast of the Netherlands since the 1970s have been compiled and published by C.J. Camphuysen (http://home.planet.nl/~camphuys/Bruinvis.html). There has been an increase in sighting rate of harbour porpoises that started in the mid-1990s and continued to 2004 (Figure 5.4).

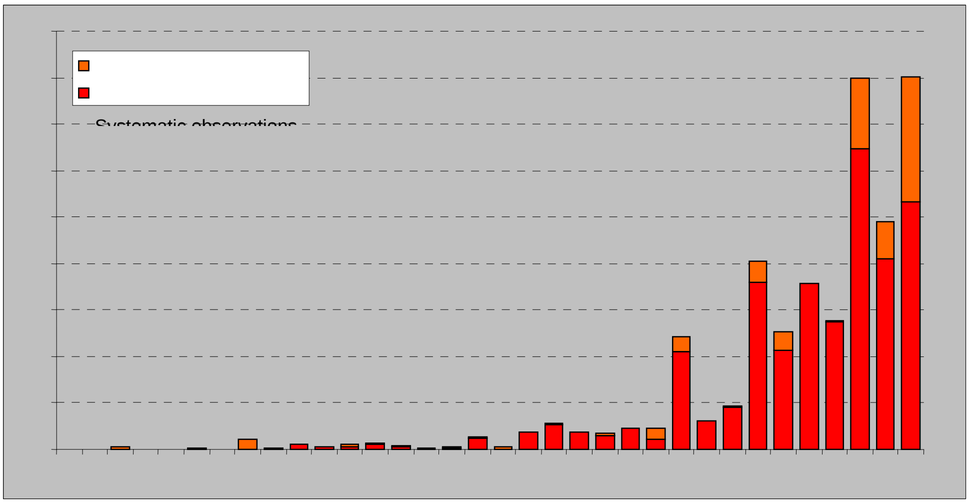

Figure 5.4. Harbour porpoises reported from coastal sites of The Netherlands since 1970 (Marine Mammal Database, updated 3/1/2004, http://home.planet.nl/~camphuys/Bruinvis.html.)

Similarly, the strandings along the Belgian coast have increased (Haelters *et al.*, 2002 and pers. comm) (Figure 5.5). The increase in sightings and strandings along the Dutch and Belgian coasts could mirror a change in distribution of porpoises, but the reason for this change is not known. Camphuysen and Leopold (1993) suggest that there might have been a change in food availability.

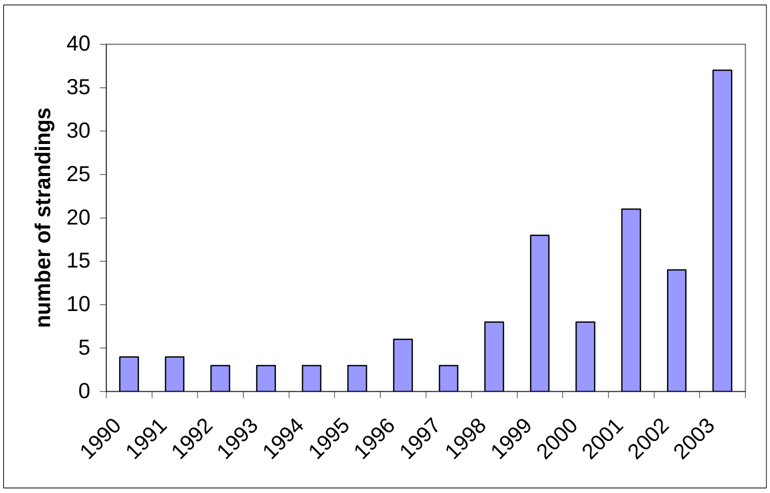

Figure 5.5. Number of harbour porpoise strandings on Belgian coasts, 1990 to 2003. From Haelters et al. (2002) and J. Haelters, pers. comm.

In terms of conservation status, harbour porpoise are listed in Appendix II of CITES (Convention on International Trade in Endangered Species of Wild Fauna and Flora), and categorised as "Vulnerable" (Vu A1cd) by the IUCN. They are listed in Annexes I and IV of the EU Habitats Directive and appear on OSPAR's initial list of threatened and declining species.

Some information is available on the status of contaminants in harbour porpoises in the North Sea, and the degree of parasitism/disease.

5.4 White-beaked dolphin

5.4.1 Population size

The small cetacean abundance survey in 1994 estimated a summer population of 7,856 animals (CI 4,032–13,301) in the North Sea and the Channel (Hammond *et al.*, 2002). Some sightings of *Lagenorhynchus* dolphins were not specifically identified. An abundance estimate of 11,760 (5,867–18,528) dolphins was obtained when all sightings of *Lagenorhynchus* were combined.

5.4.1.1 Population distribution

During the 1994 abundance survey (Hammond *et al.*, 2002), all records of white-beaked dolphins were made in the North Sea and the area directly NW of Scotland, between c. 54°–60°N, 6°W–7°E.

Figure 5.6 shows the distribution of the white-beaked dolphin in the North Sea from most effort-related data that are available between 1979 and 1998 (Reid *et al.*, 2003). The species occurs over a large part of the North Sea continental shelf, north of the Flamborough Head to Jutland front (Reid *et al.*, 2003).

5.4.1.2 Population status

As only one point estimate is available of abundance, no overall population trend is available. No trend in occurrence has been reported.

The species is not listed by IUCN (despite being considerably rarer and with a narrower distribution than harbour porpoise). The species is listed in Appendix II of CITES and in Annex IV of the EU Habitats Directive.

Limited information exists on disease, contaminants, and parasites in individuals found dead.

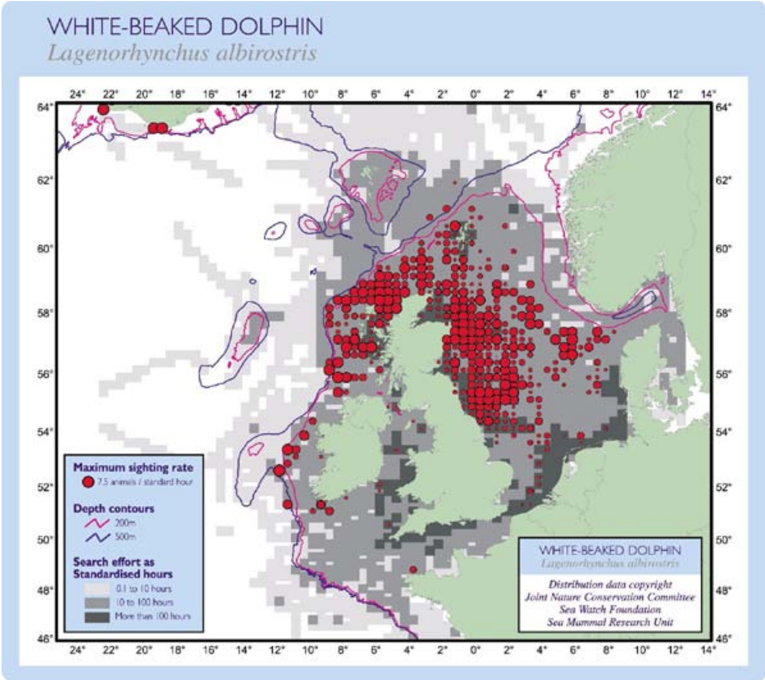

Figure 5.6. Distribution of white-beaked dolphins (Reid et al., 2003).

