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Report of the Working Group on Marine Mammal Ecology

22–25 March 2004
Pasajes, Spain

International Council for the Exploration of the Sea

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1 INTRODUCTION

1.1 Participation

The Working Group on Marine Mammal Ecology (WGMME) met at the AZTI Fisheries and Food Technological Institute, Pasajes, Spain, from 22–25 March 2004. Attendance at the meeting comprised:

Luis Arregi	Spain
Peter Corkeron	Norway
Arne Bjørge	Norway
Raul Castro	Spain
Iwona Kuklik	Poland
Finn Larsen	Denmark
Sven-Gunnar Lunneryd	Sweden
Mette Mauritzen	Norway
Meike Scheidat	Germany
Mark Tasker	UK
Santiago Lens	Spain
Tero Sipilä	Finland
Ainhize Uriarte	Spain
Gordon T. Waring (Chair)	USA

See Annex I for addresses.

The Working Group members were welcomed by Raul Castro and Lorenzo Motos, Head of AZTIMAR (Marine Research Area). The WG reviewed the Terms of Reference and a work schedule was adopted.

1.2 Terms of Reference

The Working Group on Marine Mammal Ecology (WGMME) meet will meet in Pasajes, Spain from 22–25 March 2004 to:

a) review of the usefulness of marine protected areas in marine mammal management,

b) review the scientific and management basis for seal removal programs in the North Atlantic, including:

   i) are monitoring programs adequate to access the direct impacts on seal populations;

   ii) are the monitoring programs adequate to assess the biological effects on key competitors of seals;

c) review the influence of the epizootic on seal populations in the North Sea;

d) for EcoQ element (c) Seal population trends in the North Sea, EcoQ element (d) Utilization of seal breeding sites in the North Sea, and EcoQ element (e) By-catch of harbour porpoises: reconsider the formulation of the EcoQO, determine whether a more specific EcoQO is needed in terms of its specification to the metric, time and geographical area, and as necessary propose more specific EcoQO(s) [OSPAR 2004/1]. In considering elements c) and d) take into account the effects of the epizootic;

e) provide the Study Group on Multispecies Assessments in the North Sea with data on the consumption of different prey by marine mammals in the North Sea, in a format specified by the Study Group;

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.

The WGMME will report by 31 March 2004 for the attention of ACE, as well as the Marine Habitat and the Living Resources Committees.

1.3 Justification of Terms of Reference

a) There is worldwide interest in establishing marine protected areas for marine mammals, including ICES waters. A literature review will be useful to evaluate their impact on marine mammal populations and human activities.

b) There is a need to understand the population and ecosystem impacts of seal removal programmes as a key input to considerations of societal benefits. An implicit paradigm guiding these programmes is "fewer seals will result in more fish production," but research to examine this hypothesis still needs to be developed.

c) The epizootic may have significant effects on the population dynamics of seals in the North Sea.

d) This is work in response to an OSPAR request.

e) SGMSNS require a compilation of data on quantities of foods consumed by marine mammals in the North Sea for input to an MSVPA model. These data will be compiled in the format required by SGMSNS.

f) This is required as the working groups input to the thematic writing panels working under the coordination of REGNS to develop an integrated assessment of the North Sea. For the purposes of this study the North Sea comprises ICES Area IV and IIIa and does not include intertidal areas. As far as possible, significant seasonal variation should be described.

The WGMME will report by 31 March 2004 for the attention of ACE, as well as the Marine Habitat and the Living Resources Committees.

1.4 Acknowledgements

WGMME thanks Raul Castro, Project Manager Researcher, Lorenzo Motos, Head of AZTIMAR (Marine Research Area), AZTI Fisheries and Food Technological Institute for their excellent hospitality and support to the meeting. We also thank Ian Boyd, Callan Duck, Ailsa Hall and Cecile Vincent (UK) for providing SMRU reports and data on the status of grey seals and harbour seal in the North Sea.

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 REVIEW OF MARINE PROTECTED AREAS FOR MARINE MAMMALS

Term of reference a*) review of the usefulness of marine protected areas in marine mammal management*

2.1 Definitions

WGMME discussed the usefulness of Marine Protected Areas (MPAs) as tools to manage human activities that affect marine mammals ("marine mammal management"). WGMME accepted the recent definitions of the US National Center for Ecological Analysis and Synthesis's working group on marine reserves (Lubchenco *et al.*, 2003). **MPAs** are ''*areas of the ocean designated to enhance conservation of marine resources*'', and fully protected **marine reserves** as ''*areas of the ocean completely protected from all extractive and destructive activities*''. "No-go areas", where all human entry is banned subject to permit, are a stricter categorisation than marine reserves, but are extremely rare and were not considered, except for the Moffen Nature Reserve off Svalbard, Norway (see below). As part of the management approach associated with MPAs, a commonly used technique for regulating disparate human uses is spatial zoning, where different areas exhibit different levels of protection from human intrusion.

Until the advent of MPAs, managing human activities in the marine environment tended to be sectorally based, often with poor communication between managers of different industry sectors (e.g., fishing, shipping, and tourism). Spatial zoning of MPAs allows coordination of the multiple users of marine systems, and operates in a manner similar to town planning (Day, 2002). There are areas where an MPA has been declared, but no management actions specified, so that peoples' activities allowed within the MPA are no different from activities outside the MPA. In these instances, MPAs allow little (if anything) more than the opportunity to integrate coastal management.

2.2 Issues

2.2.1 Marine Reserves

There are very few examples of marine reserves that have obvious direct bearing on marine mammal management. Fisheries can influence the behaviour and habitat use of cetaceans in unforeseen ways (e.g., Chilvers and Corkeron, 2001; Chilvers *et al.,* 2003), so the use of marine reserves as a tool in marine mammal management clearly requires further investigation. None of the MPAs listed in Tables 1 and 2 of Reeves (2000) or in Hooker and Gerber (2004, Table 1) are marine reserves (as defined above) where *all* extractive and destructive activities are banned. Moffen Nature Reserve, lies within the Northwest Spitzbergen National Park, Norway, and is a small ($16km^2$) no-go area established to protect an important walrus haulout site from human intrusion during the summer (Reeves, 2000). In 1983 the area was declared a sanctuary, and that people cannot approach to within 300m of the shoreline in the area.

2.2.2 Zoned MPAs

Some MPAs have been zoned in a manner to enhance marine mammal management. Examples (both from Australia) include the Hervey Bay Marine Park established for managing whale watching on humpback whales, *Megaptera novaeangliae*; and the Dugong Protection Areas (DPAs) in the Great Barrier Reef Marine Park (GBRMP), off Queensland, established to reduce incidental anthropogenic mortality on dugong, *Dugong dugon*. The evidence available suggests that the Hervey Bay MPA has worked well as a tool for marine mammal management, within the restricted context of managing whale watching (which is the reason for which it was established). The Hervey Bay MPA's existence made establishing a Dugong Protection Area there easier than it would otherwise have been. The success of DPAs as a tool for conserving dugongs remains to be determined. They have been in place for eight years. The aerial survey design in place at present is unlikely to have sufficient power to assess this in the short or medium term.

2.2.3 Unzoned MPAs

There are examples of MPAs that have no zoning plans associated with them, but have been established with an aim to enhance marine mammal management. Examples include the Banks Peninsula Marine Mammal Sanctuary for Hector's dolphins *Cephalorhynchus hectori*; (New Zealand); the Moray Firth candidate Special Area of Conservation (cSAC, United Kingdom) for bottlenose dolphins; and the Froan nature reserve for harbour seals in Norway. Factors that limit the capacity to assess the success of the Banks Peninsula Sanctuary include the small size of the dolphin population in the region; and that observer coverage on the recreational fishery within and immediately adjacent to the Sanctuary is non-existent. The evidence available suggests that the Sanctuary is succeeding in its aim of reversing the decline in the local population of Hector's dolphins (Burkhart and Slooten, 2003). This is possibly the most successful example of an MPA being used in marine mammal management. The draft Conservation Objectives relating to bottlenose dolphins in the Moray Firth cSAC appear oriented towards ecosystem-based management to ensure conservation through maintenance of habitat (The Moray Firth Partnership 2003). No activities appear to be explicitly banned by the current management scheme. It appears that the cSAC offers the opportunity for integrating coastal activities by engaging all sectors in management. The success of the current management scheme has not yet been examined.

Drowning in gillnets is a substantial source of mortality of weaned harbour seal pups in Norway, but once over one year of age, harbour seals are far less likely to be entangled in gillnets. Tagging studies indicate that the size of the Froan MPA includes the ranges of weaned pups born in the archipelago. This is because around Froan Marine Protected Area is the only area on the Norwegian coast where recently weaned pups do not suffer mortality from gillnet entanglement (Bjørge *et al*., 2002).

2.3 Emergent issues on MPAs in marine mammal management.

Scientific understanding of human effects on marine ecosystems is improving, and our impacts are far more pervasive and destructive than previously thought (e.g., for fisheries, Jackson *et al*., 2001; Pauly *et al*., 2002). MPA zoning is similar to the manner in which we regulate the spatial distribution of human activities on land (Day, 2002), so designation of an MPA is the start, not the end, of a management process (Reeves, 2000). Unzoned MPAs that do not include the capacity to ensure changes to the manner in which people's activities affect the environment inside the MPA can do little more than offer the capacity to integrate coastal management.

2.3.1 The usefulness of MPAs

1. Bringing a group of managers into marine environmental management whose primary focus is not just one industry sector, and who are more likely to be interested in maintaining the integrity of marine ecosystems than other managers.
2. Making it more likely that new industrial developments will have their impacts assessed, in the context of all anthropogenic influences on the site in question.
3. Raising public awareness about an area, giving an area a more coherent identity in the minds of the public, and raising the concept of the intrinsic value of marine ecosystems and their components.
4. Encouraging interaction between all stakeholders.
5. Integrating coastal management. For some MPAs (e.g., GBRMP, Moreton Bay Marine Park), some of the purportedly intractable problems with MPAs (Jameson *et al.,* 2002), for example, terrestrial runoff from coastal watersheds, are being addressed. Nothing else has been demonstrated to work as well as MPAs intregrating the management of all sectoral interests.
6. Once an MPA is declared, if it becomes clear that current management in place is insufficient to achieve a stated management goal, it can be easier to change zoning to revise management approaches than to establish a management regime outside of an MPA.
7. MPAs may be more likely to have comprehensive monitoring programmes instituted than areas without MPAs.

2.3.2 Problems with MPAs

1. "Paper parks" problems. A "paper park" is a protected area that is declared but where the park offers no extra protection than that available to animals outside the park. These issues remain areas of debate for terrestrial parks as well as MPAs, and whether a paper park is better than no park at all is unresolved. Issues include:

    a. If activities in a "protected" area are the same as those permitted outside the "protected" area, what is the protection?

    b. If enforcement is weak, what is the guarantee that illegal activities will not take place (see Gribble and Robertson (1998) for an example of this regarding fishing in the GBRMP)?

    c. Is the park achieving anything more than allowing reduction in political pressure for some form of management (e.g., endangered species conservation).