5.5 Atlantic white- sided dolphin

5.5.1 Population size

During the SCANS survey abundance estimate was calculated for both *Lagenorhynchus* species together at 10,927 animals (Hammond *et al.*, 2002), but not for this species alone.

Weir *et al*. (2001) carried out surveys to the north and west of Scotland, partly in the North Sea, and found that Atlantic white-sided dolphin was the most abundant species in the region with a total of 6,317 animals recorded.

5.5.1.1 Population distribution

In the North Sea, the Atlantic white-sided dolphin is mainly found in the far north and to the west of Shetland (Figure 5.7) (Reid *et al.*, 2003).

5.5.1.2 Population status

As only one point estimate is available of abundance, no overall population trend is available. No trend in occurrence has been reported.

The species is not listed by IUCN (despite being considerably rarer and with a narrower distribution than harbour porpoise). The species is listed in Appendix II of CITES and in Annex IV of the EU Habitats Directive.

Limited information exists on disease, contaminants, and parasites in individuals found dead.

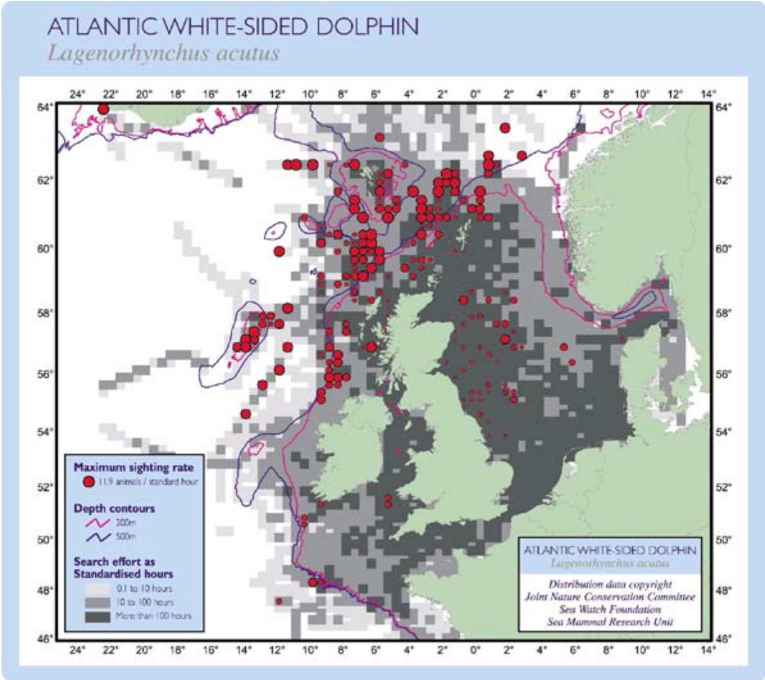

Figure 5.7. Distribution of Atlantic white-sided dolphins (Reid et al., 2003).

5.6 Bottlenose dolphin

5.6.1 Population size

The population of bottlenose dolphins in the Moray Firth is estimated at 129 (95% CI 110–174) animals (Wilson *et al.*, 1997). A collaborative photo-identification project has catalogued 85 individuals in the Channel, including northwest France (Liret *et al.*, 1998).

5.6.1.1 Population distribution

In the North Sea, bottlenose dolphins are found in the Moray Firth and off eastern Scotland and in coastal areas of the western Channel (Figure 5.8).

5.6.1.2 Population status

Wolff (2000) notes that bottlenose dolphins have disappeared along the Dutch coast in the last few decades. Prior to this, bottlenose dolphins were moving into the Zuiderzee every spring apparently following herring shoals. The herring disappeared in this area in 1937, but bottlenose dolphins still stranded on the coast until around 1965. Then the numbers dropped further and the bottlenose dolphin is not considered a resident species in the southeastern North Sea any longer (Verwey and Wolff, 1981; Bakker and Smeenk, 1990).

The species is listed as “data deficient” by IUCN. The species is listed in Appendix II of CITES and in Annexes II and IV of the EU Habitats Directive.

Limited information exists on disease, contaminants, and parasites in individuals found dead.

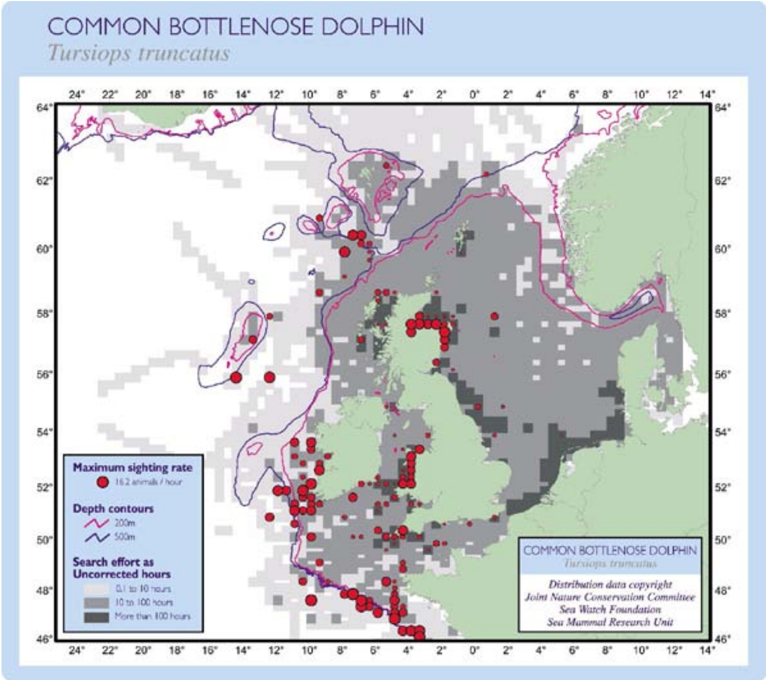

Figure 5.8. Distribution of bottlenose dolphins (Reid et al., 2003).

5.7 Minke whale

5.7.1 Population size

The Hammond *et al.* (2002) estimate has been revised from 7,201 to 8,400 (95% CI 5,000–13,500). The new "Schweder *et al.*" abundance estimate for the Norwegian Sea and Barents Seas is 107,205 (CV= 0.13). The estimate is lower than the 1995 estimate OF 112,000 (95% CI 91,000–137,000). The lower estimate may be related to multi-year survey design.

5.7.2 Population distribution

During the 1994 survey, minke whales mostly detected in the north-western North Sea (north of 55°N and west of about 4°E) and in the western English Channel.

Minke whales appear to be more abundant in the western part of the North Sea (but with a cluster of sightings in the centre of the North Sea between 56º30' and 58º30' N and 0-2º E) (Reid *et al.*, 2003, Figure 5.9).

5.7.3 Population status

The two abundance surveys reported above covered differing areas with differing sampling strategies, no overall population trend is available. Although the Northeast Atlantic population appears to be stable, there are variations in patterns of occurrence between surveys.

The species is listed as lower risk/near threatened by IUCN. The species is listed in Appendix I of CITES and in Annex IV of the EU Habitats Directive.

Limited information exists on disease, contaminants, and parasites in individuals.

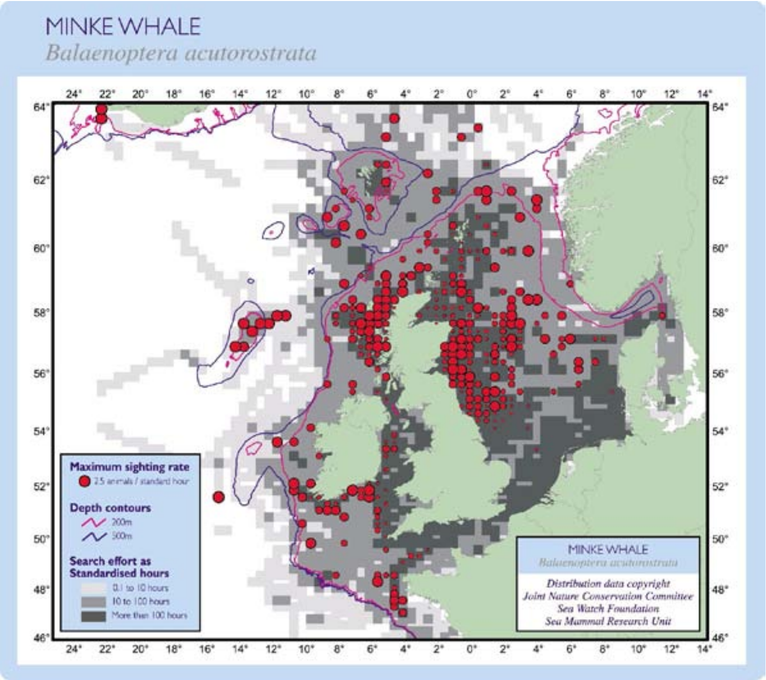

Figure 5.9. Distribution of minke whales (Reid et al., 2003).

5.8 Harbour seal

5.8.1 Population size

Harbour seals breed more widely around the North Sea than grey seals and BDC 2004 agreed that 15 sections of coast should be used to describe population trends (Table 5.1). Further details on trends are available for a number of sub-areas (Table 5.2) and these are detailed in following paragraphs. Harbour seals have been affected by two epizootics in recent years that have caused dramatic declines in numbers, particularly in the southern and eastern North Sea. The results of the first such epizootic (1988-89) are included below, but those of the second (2002-03) are not. Consequently current population sizes will be smaller than those shown here.

Table 5.1. Subunit boundaries for the North Sea seal populations. Superscripts indicate the counting technique.

Harbour seal	
UK	Shetland[1]
	Orkney[1]
	North and East Scotland[1,2,3]
	South-east Scotland[2]
	Greater Wash/Scroby Sands[2]
Netherlands	Delta area[?]
Germany	Schleswig-Holstein Wadden Sea[2]
	Niedersachsen/Hamburg W. Sea[2]
	Helgoland[3]
	Wadden Sea[2]
Denmark	Wadden Sea[2]
	Limfjord[?]
DK + SE	Kattegat[2]
DK, SE + N	Skagerrak & Oslofjord[2]
Norway	

[1] Aerial surveys using thermal imaging
[2] Aerial surveys using oblique photography
[3] Land-based counts

Table 5.2. Current estimates of abundance of harbour seals in the North Sea.

AREA	YEAR	ESTIMATE			TREND	
		Hauled out	CI	Total	Years	Estimate
Shetland	2001	4883	na	na		
Orkney	2001	7752	na	na		
North and East Scotland	1997, 2004	1944	na			
South-east Scotland	1997	40	na	na		
Greater Wash	2004	3143		na		
Scroby Sands	2001	75	na	na		
Other UK east coast sites	1994, 2000, 2002	225				
South and west England (estimated)				20		
Total UK North Sea		18062				
Delta area Netherlands	2000	97			1989 2000	+21 %
Wadden Sea, Netherlands	2000	3330			1989 2000	+18.2 %
Wadden Sea Niedersachsen	2002	6481			1989 2000	
Wadden Sea, Schleswig- Holstein	2002	7876			1989 2000	
Wadden Sea Denmark	2000	2140			1989 2000	
Wadden Sea total	2000	18000	na	na	1989–1999	+13 %**
Limfjord east	2000	410		732.1	1998 2000	-46 %
Limfjord west	2000	85		151.8	1998 2000	- 5 %
Limfjord total	2000	495		883.9	1998 2000	-40 %
Kattegat	2000	5814	696	10400	1988-2000	+9.4 %*
Skagerrak	2000	3658	596	6500	1988-2000	+14.2 %
Oslofjord	2000	280	56	500	1988-2000	+12 %
Kattegat-Skagerrak total	2000	9752		17414	1988–2000	
Norwegian west coast	1996–1998	2285	na	na		

*For the period 1996 2000 the rate of increase was 5.2%
**=6 % for 1998 2000

Harbour seal populations in the UK are monitored using aerial surveys. These take place at the height of the moulting season (August) when the greatest proportion of the population is present on land. Surveys use a thermal imager mounted in a helicopter allowing seals to be identified using a heat trace. There is currently no reliable method for translating the number of seals counted to an estimate of the total population or to an estimate of the productivity of the population. Therefore, these counts represent indices of minimum population size. Costs and logistics also mean that it is only possible to carry out annual surveys of sub-sections of coastline. The objective is to survey the whole of Scotland on a 5-year time cycle. Specific regions, such as the Moray Firth, Firth of Tay and The Wash are surveyed more frequently using fixed-wing aircraft. Time-series of counts for particular locations on the UK coast of the North Sea are presented in Tables 5.3 and 5.4 and Figure 5.10.

Even though these surveys have been conducted regularly they are mostly insufficient in number to allow an estimate of the trend in abundance within a particular area. With the possible exception of The Wash, it is also difficult to interpret trends in abundance in particular regions because of the inherent inaccuracies in the survey methods. However, it is thought that the decline in the abundance of harbour seals in the Moray Firth (Table 5.3) is real even though it is not currently possible to provide a level of statistical confidence in this conclusion. At present there is no reliable way of relating the current estimates of harbour seal abundance to the total pup production for the species. Therefore, based on the current definition of the EcoQO, these data would not provide the necessary information about trends in pup production.

Table 5.3. Numbers of harbour seals in the Inner Moray Firth.

LOCATION	07/08/92	13/8/94	15/8/97	11/8/00	11/8/02
Ardersier	154	221	234	191	110
Beauly Firth	220	203	219	204	66
Cromarty Firth	41	95	95	38	42
Dornoch Firth	662	542	593	405	220
Inner Moray Firth Total	1077	1061	1141	838	438

Table 5.4. Numbers of harbour seals in the Firth of Tay.