2. Time to establishing an effective management regime for an MPA. Sometimes, this is excessively long, especially when international agreements are required. For instance, the Ligurian Sea Cetacean Sanctuary (an agreement between the governments of France, Italy, and Monaco) was initially proposed in 1991 and took until 2002 to be ratified, and a management plan for the Sanctuary has been completed only recently (G. Notarbartolo di Sciara, pers. comm., January 2004).

3. Jurisdictional issues. Within the European Union, it may be possible for a country to prevent its nationals from engaging in an activity (e.g., fishing with gillnets) within an MPA, but have no legal recourse to preventing nationals of other countries from engaging in the activity in the MPA.

4. "Pocket handkerchief" MPAs, that is, parks that have restricted spatial coverage, and so cannot achieve their stated aims. Although small marine reserves can be valuable (e.g., Gell and Roberts 2003), small MPAs, especially unzoned MPAs, are less likely to achieve a conservation benefit. For example, the mean area of the cSACs in the UK that have been established at harbour seal haul-out sites (Duck, 2003) is in the order of 150 km^2. These small MPAs may not achieve protection for seals if local seals change their spatial distribution over relatively small distances. This may have happened in one instance in Scotland (Thompson *et al*., 2001).

5. Potentially high data costs. Experience within the ICES area has shown that finding appropriate boundaries for MPAs, particularly when full spatial and temporal variability in animal distribution needs to be accounted for, can be very costly in research resources. Temporal variability may be on a decadal scale. Inadequate data may lead to inappropriate boundaries, with potential disadvantages to the populations being "protected". On the positive side, data collected for MPA boundary selection often has other benefits.

2.4 Conclusions

There are cases where MPAs have clearly been successfully used in marine mammal management. The monitoring requirements that tend to be instituted with MPAs have proved to be particularly useful. However, there are also examples where declaring an MPA may serve a political or management purpose, but where restrictions on human activities in the MPA are either extremely limited or non-existent. Also, if MPAs as management tools are to be held to rigorous scrutiny, then other management tools should be subject to the same level of scrutiny (Gell and Roberts, 2003).

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3 POPULATION AND ECOSYSTEM IMPACTS OF SEAL REMOVAL PROGRAMES

Term of reference b) review the scientific and management basis for seal removal programs in the North Atlantic, including:

i) are monitoring programmes adequate to access the direct impacts on seal populations;

ii) are the monitoring programmes adequate to assess the biological effects on key competitors of seals

The working group 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.

We did not discuss harp and hooded seals, as they are covered by the joint ICES/NAFO WGHARP.

An issue with assessing seal removal programmes is defining what is the appropriate management unit for consideration (i.e., what is a “population”). This has been, and remains, one of the complex issues in all aspects of applied ecology (e.g., Crandall *et al.,* 2000). Should an individual breeding aggregation (that may not be genetically distinct from other breeding aggregations, but may be of importance in local ecosystems or whose existence may be important to local human communities) be considered a population? This is discussed further, in the context of EcoQOs, in Section 5 of this report. Populations may also be migratory or resident.

We assessed each country’s seal removal programmes with regard to the points above. Only countries whose Atlantic waters include the known range of grey and harbour seals are included here.

3.1 The Baltic

Details of the status of marine mammals in Baltic States are available in the WGMME report of 2003. Most states do not have seal reduction programmes, but for the states that do, monitoring programmes are not adequate to either assess the direct impacts on seal populations or to assess the ecosystem-wide effects. A protection removal programme and research are monitoring the reduction in fishery damage caused by the protection hunt.

3.2 Belgium

As far as WGMME are aware, there are no seal removal programmes in Belgium.

3.3 Canada

Harbour seals in Canada are protected from hunting. First Nations Canadians are allowed to hunt them for subsistence in northern Canada, and some permits have been issued to remove nuisance seals from around aquaculture facilities (i.e., there is a small protection removal programme in place). Grey seals in Canada are not hunted, other than by people issued with “personal use” licences that entitle the licensee, once suitably qualified, to take up to five grey seals per year for personal use. Very few of these licences are issued. Culling programmes ended in the early 1980s, and the last bounties were paid on grey seals in the early 1990s (all information M. Hammill, pers. comm., DFO, Quebec, Canada, March 2004). WGMME are unaware of any data assessing the efficacy of current or historical programmes.

There is a proposed programme to experimentally assess the ecological efficacy of a seal removal programme in Canadian waters, but this has yet to be established, and deals with harp seals and so is not considered further.

3.4 Denmark

There is a small protection removal programme for harbour seals in the inner Danish waters and the Kattegat regulated by the number of licences issued. There are a few issued annually in Denmark. WGMME are unaware of any programme to assess the biological effects and direct impacts of this programme.

3.5 Faroe Islands, Denmark

Hunting by humans has extirpated harbour seals in the Faroe Islands. The status of grey seals in Faroese waters was reviewed recently (NAMMCO, 2003). There appears to be a small breeding population of grey seals in Faroese waters whose size is not known precisely. Grey seals breeding in the UK are known to use Faroese waters. Currently, a protection removal programme is in place and approximately 200–250 seals in the vicinity of aquaculture facilities are shot each year. There was also a scientific take of grey seals to investigate their stomach contents, between 1993 and 1995 (Mikkelsen *et al.,* 2003).

The Faroese grey seal population is subject to an apparently high but unknown level of exploitation. This exploitation has developed since the recent advent of fish farming activities. The abundance of breeding and migrant seals in the area is unknown. However, the number of seals breeding in the Faroes is unlikely to be large because breeding habitat is limited. Therefore, even if the human take includes a large proportion of migrant animals, the local population might still be subject to depletion.

At present there is no programme to estimate the abundance of Faroese grey seals, nor to assess what proportion of shot animals are from British breeding colonies. The lack of any monitoring programme means that there are no data available to assess the direct impacts on seal populations of the current levels of take. WGMME are unaware of any programme to assess the biological effects and direct impacts of this programme.

3.6 France

As far as WGMME are aware, there are no seal removal programmes in place in French metropolitan or overseas territorial waters.

3.7 Germany

As far as WGMME are aware, there are no seal removal programmes in place in Germany.

3.8 Greenland, Denmark

There are a few harbour seals in Greenland. As far as WGMME are aware, there are no seal removal programmes in place in Greenland.

3.9 Iceland

The Icelandic grey seal population numbers approximately 5000 animals; it is currently in severe decline and was reviewed recently (NAMMCO, 2003). It has been declining at over 6% per year for longer than a decade. As direct mortalities have been above replacement levels for many years and the population is clearly in decline, there appears to be an implicit aim of grey seal management in Iceland to reduce the abundance of grey seals, so this is a population reduction programme. Some serious problems with technical aspects of the methods used to estimate the abundance of grey seals in Iceland were identified in NAMMCO (2003), and changes to the monitoring programme are currently being introduced. The monitoring programme in place has detected a decline in grey seal abundance along the entire Icelandic coast, but spatial detail is lacking. No monitoring programme has been established to assess the biological effects on key competitors of seals.

3.10 Ireland

No data were available on seal removal programmes in the Republic of Ireland.

3.11 The Netherlands

As far as WGMME are aware, there are no seal removal programmes in place in the Netherlands. Grey seal populations in the waters off the Netherlands are increasing (SCOS, 2003).

3.12 Norway

Since 1996, Norwegian management of grey and harbour seals has been based on a regulated game hunt. Up to 2002, regulations included closed seasons, and quotas based on seal abundance (5% of point estimates of abundance), with provision for small quota increases in areas of existing fisheries conflicts. The implicit aim of management was to maintain population sizes, except in areas with perceived fishery conflicts. Figure 3.1 shows the relationship between scientific advice given and administrative quotas set for 1997–2003. However, quotas for grey seals were generally not reached: 11-–35% of the quota was taken 1997–2002. Harbour seal quotas have generally been reached.

In 2003, quotas for grey seals were at 25% of the current estimated abundance, and a bounty is being paid. The grey seal quota was not reached in 2003. Quotas for harbour seals were set at 13% of the last (pre-epizootic) estimate of abundance. The size of the unlicensed hunts is unknown.

Were quotas to be reached, then the size of the population change relative to the best population estimate available for either species in Norwegian waters should trigger action under the EcoQO “Trends in harbour seal populations,” Section 5.2, as Norway is signatory to the Bergen Declaration.

Weaned grey seal pups were harvested prior to the introduction of a closed season in 1973. Apparent increases in population sizes of grey seals along the Norwegian coast could be due to reductions in hunting pressure. Although, immigration from increasing populations in the British Isles to the southwest and Russia to the north and east, cannot be excluded (Haug *et al.,* 1994).

Surveys of grey seals have been ongoing in some parts of Norway since at least 1974 (Wiig, 1986), but population sizes and trends in most areas remain unclear. Population estimates with associated confidence intervals are available for grey seals in Froan MPA in 1993 (Bakke and Lorentsen, 1999) and recently for most of the known grey seal breeding localities (Nilssen *et al.,* 2003). The techniques used to estimate the abundance of grey seals in Norwegian waters require improvements to reduce bias and increase their precision (Corkeron *et al.,* 2003).

Harbour seal abundance along the Norwegian coast is now estimated from one photographic survey of known haul-out sites during the moult. The most recent survey was conducted in 2003 and photographs are still being analysed. Recent radiotracking work seeks to estimate the proportion of animals available for photography at the instance of the survey. The current survey programme includes no consideration of moulting phenology. The techniques used to estimate the abundance of harbour seals in Norway require improvements to reduce bias and improve precision.

No formal analysis of the effect of this level of harvest on either seal species, including the risk of extinction, nor of the sensitivity of the survey programme to detect a population decline, has been conducted. The information available indicates that the monitoring programme in place currently is not adequate to assess the impact of the removal programme or biological effects on either species of seal.