LOCATION	13/8/90	11/8/91	07/08/92	13/8/94	13/8/97	12/8/00	11/8/02
Eden Estuary	31	0	0	80	223	267	341
Abertay & Tentsmuir	409	428	456	289	262	153	167
Upper Tay	27	73	148	89	113	115	51
Broughty Ferry		83	97	64	35	52	
Buddon Ness		86	72	53	0	113	109
Firth of Tay Total	467	670	773	575	633	700	668

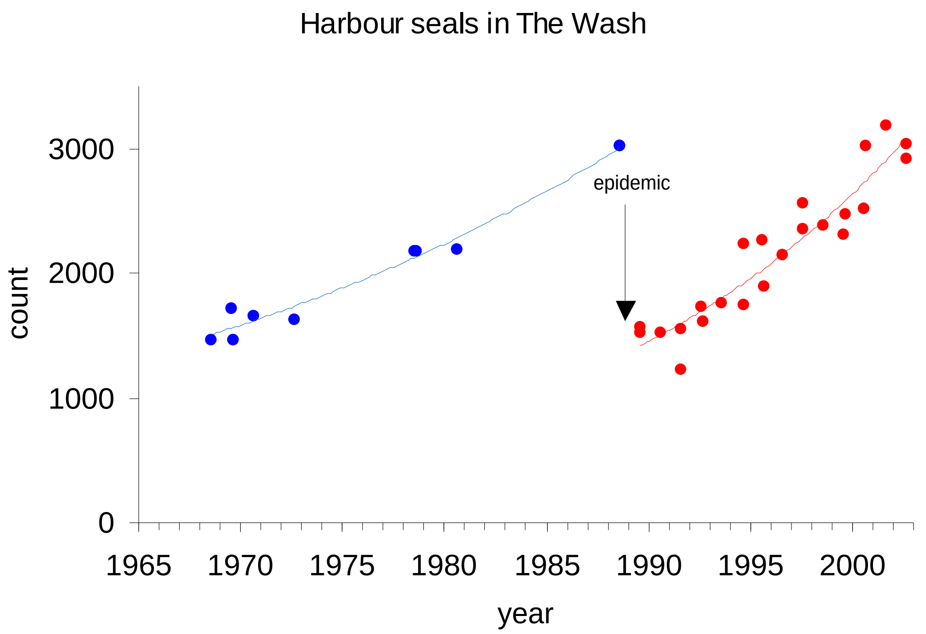

Figure 5.10. Counts of harbour seals in The Wash in August. These data are an index of the population size through time. Fitted lines are exponential growth curves.

Counts of harbour seals in the Wadden Sea are also undertaken by aircraft (Table 5.5, Figure 5.11) and a time series is available from 1975 onwards. In 2003, the maximum number of common seals counted during the moult period (August) in the Wadden Sea was around 10,800 animals. A high birth rate of pups was noted in June 2003 leading to an expectation of a quick recovery of the population from the massive decline in 2002 due to the seal epizootic.

Table 5.5. Time series of counts of harbour seals from the Wadden Sea

YEAR	NETHERLANDS	NIEDER-SACHSEN	SCHLESWIG HOLSTEIN	DENMARK	WADDEN SEA TOTAL
1975		1049	1749		3492
1976		1163	1682		3526
1977		1140	1741		3622
1978		1228	1712		3620
1979		1109	1856		3745
1980		1298	2025		4410
1981		1441	2200		4672
1982		1543			5247
1983		1777			5851
1984		1936	3300		6249
1985		2062			6878
1986		2272			7740
1987		2400	3986		8790
1988		2508	4124		9800
1989		1401	1685		4355
1990		1620	1930		5005
1991		1924	2304		5921
1992		2255	2792		6988
1993		2482	3269		8107
1994		3111	3266		8916
1995		3214	3745		9761
1996		3529	4537		11013
1997		4319	5003		12927
1998		4588	5568		14446
1999		4809	6134		15244
2000		5233	6700		17008
2001		6223	7534		19387
2002		6481	7876		20975
2003	2365	3050	4235	1160	10810

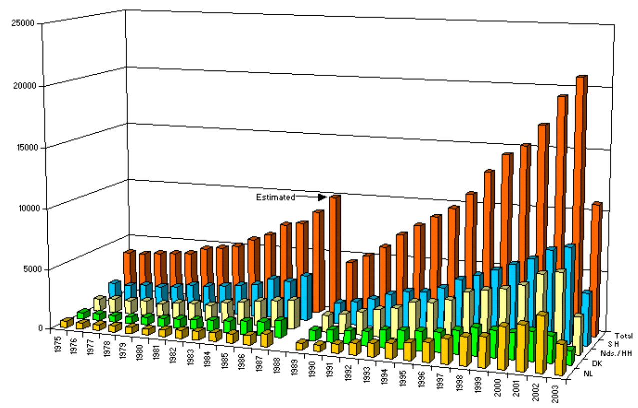

Figure 5.11. Number of harbour seals counted in the Wadden Sea since 1975 (NL = Netherlands; DK = Denmark; Nds/HH = Niedersachsen and Hamburg, SH = Schleswig-Holstein.

5.8.2 Population distribution

It has been long thought that the distribution of harbour seals in the North Sea was predominantly coastal (Figure 5.12). This impression is though probably erroneous. During a study to assess the environmental impact of an offshore windmill park (Horns Rev), a total of ten harbour seals were caught on three separate occasions on the islands of Rømø and Mandø and tagged with satellite-linked position and time-depth recorders. The first transmitters were deployed in early January 2002 and the last transmissions were received in late June/early July 2002. The transmitters provided detailed information on the movement of the animals in the Wadden Sea and the North Sea as well as detailed information on dive and haul-out behaviour (Tougaard *et al.*, 2003). Positional information revealed that animals move about more extensively than previously believed. Substantial variation between individuals and time of year was observed, with some animals, especially the pups, exploiting areas of more than 10,000 square kilometres (maximum 72,000 km^2), whereas others remained more local in the area just west of the Wadden Sea. The foraging area of Danish Wadden Sea harbour seals extends from the northern German Bight and covering most of the Danish North Sea territory, stretching to the central North Sea (including the oil fields) and into the southern Norwegian North Sea sector (Figure 5.13).

Early results from similar satellite telemetry studies off eastern Scotland indicate that a substantially wider area of the North Sea is used by harbour seals in that area than was previously thought (C.Duck, pers comm.).

5.8.3 Population status

The species is not listed by IUCN. The species is listed in Annexes II and IV of the EU Habitats Directive.

Information exists on the health status of the population.

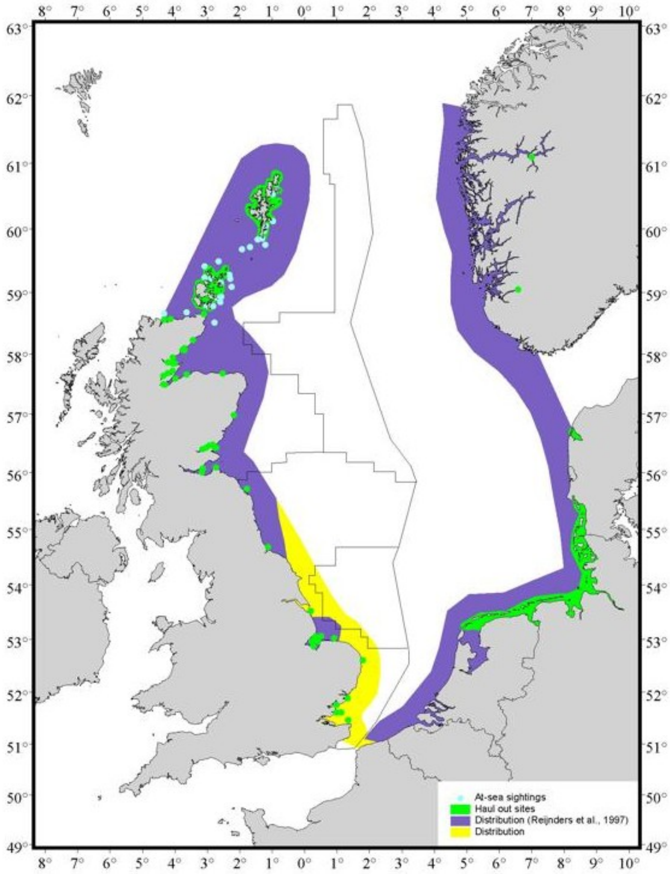

Figure 5.12. Distribution of harbour seals in the North Sea. Map extended from Reijnders et al. (1997) to take into account additional known haul-out sites in the southwestern North Sea. At-sea sightings from Pollock et al. (2000) are also shown. Source: DTI, 2002.

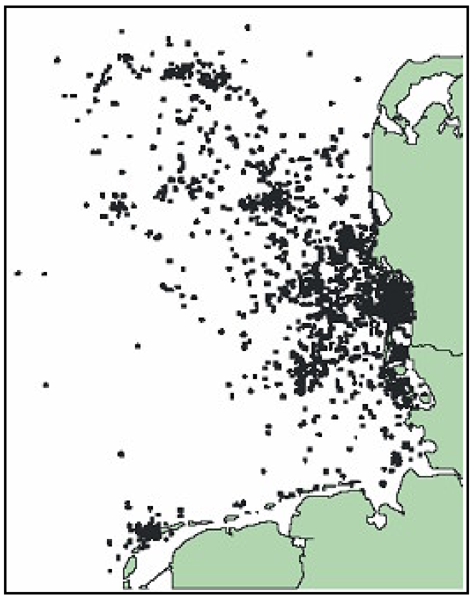

Figure 7.13. Telemetry data from harbour seals tagged at Horns Rev (http://www.hornsrev.dk/).

5.9 Grey seal

5.9.1 Population size

Grey seal populations sizes in the North Sea are estimated by extrapolating from counts of pups.

5.9.1.1 Norway

Grey seal surveys were undertaken along the Norwegian coast in 2000–2002 (Nilssen *et al*., 2003). In Rogaland, pupping occurred only on the Kjør Islands where 28–30 pups were counted each year in the period 2000–2002, which gives an abundance estimate of 128–160 seals (1+). No whelping was observed between the Kjør Islands in Rogaland and Froan in Sør-Trøndelag.

5.9.1.2 UK

In the British population, the total number of pups born in 2002 at North Sea sites was 4,418 (and 17,598 in Orkney). Orkney produces 80% of the pups born in colonies bordering the North Sea and is the location in the UK with the largest grey seal pup production. Pup production at Orkney increased year on year by about 8% per annum until 1997. The increase has continued since then, but at a slower rate of 4.6% per annum (Table 5.6).

The grey seal breeding population at the Farne Islands has been managed in the past both by culls of adults in 1972 and 1975 and by small culls of pups born on specific islands up to the present day. Consequently, there has been a highly variable rate of increase at this location. A probable consequence of the management activities at the Farne Islands was the establishment of satellite colonies at the Isle of May, Fast Castle, and Donna Nook. The Isle of May and Fast Castle are considered here as a single location. Both the Isle of May/Fast Castle and Donna Nook sites have shown relatively rapid annual rates of increase, although the increase at the Isle of May/Fast Castle appears to have reduced in recent years. The pup production attributable to further North Sea locations that are not included in the annual surveys amounted to about 3765 pups or about 17% of the total pup production on the UK North Sea coasts (OSPAR, 2004).

Table 5.6 The number of grey seal pups born at each of the major UK breeding sites bordering the North Sea. (OSPAR, 2004).

YEAR	ORKNEY	ISLE OF MAY AND FAST CASTLE	FARNE ISLANDS	DONNA NOOK	TOTAL
1984	4,741		778	30	5,549
1985	5,199		848	53	6,100
1986	5,796		908	35	6,739
1987	6,389		930	72	7,391
1988	5,948		812	54	6,814
1989	6,773		892	94	7,759
1990	6,982		1,004	152	8,138
1991	8,412		927	223	9,562
1992	9,608	1,251	985	200	12,044
1993	10,790	1,454	1,051	205	13,500
1994	11,593	1,325	1,025	302	14,245
1995	12,412	1,353	1,070	334	15,169
1996	14,273	1,567	1,061	310	17,211
1997	14,051	2,032	1,284	382	17,749
1998	16,352	2,241	1,309	439	20,341
1999	15,455	2,034	843	503	18,835
2000	16,281	2,514	1,171	618	20,584
2001	17,928	2,253	1,247	634	22,062
2002	17,598	2,509	1,200	709	22,016
2003	18,652	2,599	1,266	792	23,309

5.9.1.3 Germany

Relatively few grey seal pups are born on German coasts (Table 5.7). There is a gradual increase in numbers, but note the large inter-annual fluctuations (SCOS 2003).

Table 5.7 The number of grey seal pups born at regular German breeding sites in the North Sea, 1988–2004.

SEASON	JUNGNAMENSAND (SCHLESWIG-HOLSTEIN)	HELGOLAND	TOTAL
1988/89	9		
1989/90	2		
1990/91	6		
1991/92	5		
1992/93	9		
1993/94	4		
1994/95	3		
1995/96	8		
1996/97	8	1	9
1997/98	9	2-3	~11
1998/99	9	2-3	~11
1999/00	10	5	15
2000/01	11	?	11+
2001/02	21	6	27
2002/03	24	8	32
2003/04	~23	7	~30

5.9.1.4 Total numbers of grey seals breeding in the North Sea

Table 5.8 shows current estimates for total numbers in the North Sea.

Table 5.8. Current estimates of abundance of grey seals in North Sea waters (SCOS, 2004).

REGION	YEAR	ESTIMATE OF ABUNDANCE
UK	2002	56,600
Germany	1998	100
The Netherlands	2000	500
France		>80
Norway	2003	35 (pup count, not extrapolated)

5.9.2 Population distribution

The distribution of grey seal pupping (North Sea) and moult haul-out is well-known and included in Tables 5.6, 5.7, and in parts of Section 5.9.1.

The UK's Sea Mammal Research Unit has been undertaking a programme using satellite tags to determine grey seal distribution at sea for the past (15) years. Results are reported periodically (e.g., Figure 5.14). These results have also been spatially modelled using geophysical and hydrographic variables to provide predictive maps of areas likely to be most favoured by grey seals (e.g., Figure 5.15).