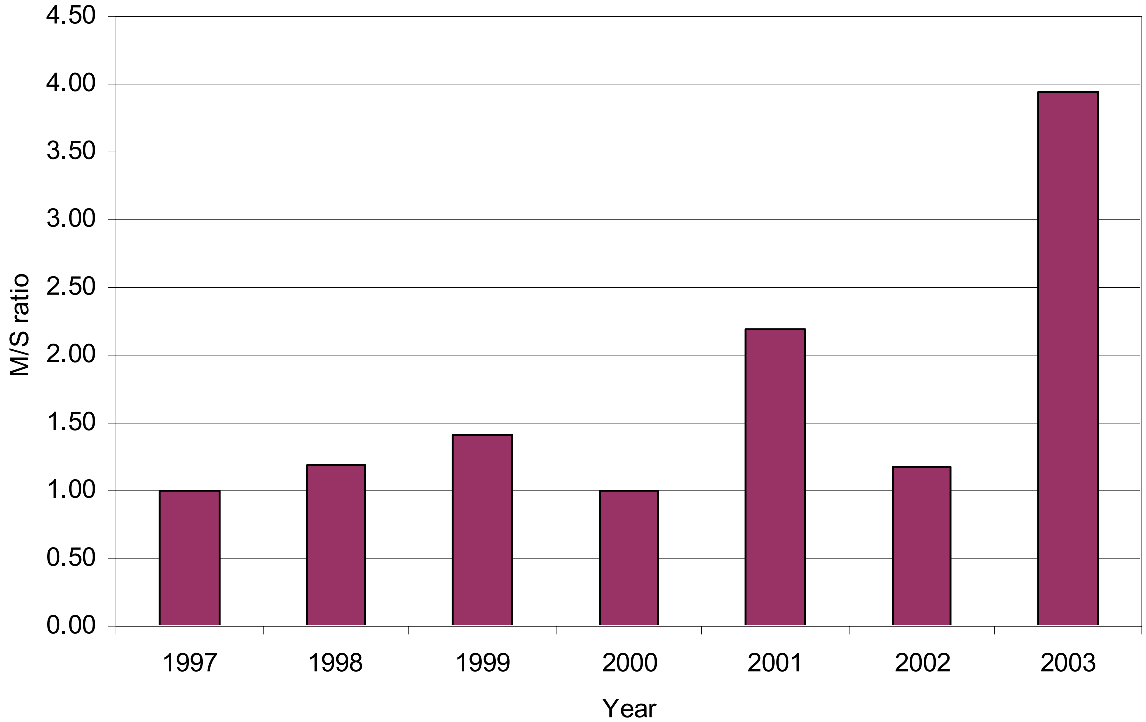

Figure 3.1. The relationship between scientific advice on quotas for grey seals in Norwegian coastal waters, 1997–2003. M/S ratio is the ratio of quotas issued by management (M) divided by the quotas issued by scientists (S). When management-issued quotas equal scientific advice, the M/S ratio is 1. Values over 1 indicate management-issued quotas that are larger than scientific advice.

3.13 Russia

As far as WGMME are aware, there are no seal removal programmes in place in Russia.

3.14 Sweden

There is a small protection removal programme for harbour seals in the Kattegat-Skagerrak regulated by the number of licences issued. These are approximately ten in Sweden. WGMME are unaware of any programme to assess the biological effects and direct impacts of this programme

3.15 United Kingdom

Grey seals in the UK are surveyed annually, harbour seals over a five-year period but with more frequent surveys in specific regions. There is no population reduction programme in place, but there is a protection removal programme in place. There is a lack of information on the size of the take of either seal species in UK waters. However, monitoring programmes, backed up by model-based estimation of mortality, may be adequate to estimate the overall impact of removal programmes on the species in UK waters (SCOS, 2003), although the statistical power to make such assessments is greater for grey seals than for harbour seals. Only in some regions are harbours seal population data adequate to make these assessments

There is a substantial effort under way to assess and model the role of seals in UK marine ecosystems (SCOS, 2003). During the coming year, SMRU will use the mathematical model described in SCOS-BP 03/3 to investigate the effects of different levels of shooting of seals outside the closed season on the dynamics of the British grey seal population. There are no data about the number of animals being killed, but preliminary calculations indicate that the observed reduction in the growth of the population could be explained by the killing of 4,000–8,000 juvenile or adult grey seals each year since the mid-1990s. If these deaths were the result of the deliberate killing and/or by-catch, then the current size of the population is likely to be higher than estimated in SCOS-BP 03/3. Obtaining data about the number of seals being killed will reduce the uncertainty surrounding current estimates of the total population size.

3.16 United States of America

There are no seal removal programmes in place along the Atlantic coast of the USA. Prior to the reauthorization of the U.S. Marine Mammal Protection Act in 1991, staff at aquaculture facilities were permitted to shoot "problem" seals. Seals are still shot occasionally (presumably by fishermen), but the full extent of this is unclear (Waring *et al.*, 2002).

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4 THE INFLUENCE OF THE 2002 SEAL EPIZOOTIC ON NORTH SEA SEAL POPULATIONS

Term of reference c) *review the influence of the epizootic on seal populations in the North Sea*

This review builds primarily on the following published information: Harding *et al.* (2002); Lonergan and Harwood (2003); Reijnders *et al.* (2003); Reineking (2002).

Chronology and spreading pattern of the 2002 PDV harbour seal epizootic

The 2002 PDV epizootic amongst harbour seals started on Anholt in the Kattegat in April–May, and the first unusual mortality was reported on 4 May 2002. The epizootic spread in summer northwards and leaped to the western part of the Wadden Sea in mid-June, from where it spread eastwards throughout the Wadden Sea (Figure 4.1). The first victims in the UK were found in mid-August in the Wash (SMRU, 2003). Details of the chronology and spatial and temporal spreading are provided by Reineking (2002). In the North Sea and Baltic Sea together at least 22,500 seals were found dead (Reineking, 2002).

The disease spread rather quickly from the Danish Kattegat to the Skagerrak in the north. Within about a month, seal deaths were reported from nearly all sites in the Kattegat/Skagerrak area and the Oslofjord. The first case of mortality with confirmed PDV in the Wadden Sea occurred in the western part of the Dutch Wadden Sea far from the outbreak progressing gradually from the Kattegat (Reineking, 2002). Such an isolated case contradicts the anticipated pattern of spreading based on the assumption of transmission from animal to animal when hauled out (Kennedy, 1990), and the current knowledge of dispersal of Wadden Sea harbour seals (e.g., Nørgaard, 1996). However, satellite tracking has demonstrated that single individuals can wander widely. Reijnders *et al.* (2003) indicated the possibility of another marine mammal or even an anthropogenic carrier of the virus to the western Wadden Sea.

The population in the Limfjord was affected only from 16 September 2002. This indicates that this seal stock has little exchange with the Kattegat-Skagerrak colonies, at least not in the summer.

The subsequent spread of the disease after the Wadden Sea to the Wash and later on to Scotland, Wales, Northern Ireland, and the Republic of Ireland, as well as from the Wadden Sea to the Delta area (SW-Netherlands) and further on to the Belgian and French coastal waters, was rather similar to that observed during the 1988 epizootic (Dietz *et al.*, 1989).

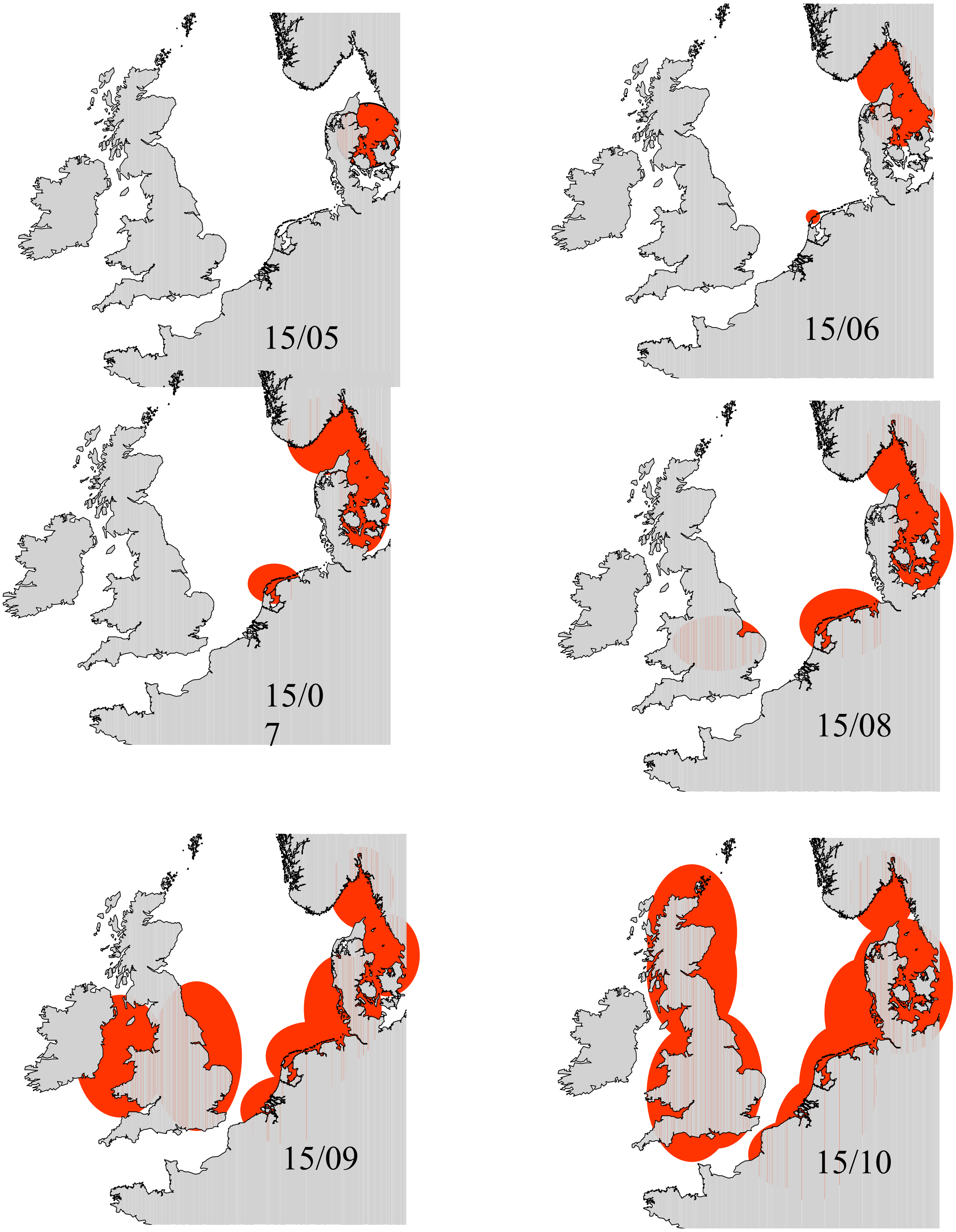

Figure 4.1. The geographical spreading of the 2002 PDV epizootic among North Sear harbour seals from the outbreak at Anholt, Danish Kattegat, in early May to the culmination of the disease in October 2002. Figure from Reineking (2002).

4.1 Short-term effects

The total number of dead seals in Table 4.1 includes 33 dead seals in areas where there are no documented seal colonies. In addition to the total of 22,336 dead seals, there were 161 dead seals reported from Ireland.

It is obvious that the proportions of dead seals found and reported were influenced by the environmental conditions, topography, and the effectiveness of the recording and reporting system. The influence of wind direction and force was demonstrated by, e.g., an unlikely drop in animals found dead in the Netherlands around September 1st, followed by an increase in the third week of September.

Lower Saxony probably received dead seals from the Netherlands, Schleswig-Holstein, and Denmark. Animals found dead after a period of offshore winds were usually in a worse condition (longer time since death) than those found in earlier periods (M. Stede, cited in Reineking, 2002), indicating they had drifted some time at sea before arriving at the coast and being collected. Due to the mainly northwards direction of currents in the Kattegat and in the eastern Skagerrak, some of the dead seals reported from the Oslofjord area may have originated from Swedish or Danish waters.

These ambient environmental factors may contribute to the differences in mortality observed, e.g., within the Wadden Sea and within the Skagerrak-Kattegat areas. Different topography may contribute to the observed difference in mortality between the Wadden Sea and the Skagerrak-Kattegat.

It is also possible that the timing of the disease in different areas affected the mortality. Assuming that the virus is spread between hauled-out animals, the transfer of virus would be enhanced by the more extensive haul-out bouts during moult in August. The duration of the epizootic within an area seems to be about five weeks (Heide-Jørgensen *et al*., 1992). Therefore, the outbreak was over in the Kattegat-Skagerrak at the onset of the moult, while the outbreak was still ongoing in parts of the Wadden Sea during moulting season. In the Limfjord, Danish Baltic, and UK, the mortality peaked after the moulting season.

4.2 Medium-term effects

Reijnders *et al.* (2003) calculated population development for scenarios where the epizootic cycle length would be respectively two, seven, and fourteen years, and a scenario where no epizootic would occur. The modelling was based on the population parameters obtained over the past years since the last epizootic (Reijnders *et al.,* 1997; Reijnders and Brasseur, 2003; Reijnders *et al*., 2003) and it was assumed that the combination of parameters found in the period 1990–2002 were also valid for the period of the prognosis. The epidemiological modelling was based on the method used by Grenfell *et al.* (1992) and Heide-Jørgensen and Härkönen (1992). The results are shown in Figures 4.2a–d.

Table 4.1. Number of dead seals reported in North Sea harbour seal colonies with the best available pre-epizootic abundance estimates of the respective areas. The data are from Reineking (2002).

Area	Observed no. of dead seals	Pre-epizootic population estimate	Observed mortality %
Dutch Wadden Sea	2,244	3,600	62
Lower Saxony	3,851	6,220	62
Schleswig-Holstein	3,338	7,190	46
Danish Wadden Sea	962	2,380	40
Total, Wadden Sea	***10,656***	***Ca 20,000***	***53***
Helgoland	270	Ca 400	68
Danish Kattegat	2,049	3,250	63
Swedish Kattegat/Skagerrak	4,000	15,000	27
Norwegian Skagerrak/Oslofjord	878	1,200	73
Total Kattegat/Skagerrak/Oslofjord	***6,927***	***19,000***	***37***
Limfjord	365	886	41
Danish Baltic	95	270	35
United Kingdom	3,990	34,100	12
Total North Sea and adjacent areas	***22,336***	***74,496***	***30***

The cycle of two years was chosen because it was calculated that only after this point of time could a new epizootic theoretically happen. The period of fourteen years is a representation of the period between the last two epizootics, and seven years is the mid-value thereof.

Figure 4.2a shows a rapid recovery of the population to its pre-epizootic level of around 27,000 seals and a level of approximately 70,000 would be reached in 35 years. Under the two-year cycle (Figure 4.2b), the epizootic would finally damp out and the population will slightly decrease and amount to approximately 15,000 animals in 2038. The seven-year cycle (Figure 4.2c) would result in an overall slight increase and the fourteen-year cycle (Figure 4.2d) would result in a stronger overall increase.

The estimates are based on the estimated disease-free equilibrium of the population (see Grenfell *et al*., 1992), on the growth rates observed between 1990 and 2001 (Reijnders and Brasseur, 2003), and the assumptions that the Allee Effect applies.

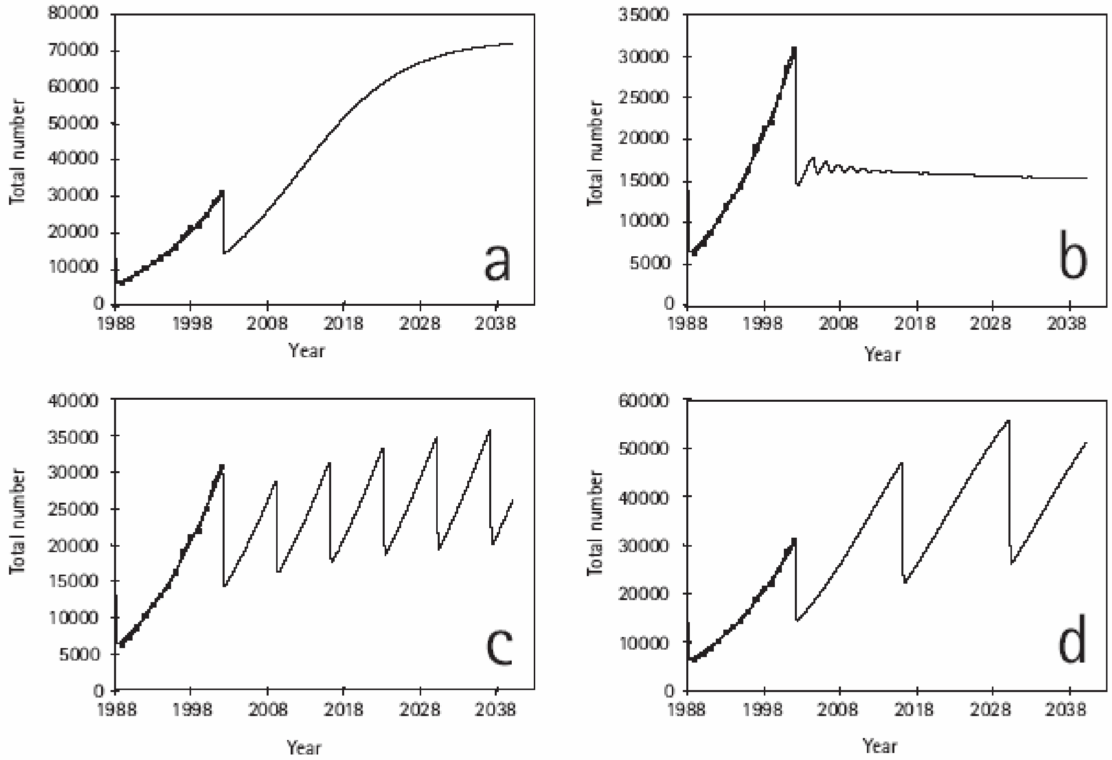

Figure 4.2. Estimated population trajectory of four scenarios, including no further epizootics (a), epizootics recurring every two years (b), every seven years (c), and every fourteen years (d). From Reinders *et al.* (2003).

Taking into account the scientific debate (Murray, 1994; Sinclair and Pech, 1994; Morris, 1996) on density dependence in, e.g., time and space, related to environmental stochasticity, compensatory processes, Reijnders *et al.* (2003) pointed out the need for further modelling in order to obtain the range of confidence intervals around the estimated carrying capacity, given the variance in the data used under the afore-mentioned assumptions.

Irrespective of the exact final population size reached after 35 years, it is obvious that under the assumed scenarios different net, long-term population growth rates will be achieved. Under all scenarios tested, the net growth rate would be considerably below the value reached if no epizootic would occur. If the interval of fourteen years between the last two epizootics is taken, the net population growth would be around half of what it would have been without a new epizootic.

4.3 Potential long-term effects

During the 2002 outbreak of PDV, Harding *et al.* (2002) predicted that this outbreak would cause an infection identical to that of the 1988 outbreak (which killed 58% of the population) because immunity was assumed to play no significant role in the dynamics of the current outbreak. Although it is generally believed that survivors of PDV develop life-long immunity (Kennedy, 1990), Harding *et al.* (2002) estimated that at most 7% of the current population are survivors of the 1988 epidemic, which would have a negligible impact on mortality. Because the timing of the outbreak is important in determining local mortality rates, they predicted higher mortality rates on the European continent than in Great Britain or Ireland. A stochastic model was used to quantify how recurrent epizootics affect the long-term growth, fluctuation, and persistence of the population.

Harding *et al.* (2002) calculated that at the 1988 mortality rate and the 1988–2002 recurrence interval, the PDV epizootic reduces the stochastic growth rate by half, from 0.12 to 0.06. It increases the risk of a 50% population decline ten-fold, from 0.06 to 0.61. The risk of crashes to 10% of the current population size (defined as quasi-extinction level) increases from negligible in the absence of epizootics to a serious risk of 0.18.

In a rebuttal to Harding *et al.* (2002), Lonergan and Harwood (2003) showed that incorporating the effects of observation error during population surveys and of the long-term immunity of survivors (Figure 4.3) resulted in a much lower level of risk (<1%) for quasi-extinction (reduction to less than 10% of pre-epizootic population levels) (Figure 4.4).

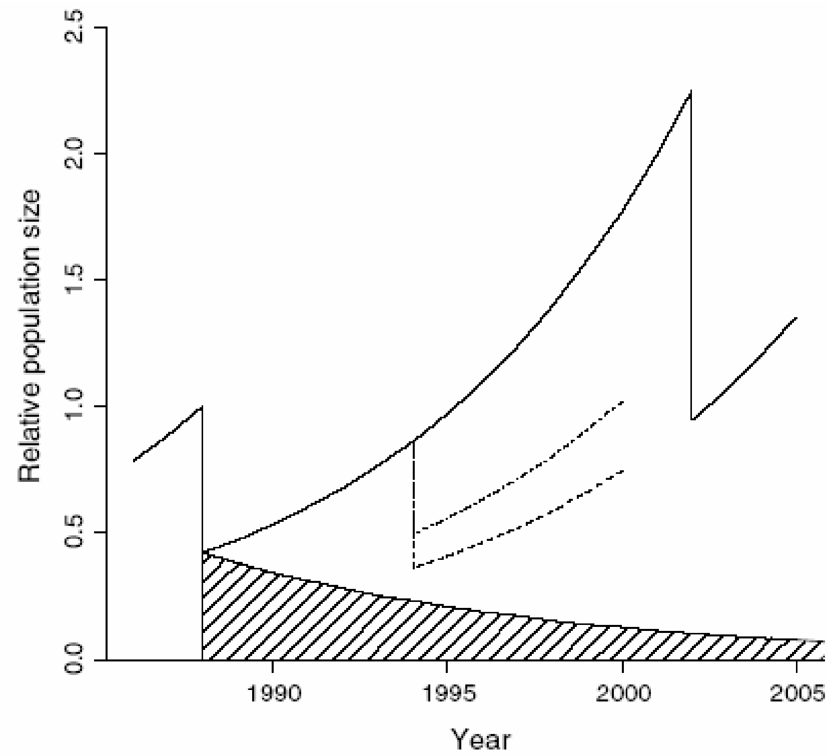

Figure 4.3. The effect of immunity on the size of recurrent epizootics. The solid line shows the effects of 58% mortality in 1988 and 2002. The shaded area indicates those animals that survived the first outbreak. These animals are assumed to be immune to the disease but die off from other causes at a rate of 10% per year. It can be seen that approximately 5% of the population in 2002 was immune to the disease. The broken lines show the consequences if a second outbreak had occurred after only six years when the presence of immune animals is ignored (dashed line) or accounted for (dotted and dashed line). From Lonergan and Harwood (2003).