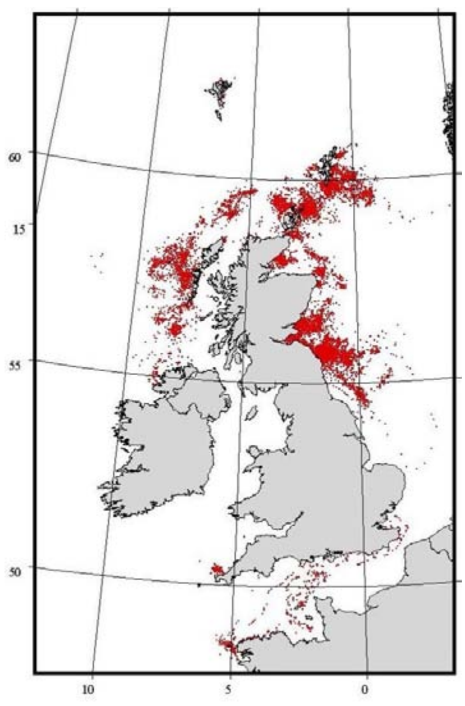

Figure 5.14. Locations of 108 grey seals fitted with satellite-relay data loggers over a period of about ten years (McConnell et al., 1999).

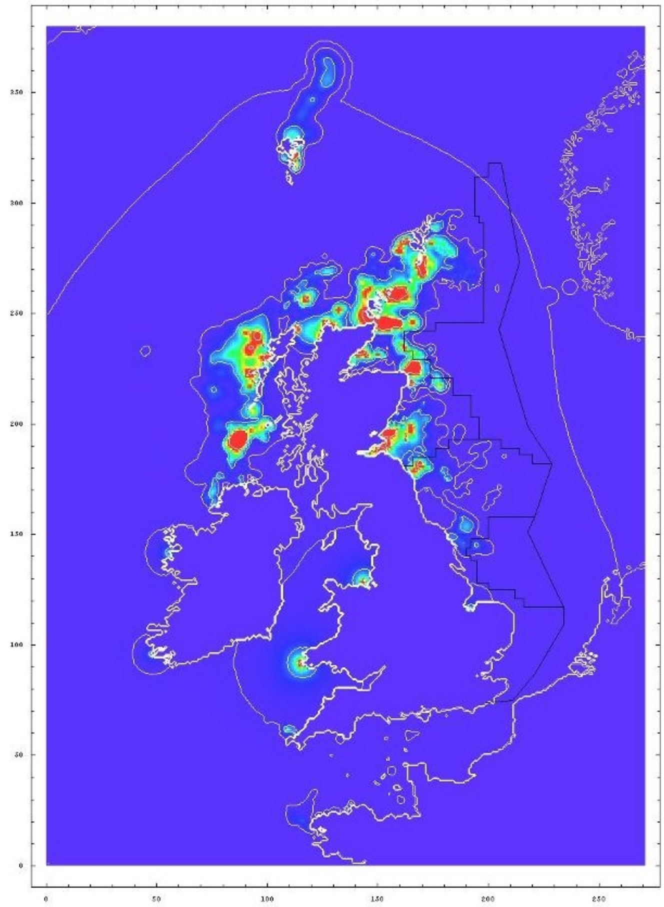

Figure 5.15. Distribution of grey seals foraging around the British Isles (predicted by a spatial model using the satellite-linked telemetry data from Figure 15 and other SMRU unpublished data). Source: Matthiopoulos et al. (in press) cited in: DTI, 2002.

5.9.3 Population status

Trends in pup production in UK and German North Sea colonies over the past twenty years are shown in Tables 5.6 and 5.7, summarised in Table 5.9. Pup production remained nearly static between 2000 and 2001 and showed a small decline in 2002.

Table 5.9. The mean annual rate of change in grey seal pup production during five-year periods from 1987 to 2002. (OSPAR, 2004).

YEARS	ORKNEY	ISLE OF MAY AND FAST CASTLE	FARNE ISLANDS	DONNA NOOK	OVERALL
1987–1992	8.5		1.1	22.7	10.3
1992–1997	7.9	10.2	5.4	13.8	8.1
1997–2002	4.6	4.3	-1.3	13.2	4.4

Grey seals were extinct in the Wadden Sea area (southeastern North Sea) for centuries (Reijnders *et al*. 1995). Some 25 years ago, grey seals started to re-establish themselves in a few colonies both off the German island of Amrum and in the Western part of the Dutch Wadden Sea (Reijnders *et al.*, 1995; Abt, 2002). Most probably, the animals originated from the UK, possibly the Farne Islands where grey seals are abundant. In Dutch waters, the development of the colony was established in about the same period (in the late 1970s); surveys during the moult have been showing an annual increase of 20% in average, amounting to over a thousand animals counted during the moult in 2003 (Reijnders and Brasseur, 2003a).

This is a very high growth rate that can only be explained by a continuous influx (likely from the British Islands) (Reijnders *et al*., 1995; Reijnders, 1996).

The species is not listed by IUCN. The species is listed in Annexes II and IV of the EU Habitats Directive.

Information exists on the health status of the population.

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6 Size, distribution and incidental catches of marine mammals in ICES Areas VII, VIII, IX and X

Term of Reference d) start preparations to summarize the size, distribution and incidental catches of marine mammal populations in the ICES areas VII - X.

6.1 Introduction

The oceanographic characteristics of this area (ICES Areas VII - X), with cold and warm water masses, led to the existence of a relatively diverse number of both boreal and temperate species. Information on the presence of marine mammals is based on past whaling activities, strandings on coasts and systematic and opportunistic sighting surveys (Aguilar, 1997; Aguilar and Lens, 1981; Aguilar *et al*., 1983; Berrow and Rogan, 1997; Clark and Charif, 1998; Duguy *et al*., 1989; Hammond *et al*., 2002; Harwood and Grellier, 2001; Lens, 1991; López *et al*., 2002; López *et al*., 2004; Sanpera *et al*., 1984; Sequeira and Teixeira, 1988). At least 31 cetacean species have been mentioned once or more in southern European Atlantic waters, 6 belonging to Mysticeti and 25 to Odontoceti. According to their relative presence it is possible to distinguish between common and occasional or rare species (Table 6.1).