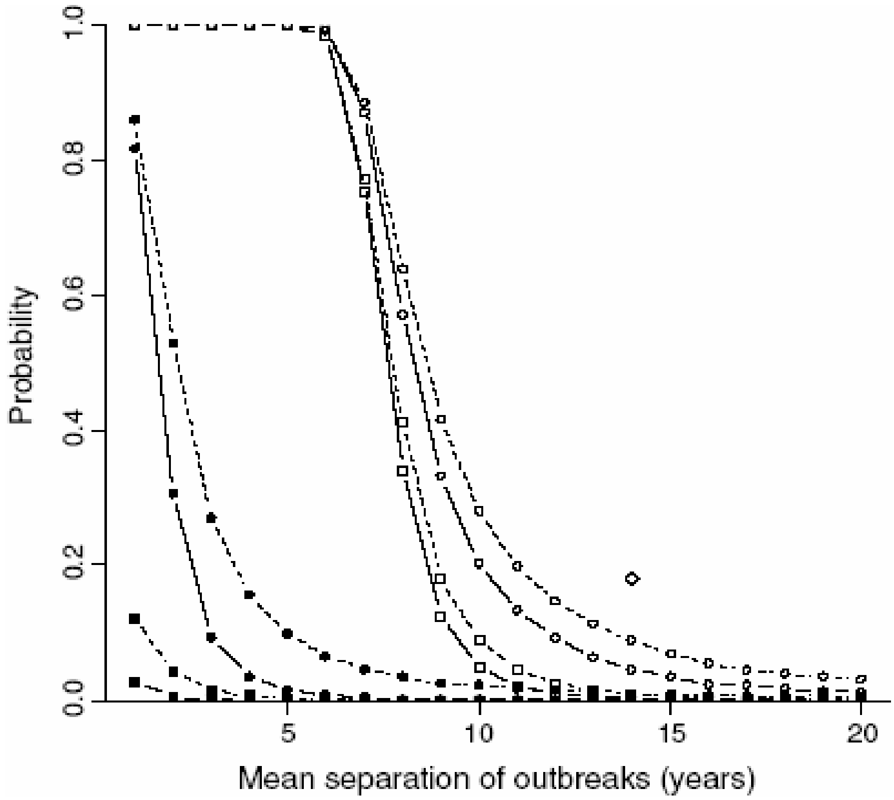

Figure 4.4. The probability of quasi-extinction for recurrent outbreaks of phocine distemper virus (PDV) at different frequencies. Disease-induced mortality is taken to be 58%, the long-term mean of the logged population growth rate is 0.12, its variance is either 0.03 (solid lines) or 0.06 (dashed lines). Natural (non-disease) mortality is taken to be 10% per year. Hollow symbols indicate probabilities that were calculated ignoring the effects of immunity and the filled ones those incorporating it. The circles are on the 90% probability lines and the squares on the 99% probability lines. Diamond symbols are the 18% risk of a 90% decline in the population for a fourteen-year mean inter-epidemic period suggested in Harding *et al.* (2003). From Lonergan and Harwood (2003).

Lonergan and Harwood (2003) demonstrated that, while the immediate effects of the disease are dramatic, it is unlikely that recurrent epidemics will pose serious conservation problems for this species under current conditions.

4.4 The effect of the 2002 epizootic on other species

A total of 881 grey seals were recorded dead in the North Sea region following the epizootic. The effect on the total population is regarded as insignificant (Reineking, 2002). Mortality in other species is not known.

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5 ECOQ FOR SEAL POPULATION TRENDS IN THE NORTH SEA

Section 5 Ecological Quality Objectives

Term of reference *d) for EcoQ element (c) Seal population trends in the North Sea, EcoQ element (d) Utilization of seal breeding sites in the North Sea, and EcoQ element (e) By-catch of harbour porpoises: reconsider the formulation of the EcoQO, determine whether a more specific EcoQO is needed in terms of its specification to the metric, time and geographical area, and as necessary propose more specific EcoQO(s) [OSPAR 2004/1]. In considering elements c) and d) take into account the effects of the epizootic*;

5.1 Introduction

The Fifth North Sea Conference in 2002 agreed that two Ecological Quality Elements relating to seals in the North Sea would be further developed. These elements were:

- 3 (c) Seal population trends in the North Sea;
- 3 (d) Utilisation of seal breeding sites in the North Sea.

An Ecological Quality Objective was agreed for the first of these elements:

“No decline in population size or pup production of ≥10 % over a period of up to 10 years”.

The Conference in 2002 also agreed that an Ecological Quality Element relating to cetaceans would be further developed:

- 3 (e) By-catch of harbour porpoises in the North Sea.

An Ecological Quality Objective was agreed for this Element:

“Annual by-catch levels should be reduced to levels below 1.7% of the best population estimate”.

Progress made in the development of these EcoQ Elements was reported to OSPAR’s Biodiversity Committee in early 2004 (OSPAR BDC 04/02/07 and 04/02/08). This Committee agreed a number of points about these Ecological Quality Elements that are reflected in the considerations below.

5.2 Seal population trends in the North Sea

There are several difficulties that have arisen with this EcoQO that might be resolved if it were better defined.

a) The Ecological Quality Objective relates to two seal species with differing biological characteristics: grey seals give birth in terrestrial habitats and are best counted as numbers of pups produced per year, while harbour seals give birth in intertidal habitats and are best counted as 1-year + seals during the period that they haul-out terrestrially to moult. The timing of counting, methods, and confidence in the population estimate differ between the two species. These differences lead WGMME to suggest that:

- **The two seal species have separate Ecological Quality Objectives.**

b) WGMME and ICES noted last year (ICES, 2003) that the EcoQO would be triggered rather often due to the inter-annual variation in numbers of seals (both pups counted or numbers on haul-outs). This level of "alarms" is felt by WGMME to be too high, and thus we suggest that a five-year running mean might be applied to these figures (see Figures 5.1–5.3). Such an approach would detect long-term changes in pup production or haul-out numbers for grey seals and harbour seals, respectively. The disadvantage of this is that mortality events, such as caused by epizootics, would not trigger the EcoQO. WGMME felt that this was not major disadvantage as large mortality events appear to already be investigated in depth (see Section 4), whereas more subtle long-term changes might easily be overlooked. If the level of "false positive" was felt to be too high with a five-year running mean, it might be possible to switch to a three-year running mean. WGMME suggests that:

- **The EcoQO(s) for seal population trends be expressed in terms of a five-year running mean.**

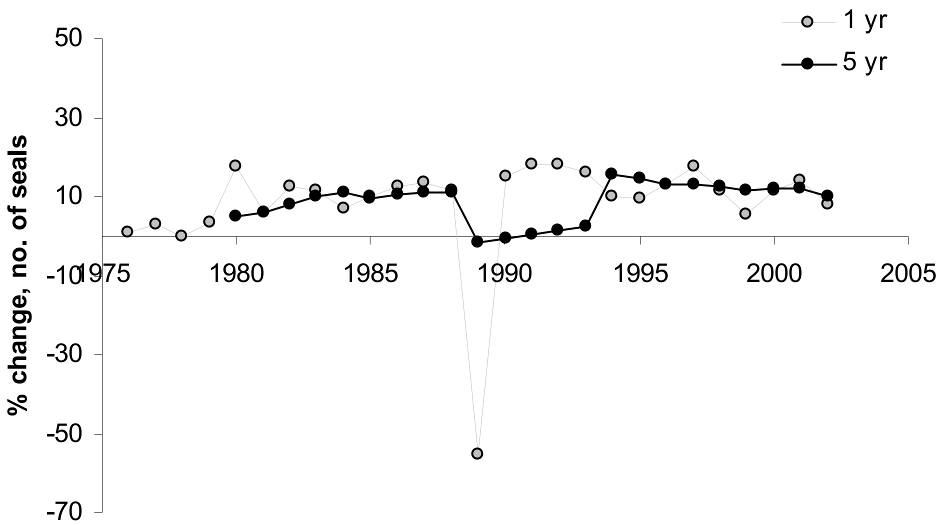

Figure 5.1. Annual and five-year running means of changes in harbour seal counts in Niedersachsen and Schleswig Holstein (M. Scheidat. pers. comm.).

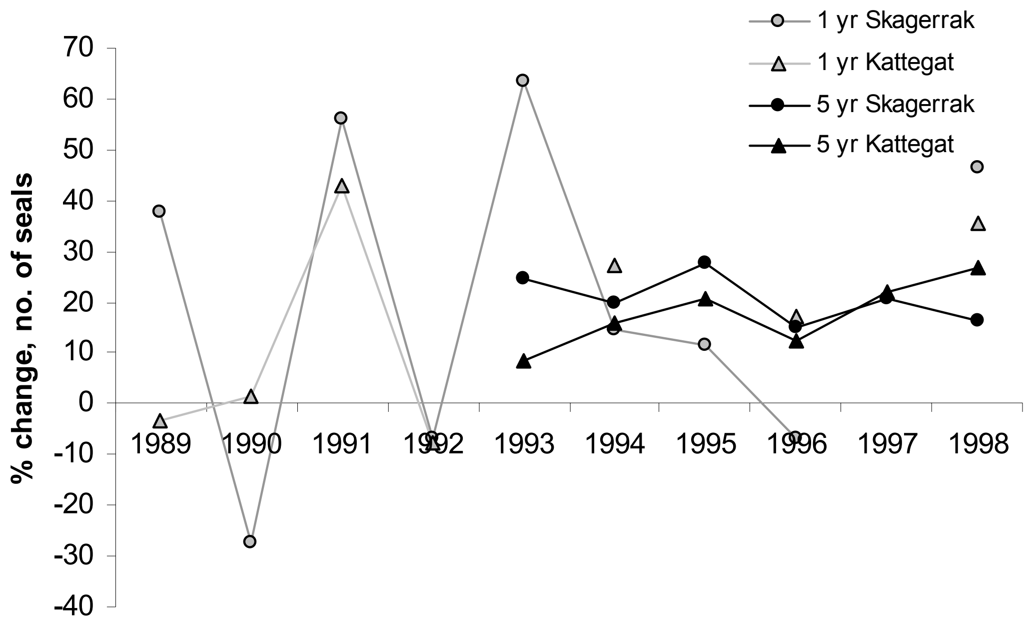

Figure 5.2. Annual and five-year running means of changes in harbour seal counts in the Skagerrak and Kattegat (Härkonen *et al.*, 2002).

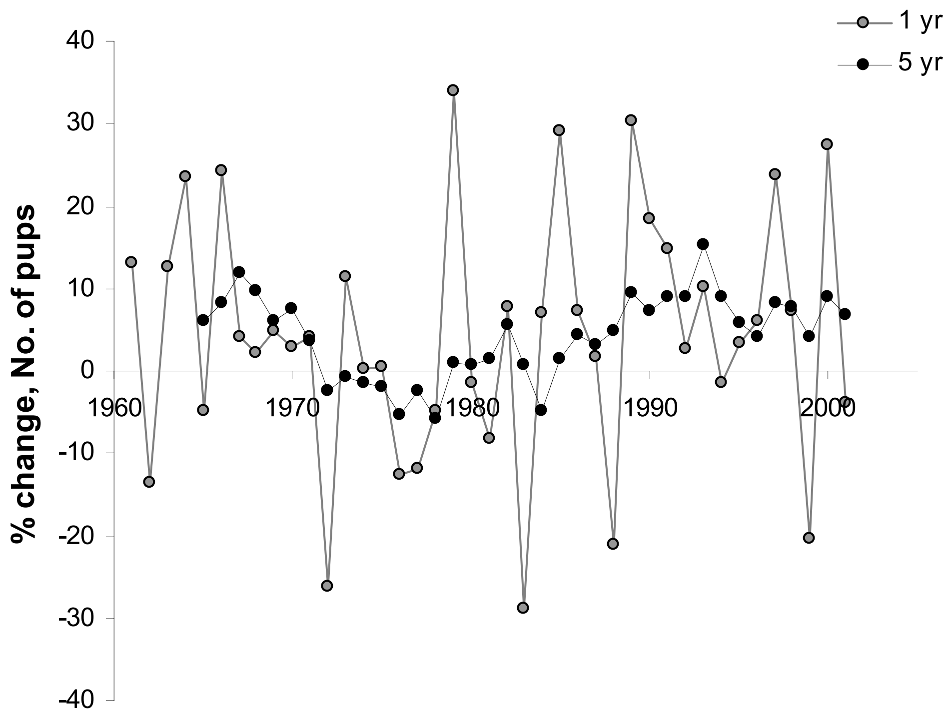

Figure 5.3. Time series of annual and five-year running mean changes in estimated grey seal pup production at major UK breeding sites in the North Sea, except Helmsdale, Orkney, and Shetland (after Duck, 2002).

The current EcoQO does not differentiate between subunits of the North Sea and it is unclear whether the EcoQO applies to the whole North Sea population or only to parts of it. This issue was reviewed in OSPAR BDC 04/02/08. It is not scientifically possible or valid to assess trends for the whole North Sea as there is (necessary) variation in count methods depending mostly upon the habitat in which the seals are giving birth or hauling out. Scientifically-consistent trends can be derived for sub-units of the North Sea, but it should be noted that these sub-units are not necessarily biologically separate. OSPAR BDC 04/02/08 proposed some sub-units that were accepted (subject to some correction) by the Biodiversity Committee (BDC 04_SR).

Table 5.1. Proposed sub-unit boundaries for the North Sea seal populations. Superscripts indicate the counting technique.

Grey seal		Harbour seal	
UK	Orkney	UK	Shetland[1]
	Fast Castle/Isle of May		Orkney[1]
	Farne Islands		North and East Scotland[1,2,3]
	Donna Nook		Southeast Scotland[2]
France			Greater Wash/Scroby Sands[2]
Netherlands		Netherlands	Delta area[2]
Germany	Schleswig-Holstein Wadden Sea	NL + DE + DK	Wadden Sea[2]
	Helgoland	Germany	Helgoland[3]
Norway	Kjørholmane (Rogaland)	Denmark	Limfjord[2]
		DK, SE, N	Kattegat, Skagerrak, Oslofjord[2]
		Norway	West coast, South of 62°N[2,3]

[1] Aerial surveys using thermal imaging.

[2] Aerial surveys using oblique photography.

[3] Land- and sea-based counts.

WGMME therefore suggests that:

- **The EcoQOs for seal population trends be subdivided as indicated in Table 5.1.**

If these three suggestions are followed, then the resulting, more specific, EcoQOs would be:

No decline in harbour seal population size (as measured by numbers hauled out) of ≥10 % as represented in a five-year running mean or point estimates (separated by up to five years) within any of eleven sub-units of the North Sea. These sub-units are: Shetland; Orkney; North and East Scotland; Southeast Scotland; Greater Wash/Scroby Sands; The Netherlands Delta area; Wadden Sea; Helgoland; Limfjord; Kattegat, Skagerrak and Oslofjord; West coast of Norway south of 62°N.

and

No decline in pup production of grey seals of ≥10 % as represented in a five-year running mean or point estimates (separated by up to five years) within any of nine sub-units of the North Sea. These sub-units are: Orkney; Fast Castle/Isle of May; Farne Islands; Donna Nook; France; Netherlands; Schleswig-Holstein Wadden Sea; Helgoland; Kjørholmane (Rogaland).

WGMME agreed with the summary of strengths and weaknesses of the harbour seal EcoQO in BDC 04/02/08.

The strengths include:

(a) Regular surveying at specific sites;

(b) even coverage of survey effort across most of the major concentrations of harbour seals in the North Sea;

(c) the ability to apply consistent methods of counting across years;

(d) long time-series of counts are already available in several key areas;

(e) several research programmes investigating the biology of the species.

Weaknesses include:

(a) Counts provide a measure of relative changes in the population of seals in a region and do not provide an accurate view of the total population using a region;

(b) counts of pups are not normally included, which means that the index of population size will have a low level of sensitivity to factors affecting reproductive rate.

WGMME agreed with the summary of strengths and weaknesses of the grey seal EcoQO in BDC 04/02/08. The strengths include:

(a) a long time series collected at a fine spatial and temporal resolution using a standardized method that will provide the statistical power to detect trends;

(b) a commitment within the UK and some other Contracting Parties/regions to collect data using consistent and robust methods into the future;

(c) compared with many other indices, data are relatively easy to collect;

(d) an active research programme exists that can underpin this index with biologically meaningful interpretations of trends in abundance; and

(e) grey seals forage throughout the North Sea so that this index is likely to integrate environmental variability across a wide range of spatial and temporal scales.

In contrast, the weaknesses include:

(a) a complex linkage between trends in pup production and probable trends in the population as a whole;

(b) uncertainty about the extent to which changes in pup production will be an indicator of environmental events or trends because they could be driven to an extent by internal population dynamics; and

(c) uncertainty about which environmental factors are likely to cause changes in pup production and about which stages in the life histories of grey seals are affected.

5.3 EcoQ for utilisation of seal breeding sites in the North Sea

No Ecological Quality Objective has been set for this metric and, as with the seal population trends (Section 5.2), the biology of the two seal species makes it sensible to separate the species. The key difference between the species for this EcoQO is that harbour seals give birth in intertidal habitats, with precise location apparently being influenced by both tidal and meteorological factors, while grey seals generally give birth in terrestrial habitats. The fluidity of precise breeding locations for many parts of the harbour seal population means that any definition of “site” would need to be drawn rather widely—at present there appears to be insufficient information to show how wide. In contrast, grey seal breeding locations are reasonably well-known and in the UK data exist for site usage over a number of years (BDC 04/02/08). For example, there are 24 sites where grey seals are known to have bred in Orkney. Of these, breeding has ceased at only two since 1960, while breeding started at several sites, roughly in parallel to the growing size of the population. There are several well-known grey seal breeding sites further south and east in the North Sea on coasts of the UK, The Netherlands, Germany, and Norway, but the sites used for breeding by the Shetland and French populations are less well-known.

If an EcoQO is to be defined for this Ecological Quality Element, WGMME thus suggests that only grey seals should be considered at present:

The number of grey seal breeding sites in Orkney, on the east coast of UK and coasts of The Netherlands, Germany, and Norway should not decline.

Further development of this Element (and Objective) could include:

a) better definition of breeding sites in Shetland and France;

b) development of techniques to distinguish separate harbour seal breeding sites.

5.4 EcoQO for harbour-porpoise by-catch

The knowledge behind this Ecological Quality Objective was relatively well developed at the time that it was first specified. One area that has remained unclear though relates to its geographical specificity. There is reasonable evidence of geographical and genetic sub-structuring of the harbour porpoise population in the North Sea, but the precise boundaries of this sub-structuring are not known. IWC (1999), in reviewing this issue in association with a request from ASCOBANS, decided that a relatively arbitrary boundary should be drawn across the northern North Sea from about Kinnairds Head (north of Aberdeen) to the Norwegian coast just north of Stavanger. This line was based mostly on the work of Walton (1997) and a hiatus in harbour porpoise distribution observed on this line by the abundance survey undertaken in 1994 (Hammond *et al*., 2002). Evidence presented in the most recent review of the sub-structure of North Atlantic harbour porpoise populations (Andersen, 2003) is consistent with this suggestion.

WGMME therefore suggests that this EcoQO might be made more specific:

Annual by-catch levels should be reduced to levels below 1.7% of the best abundance estimate in the North Sea, taking account of best information on any population sub-structure.

WGMME notes that this EcoQO therefore requires three pieces of information: an estimate of by-catch, an abundance estimate, and information on population sub-structure. There is existing information on all three of these, though WGMME notes that information on the abundance of harbour porpoises in the North Sea was collected ten years ago. There are plans to carry out a new abundance survey in 2005. There are schemes to collect information on by-catch in some fisheries, but by-catch rates in further important fisheries are not known (see review in ICES, 2003). Monitoring of harbour porpoise by-catch is under way in only three North Sea countries despite the fact that there is a duty to conduct such monitoring as part of the EU Habitats Directive. The European Commission has proposed a fishery regulation that will reinforce this requirement.

The best information available to WGMME indicates that this EcoQO is not being met and that fisheries management measures are required. WGMME are encouraged to note that some measures have recently been introduced by the European Union that should reduce harbour porpoise by-catch.