Table 6.1. Cetacean species in ICES Areas VII - X

Common species	Common dolphin	*Delphinus delphis*
	Bottlenose dolphin	*Tursiops truncatus*
	Harbour porpoise	*Phocoena phocoena*
	Striped dolphin	*Stenella coeruleoalba*
	Long-finned pilot whale	*Globicephala melas*
	Risso’s dolphin	*Grampus griseus*
	Fin whale	*Balaenoptera physalus*
	Sperm whale	*Physeter macrocephalus*
Uncommon, occasional or rare species	Atlantic spotted dolphin	*Stenella frontalis*
	Spinner dolphin	*Stenella longirostris*
	Atlantic white-sided dolphin	*Lagenorhynchus acutus*
	White beaked dolphin	*Lagenorhynchus albirostris*
	Fraser’s dolphin	*Lagenodelphis hosei*
	Short-finned pilot whale	*Globicephala macrorhynchus*
	False killer whale	*Pseudorca crassidens*
	Killer whale	*Orcinus orca*
	Melon-headed dolphin	*Peponocephala electra*
	Pigmy sperm whale	*Kogia breviceps*
	Dwarf sperm whale	*Kogia simus*
	Cuvier’s beaked whale	*Ziphius cavirostris*
	Northern bottlenose whale	*Hyperoodon ampullatus*
	Blainville’s beaked whale	*Mesoplodon densirostris*
	Gervais’ beaked whale	*Mesoplodon europaeus*
	Gray’s beaked whale	*Mesoplodon grayi*
	Sowerby’s beaked whale	*Mesoplodon bidens*
	True’s beaked whale	*Mesoplodon mirus*
	Northern right whale	*Eubalaena glacialis*
	Humpback whale	*Megaptera novaeangliae*
	Minke whale	*Balaenoptera acutorostrata*
	Sei whale	*Balaenoptera borealis*
	Blue whale	*Balaenoptera musculus*

6.2 Distribution

Cetacean are highly migratory species with very wide ranges, presenting spatial and temporal variations in distribution over the areas covered here. Precise knowledge about the distribution and abundance is limited to the most common species.

6.2.1 Common dolphin

The common dolphin is the cetacean species most frequently seen off the Atlantic coasts of the Iberian Peninsula and also constitutes about 50 % of all the strandings in the area. It is normally found outside the 200 m. isobath, but it can also be found close to shore. They are opportunistic feeders. In the continental shelf they feed on pelagic fish such as clupeids, mackerel, horse mackerel and blue whiting and also on squids and other neritic cephalopods.

6.2.2 Bottlenose dolphin

The bottlenose dolphin is a typical coastal species, although it is also found offshore. Groups of bottlenose dolphins are resident in several inshore bays and estuaries in the British Isles and from Normandy to Portugal. The oceanic form is observed beyond the continental shelf. They feed on pelagic and demersal fish and they are often found in association with shoals of fish.

6.2.3 Harbour porpoise

Harbour porpoise is distributed along the continental shelves surrounding the Bay of Biscay. It is considered the most abundant species around the British Isles. It is locally abundant and it is found in the Celtic Sea and on French, Spanish and Portuguese coasts, but it is practically absent from the inner part of the Bay of Biscay.

6.2.4 Striped dolphin

Striped dolphins are mainly pelagic and generally associated with temperate waters, but they also occurred in shallow waters. It is one of the most frequent species in the stranding records.

6.2.5 Long-finned pilot whale

Long-finned pilot whale is one of the species most commonly encountered during surveys and is also well represented in the strandings records. The southern part of its distributional range overlaps with the short-finned pilot whale.

6.2.6 Risso's dolphin

It is considered an oceanic species but is often sighted over the continental shelf. Well represented in the strandings records.

6.2.7 Fin whale

Fin whale is an oceanic species that can be found throughout the year in the region. In the Bay of Biscay it is widely distributed in waters of more than 1000 m depth. Fin whales reached the coasts in spring and remained outside the continental shelf during summer months. This period coincides with upwelling processes along the western european coasts and the subsequent production of important zooplankton blooms upon which the fin whales were feeding. Most fin whales caught off Galicia had full stomachs of euphausiids.

6.2.8 Sperm whale

The sperm whale is a cosmopolitan species. It is found offshore in deep waters and on the slopes of the continental shelf. The stomach content shows the presence of several cephalopod species, some of them associated with deep canyons close to the coasts.

6.3 Abundance

Estimations of abundance for several cetacean species can be obtained from systematic sighting surveys that were carried out in southern European Atlantic waters for different purposes, each one with a partial and specific coverage of these ICES areas.

Large-scale line transect surveys on the North Atlantic under the acronym of NASS were carried out in 1987 and 1989 with the coordination of the IWC. Sightings made in 1989 yield population estimates for fin and pilot whales (Buckland *et al*., 1993). The MICA 93 sighting survey was carried in a smaller area of the Bay of Biscay and the adjacent oceanic region. Estimations of abundance were derived for common and striped dolphins and for the fin whale. (Goujon *et al*., 1993). The SCANS surveys obtained abundance estimations for several small cetacean species in the Celtic Sea Shelf south of Ireland and west of England in 1994 (Hammond *et al*., 1995). Estimations of abundance for different species in specific areas or with a partial coverage were also made (Rogan *et al*., 2000, López *et al*., 2004). A summary of these estimations is provided in a schematic form in table 6.2.

Table 6.2. Abundance estimates of cetaceans in the Atlantic region. (Modified from CEC, 2002).

Species	Year of estimate	ICES Area or sea area	Abundance estimate	95% Confidence limits	Method	Reference
Harbour porpoise	1994	VII+g+h+j	36,280	12,828 – 102,604	Ship-based line transect	Hammond *et al.* 1995
Bottlenose dolphin	1993 1993 1993 1990s 1991/3 1994-95 1991 1999 1995 1998-99	Brittany Mont St Michel Arcachon Sado Estuary Cornwall Dorset Cardigan Bay Shannon Estuary Dingle Bay Parts of VIIIc, IXa	30 60 6 35-40 15 5 120+ 113 12 664	Na na na na na na na 94-161 na 251-1,226	Photographic identification or direct observation Boat/land based surveys	ICES 1996 Rogan *et al.*, 2000 Ingram, 2000 López *et al.*, 2004
White-beaked and Atlantic white-sided dolphins	1994	VIIf+g+h+j	833	159-4,360	Ship-based line transect	Hammond *et al.*, 1995
Atlantic white-sided dolphin	2000	Parts of VI a&b, VII b/c, VII j&k	5,490	1,134 – 10,015	Ship-based line transect	O´Cadhla *et al.*, 2001
Long-finned pilot whale	1981–1984 1987–1989	Parts of Bay of Biscay VIII (E of 15°W) VIII (W of 15°W)	12,235 128,080	3,924–38,148 45,241–362,640	Ship-based line transect	Sanpera and Jover, 1989 Buckland *et al.*, 1993
Fin whale	1989	Parts of VIIk, VIIIe, VIIId, parts of VIIIc and parts of IXa	17,335	10,391-28,920	Ship based line transect	Buckland *et al*,. 1992

6.4 Bycatch

Cetacean by catch information for fisheries in European waters, including those carried out in ICES Areas VII-X were extensively reviewed by the STECF and previous reports of this WG (see CEC, 2002a and ICES, 2003). An update of bycatches in this region was provided in previous sections of this Report.

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7 Preparations for a future workshop on health and immune status, disease agents and links to environmental quality

Term of Reference e) begin preparations for a future Workshop (associated with WGMME meeting) on health and immune status, disease agents and links to environment quality;

7.1 Introduction

From 1995 to 2004 several ICES working groups have reviewed the effect of contaminants on marine mammal populations. Because of their high trophic level marine mammals are especially susceptible to chemical contaminants aggregating in the food web, such as halogenated hydrocarbons. The review of the effects of contaminants on the animals in 1998 lead to recommendations for a research programme that looked at the cause-effect relationships between environmental contaminants and population-level effects in marine mammals and the launching of such a programme in the following years. During several meetings of ICES marine mammal working groups it has been suggested to organise a workshop addressing the relationship between habitat quality and health aspects in marine mammals.