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6 MARINE MAMMAL PREY IN THE NORTH SEA

Term of reference e) *provide the Study Group on Multispecies Assessments in the North Sea with data on the consumption of different prey by marine mammals in the North Sea, in a format specified by the Study Group*

6.1 Background

Marine mammals common in the North Sea (ICES Area IV) include: grey seal (*Halichoerus grypus*), harbour seal (*Phoca vitulina*), harbour porpoise (*Phocoena phocoena*), minke whale (*Balaenoptera acutorostrata*), white-sided dolphin (*Lagenorhynchus acutus*), and white-beaked dolphin (*L. albirostris*) (Hammond *et al.,* 2002; Reid *et al.,* 2003, SCOS, 2003). Dietary data for these species are limited in space and time, and have been obtained from multiple sources including: strandings, by-catches, scientific takes, biopsy sampling, and scats (seals) (Boyle *et al.*, 1990; Prime and Hammond, 1990; Santos *et al.*, 1994, 1995, 1996; Thompson *et al*., 1996a, 1996b; Tollit and Thompson, 1996; Brown *et al*., 1997,1998, 2001; Hammond *et al*., 1994a, 1994b; Hall *et al.*, 1998; Pierce and Santos, 1996, 2003; Börjesson *et al.,* 2003; Reid *et al.*, 2003). Sample analyses have likewise employed a variety of methods including: enumeration of stomach contents or hard parts contained in scats (pinnipeds), fatty acid and stable isotope analysis (Iverson *et al.*, 1997; Tollit *et al.*, 1997; Walton *et al.*, 2000; Carter *et al.*, 2001; Hooker *et al.*, 2001; Das *et al.*, 2003; Bradshaw *et al.* 2003; Walton and Pomeroy 2003). All data are subject to a number of biases (Harvey, 1989; Pierce and Boyle, 1991; Smith *et al.*, 1997; Tollit *et al.*, 1997; Wijnsma *et al.*, 1999; Staniland, 2002) such as: sample collection methods, predator age/size class, differential digestion rates of prey, spatial/temporal changes in diet and/or habitat. Strandings and by-caught animals are unlikely to be representative of the age/size classes of the marine mammal populations in the North Sea. Further, strandings may include sick or dying animals as well as by-caught animals that were discarded at sea.

Prey identification and size (length and weight) can be determined from stomach and scat content analyses, whereas, chemical analysis may only reflect trophic level (Iverson *et al.*, 1997; Walton *et al.*, 2000; Hooker *et al.*, 2001; Das *et al.*, 2003; Bradshaw *et al.*, 2003; Walton and Pomeroy, 2003). Further prey population dynamics, ecosystem perturbations, and fisheries (aquaculture and commercial) will likely contribute to marine mammal dietary shifts (Furness, 2002).

6.2 Data Request

No dietary data sets for North Sea marine mammals were available for review at this meeting. Therefore, to address the request from the Study Group on Multispecies Assessments North Sea (SGMSNS), WGMME will contact North Sea marine mammal research organizations to ascertain the type of diet data available for the most recent five-year period (1999–2003), and data likely to become available during the current year. The SGMSNS data requirements are:

(a) marine mammal population numbers in ICES area IV, by year and quarter. If there are known differences in the diet-at-age, then it would beneficial to split the population numbers accordingly (e.g., juveniles and adults);

(b) diet by predator (and age categories) and quarter (although the quarters need not be in the same year). Diet data should be disaggregated by prey species and size (length or age). Diet composition should also be a relative estimate, i.e., % weight or volume;

(c) diet should ideally represent the whole North Sea population;

(d) diet should not be given as an average over a longer period. Point observations are necessary for estimating the model food suitabilities. When it is impossible to give diet for a particularly quarter of a year, diet should be given for a relatively short period of time;

(e) the SGMSNS request also stated that if only numbers of prey items consumed are available, then these will need to be converted (either by WGMME or SGMSNS) to biomass using published weight-length relationships.

WGMME recommends that data conversions should be conducted by the organizations that processed the diet materials.

Numerical modelling requires diet information that is representative of the temporal and spatial patterns of marine mammal habitats (Bradshaw *et al.*, 2003; Hindell *et al.*, 2003). WGMME notes that failure to incorporate dietary data quality into consumption models will certainly lead to inaccurate representation of marine mammal consumption rates (Thompson *et al.*, 2000; Bjørge *et al.*, 2002; Bradshaw *et al.*, 2003) in the North Sea.

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7 SUMMARY OF SIZE DISTRIBUTION AND STATUS OF MARINE MAMMAL POPULATIONS IN THE NORTH SEA FOR 2000–2004

Term of Reference 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.*

7.1 Introduction

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.

7.2 Harbour porpoise

7.2.1 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 (May to August) abundance for the years 2002 and 2003 was estimated to be 36672 animals (C.V.0.10) (Scheidat et al. 2004).

A further abundance survey for the North Sea and adjacent waters is planned for 2005 if funding is forthcoming.

7.2.2 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 7.1). 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.

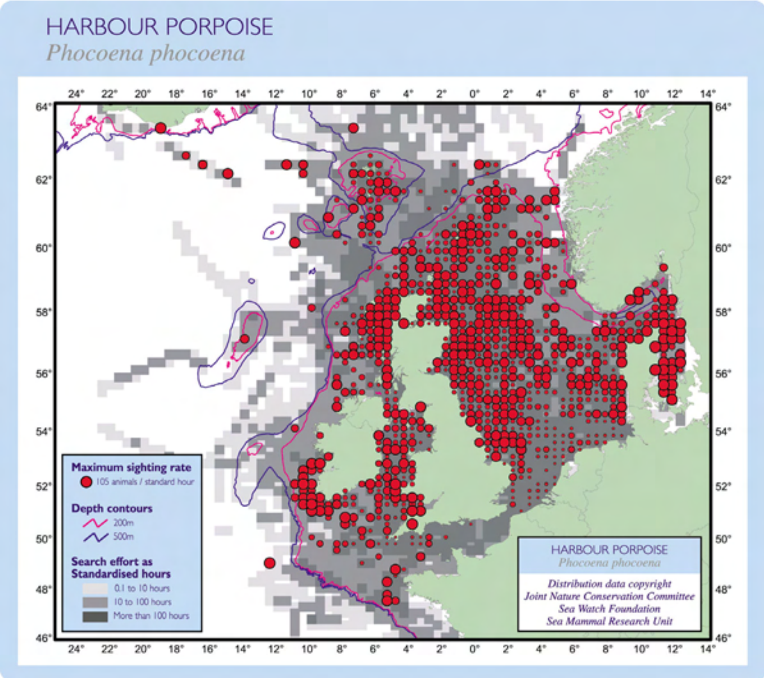

Figure 7.1. Distribution of harbour porpoises (Reid *et al*. 2003).

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 7.2).

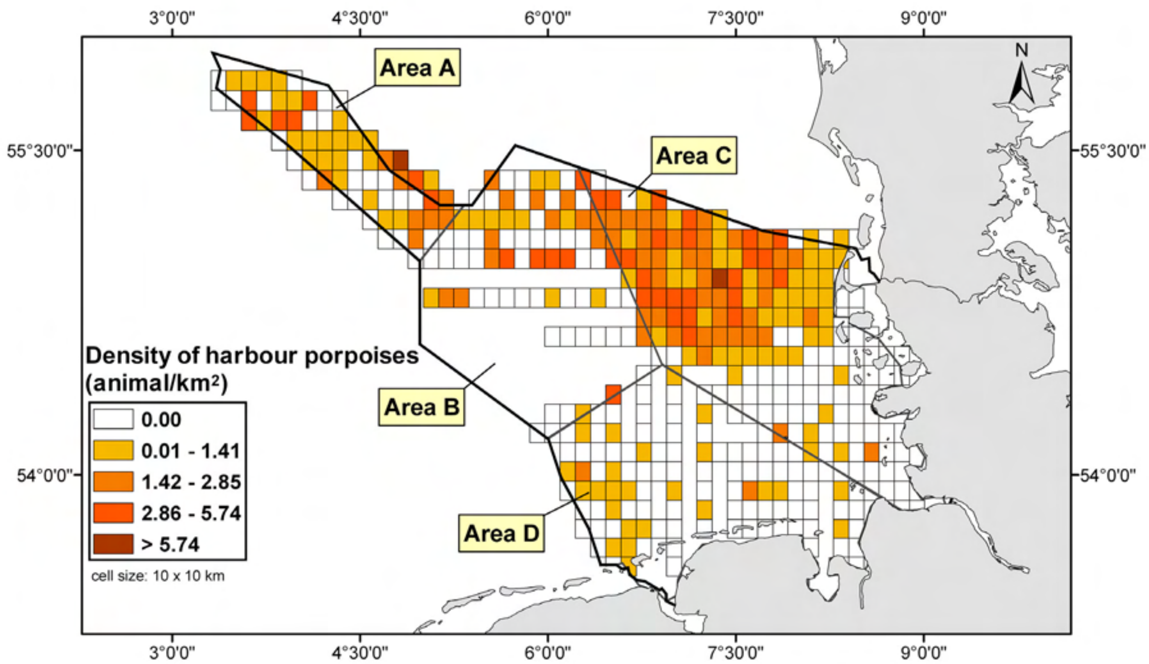

Figure 7.2 Map showing the distribution of harbour porpoises in the German North Sea for May to August 2002 and 2003. Density is shown as animals per km² per cell (10x10km²). Only flights conducted in good or moderate conditions were included (from Scheidat et al. 2004).

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 7.3).

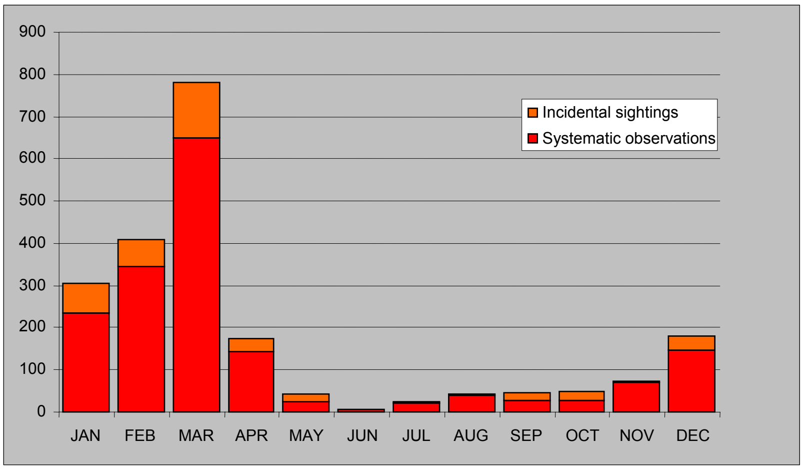

Figure 7.3. Seasonal pattern of harbour porpoises reported from coastal sites in the Netherlands since 1970 (Marine Mammal Database, updated 3/1/2004, http://home.planet.nl/~camphuys/Bruinvis.html)

7.2.3 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 7.4).