Two options to organise such a workshop are available. First, it could either take place as a theme session at the annual science conference, or second, as a workshop in relation to a meeting of the WGMME (Working Group on Marine Mammal Ecology). The pros of having a theme session include more attention from the diversity of scientists attending the ASC. However, it might be difficult to give sufficient time to entertain an in depth discussion as basis for a report with thorough recommendations. The pros of a workshop are mainly that it can be tailored for its purpose and sufficient time can be allocated for in depth discussions and reporting. However, to assure a high quality workshop with the right participation, funding for such a workshop needs to be acquired.

7.2 Draft proposal for a Workshop

Title: "Workshop on habitat quality and health aspects in marine mammals"

Time frame: 3-4 days

Venue: likely venues have been identified, and should precede a meeting of the WGMME

Year: earliest spring 2007

Specific topics:

1. The cause-effect relationships between habitat quality and immune and health status in marine mammals on an individual level.
2. The effects of relevant disease agents and pathogens in animals in a healthy environment and in animals where the immune system is compromised by the environmental quality.
3. Extrapolations to population level effects (e.g.: modelling work)

The two first should probably be the main topics at the workshop. The third topic might be conducted at the workshop, or treated as a following up from the workshop. The workshop should identify research needs and if possible develop advice for management actions. At the subsequent meeting of the WGMME, the working group members should review the workshop report and preferably agree to the advice from the workshop and turn them into ICES recommendations.

Participants:

Participation should be by invitation of the experts within each field. The number of participants still needs to be determined. It would be advisable that a representative of the WGMME participates in the workshop to connect the outcome to the working group.

Funding:

A workshop budget is needed to cover the travel and per diem costs of the participants.

Organisation:

A steering committee should be put into place to plan the workshop and develop a draft agenda.

8 Cooperative Research Report

Term of Reference f) develop a Cooperative Research Report on threats to marine mammal populations based on a compilation of prior reports of this and former marine mammal working/study groups;

8.1 Introduction

Prior reports (1991-2004) of the ICES Study/Working Group on Marine Mammals and Seals in European Seals (SGSEAL/WGSEAL), Working Group on Marine Mammal Population Dynamics (WGMMPD), Working Group on Marine Mammal Habitats (WGMMH), Ad Hoc Group on the Impact of Sonar on Cetacean and Fish (AGISC) and Working Group on Marine Mammal Ecology (WGMME) were examined to develop a historical report on AG/SG/WG reviews of anthropogenic threats to marine mammals in European Seas. The major threats addressed by the various marine mammal groups were: 1) fishery by catch, 2) environmental contaminants, and 3) deliberate removals. Secondarily, the groups addressed indirect, or ecological, impacts of commercial fisheries on marine mammals. Discussions on the latter item, however, were focused on theoretical considerations due to the lack of empirical studies. Following are brief summaries on these items.

8.2 Fishery interactions

Fishery by catch was a major theme in nearly all prior SG/WG meetings, since this problem was both wide ranging (spatial, temporal, and fishing gear) and a direct source of human caused mortality. A review of prior reports revealed that early SG/WG evaluations of by catch were hampered by data gaps (i.e., lack of national reporting and monitoring programs), therefore relied on anecdotal information (i.e., fishermen reports, strandings) to develop findings and recommendations (ICES 1996). Over time, implementation and improvements in national reporting, monitoring (i.e., independent observers), and marine mammal abundance programs enabled the WGs to develop more comprehensive recommendations (ICES 2001, 2002, 2003).

8.3 Removal programs

Direct removal programs were intermittently reviewed by prior marine mammal SG/WGs (ICES 2000, 2001, 2003, 2004). The SG/WG reviews have generally focused on national seal removal programs as opposed to cetacean programs (i.e., Faroe Island drive fishery for long-finned pilot whales (*Globicephala melas*), and Norwegian minke whale (*Balaenoptera acutorostrata*) fishery. Atlantic white-sided dolphins (*Lagenorhynchus acutus*) and some other small cetaceans are also taken. The latter programs are normally reviewed by the International Whaling Commission (IWC). Whereas, seal removal programmes are closely linked to direct and ecological fishery interactions, which are high priority issues within ICES.

Removal of seals is conducted on different scales, ranging from the removal of individual seals to protect fishing or aquaculture facilities, to the reduction of a population of seals. The seal species removed are mostly Grey seal and Harbour seal. The 2004 WG considered the following definitions in its review of North Atlantic harbour seal and grey seal removal programmes:

1. A **seal removal programme** is a management programme with the aim (explicit or implicit) to reduce a population of seals or to remove individual seals that are of management concern.

2. A **population reduction programme** is one in which the objective to remove seals occurs over and above a harvest at replacement yield (consumption, hunt, other uses). In this case, the important question for managers is to assess biological effects on key prey species.

3. A **protection removal programme** is one in which individual seals are killed in order to protect fishing or aquaculture facilities.

Generally, SG/WG reviews have noted that monitoring programmes were not adequate to either assess the direct impacts of seal removal programmes on seal populations or to assess the ecosystem-wide effects.

8.4 Environmental contaminants

Because of their high trophic level marine mammals are especially susceptible to chemical contaminants aggregating in the food web, such as halogenated hydrocarbons and trace elements. From 1995 to 2004 several ICES working groups have addressed the effect of contaminants on marine mammal populations. These effects can be very different, ranging from immune suppression to negative effects on reproduction and early development. Within the working groups data gaps were identified and it was recommended that research that describing cause-effect relationships between environmental contaminants and population-level effects in marine mammals is carried out.

In addition to chemical contaminants the working groups addressed the potential problem of acoustic disturbance on marine mammal populations. Marine mammals use sound for communication, navigation and foraging; any acoustic disturbance in the sound production or receptivity processes may have variable extents of influence at an individual or population level. Recent concerns about the use of sonar and its potential impact on marine mammals has led to the forming of an Ad-Hoc working group on the Impact of Sonar on Cetacean and Fish. This group met in 2005 to review and evaluate relevant information concerning the impact of sonar on cetaceans and to identify any gaps in our current understanding.

Information compiled from the prior reports will be synthesized into a Draft Cooperative Research Report and distributed to WGMME members for review. We envision that a Draft suitable for review by the ICES Secretariat will be available by late autumn 2005.

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9 Future activities of the Working Group on Marine Mammal Ecology

It is likely that the demand for advice from ICES client commissions and others on marine mammal issues will continue and will grow in future years. This WG should continue to be parented by the ICES Advisory Committee on Ecosystems.

10 Recommendations

The Working Group on Marine Mammal Ecology [WGMME] (Chair: To Be Determined) will meet from 30 January to 2 February 2006 at ICES Headquarters, Copenhagen, Denmark.

WGMME recommended that activities for the 2006 meetings include:

a) continue preparations for a future Workshop (associated with WGMME meeting) on habitat quality and health aspects in marine mammals;

b) …..+++

11 Other business

WGMME recommends Meike Scheidat, Germany to become the new Chair, following the 2005 ASC. She will replace Gordon T. Waring (2003-2005 Chair).

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