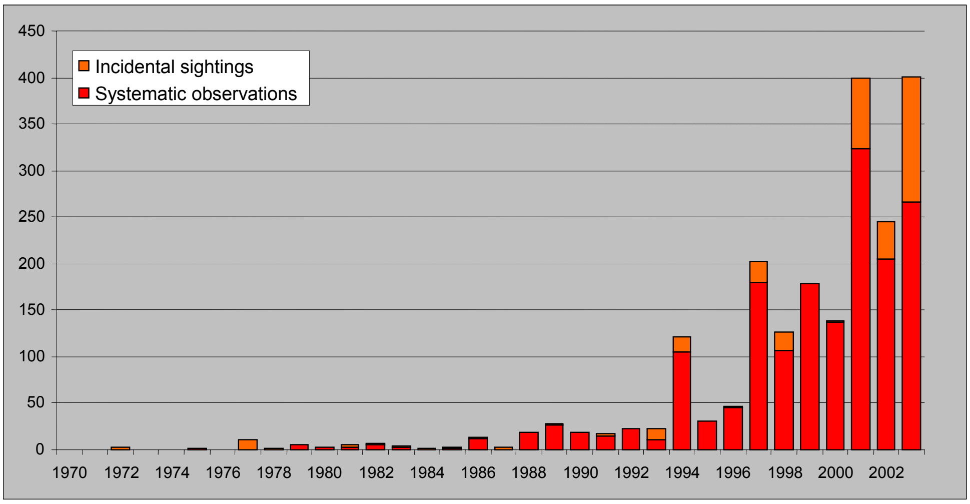

Figure 7.4. Harbour porpoises reported from coastal sites of The Netherlands since 1970 (Marine Mammal Database, updated 3/1/2004, http://home.planet.nl/~camphuys/Bruinvis.html.)

Similarly, the strandings along the Belgian coast have increased (Haelters *et al.*, 2002 and pers. comm) (Figure 7.5). 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.

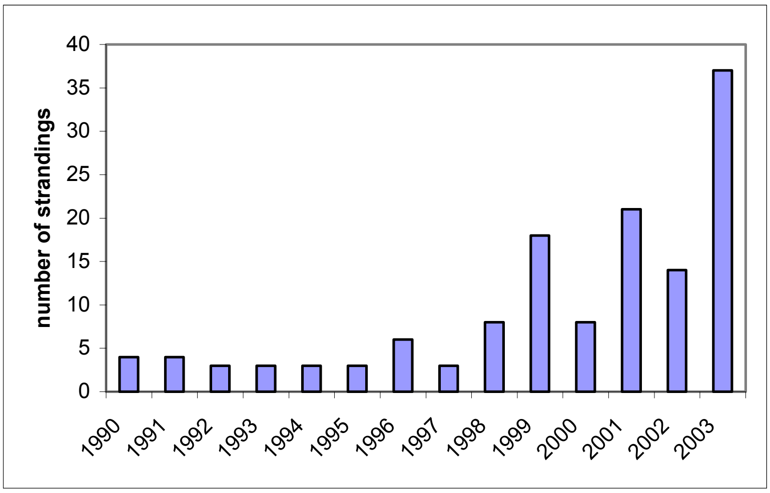

Figure 7.5. Number of harbour porpoise strandings on Belgian coasts, 1990 to 2003. From Haelters *et al*. (2002) and J. Haelters, pers. comm.

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.

7.3 White-beaked dolphin

7.3.1 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.

7.3.2 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 7.6 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).

7.3.3 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.

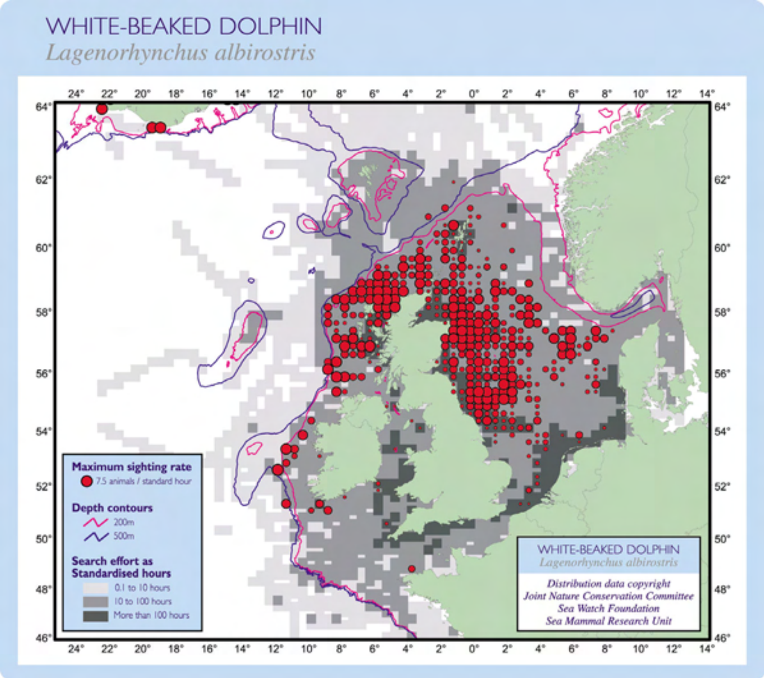

Figure 7.6. Distribution of white-beaked dolphins (Reid *et* ***al.***, 2003).

7.4 Atlantic white-sided dolphin

7.4.1 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.

7.4.2 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 7.7) (Reid *et al.*, 2003).

7.4.3 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.

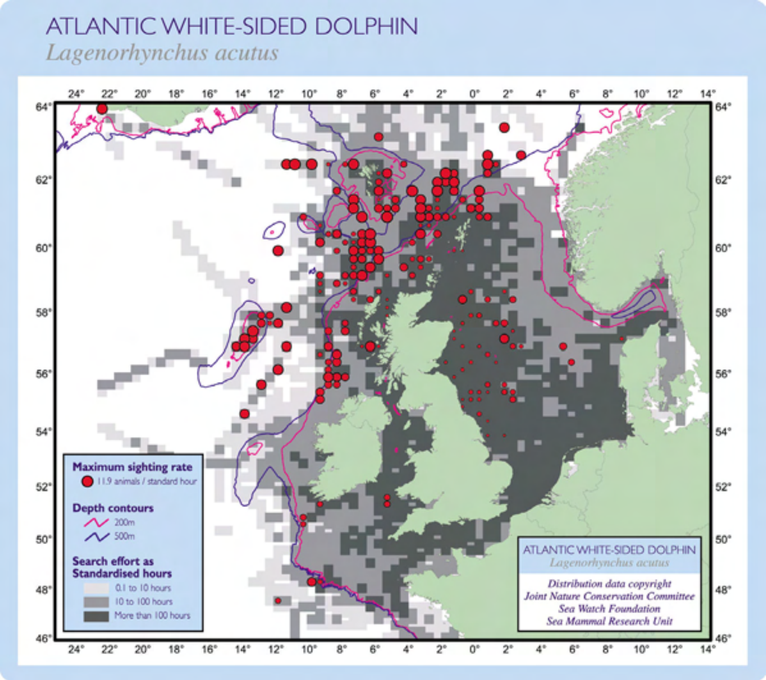

Figure 7.7. Distribution of Atlantic white-sided dolphins (Reid *et al.*, 2003).

7.5 Bottlenose dolphin

7.5.1 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).

7.5.2 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 7.8).

7.5.3 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.

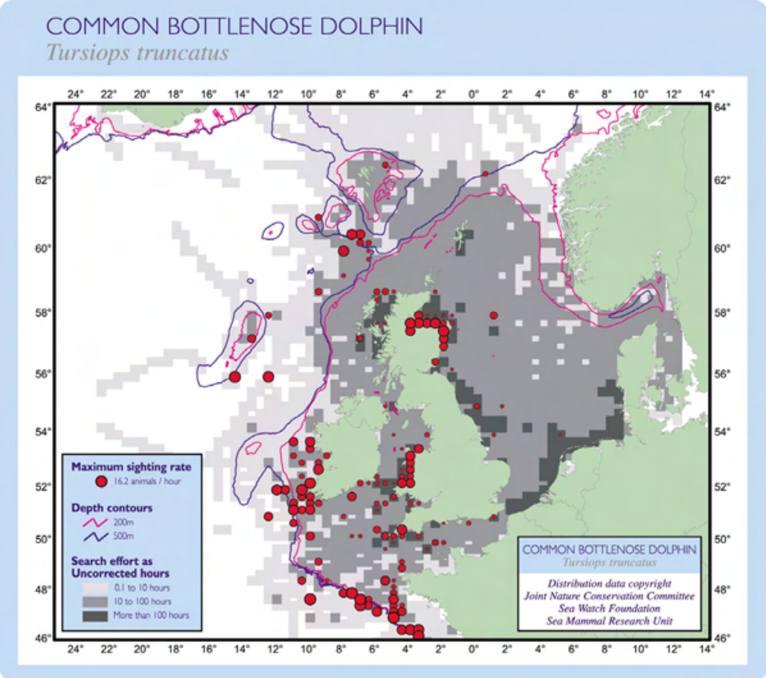

Figure 7.8. Distribution of bottlenose dolphins (Reid *et* ***al.***, 2003).

7.6 Minke whale

7.6.1 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.

7.6.2 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 7.9).

7.6.3 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.

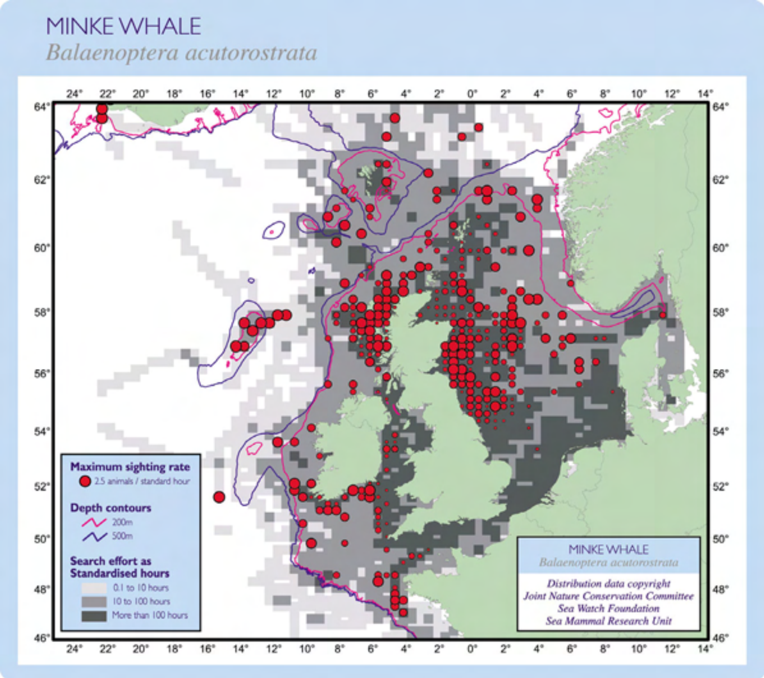

Figure 7.9. Distribution of minke whales (Reid *et al.*, 2003).

7.7 Harbour seal

7.7.1 Population size

The size of harbour seal populations in the North Sea is discussed in the Section 4 (epizootic) and Section 5.2 (trends) in this report.

7.7.2 Population distribution

It has been long thought that the distribution of harbour seals in the North Sea was predominantly coastal (Figure 7.10). 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 7.11).

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.).

7.7.3 Population status

Trends are discussed in Section 5.2 of this report.

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 (see Section 4 on PDV).

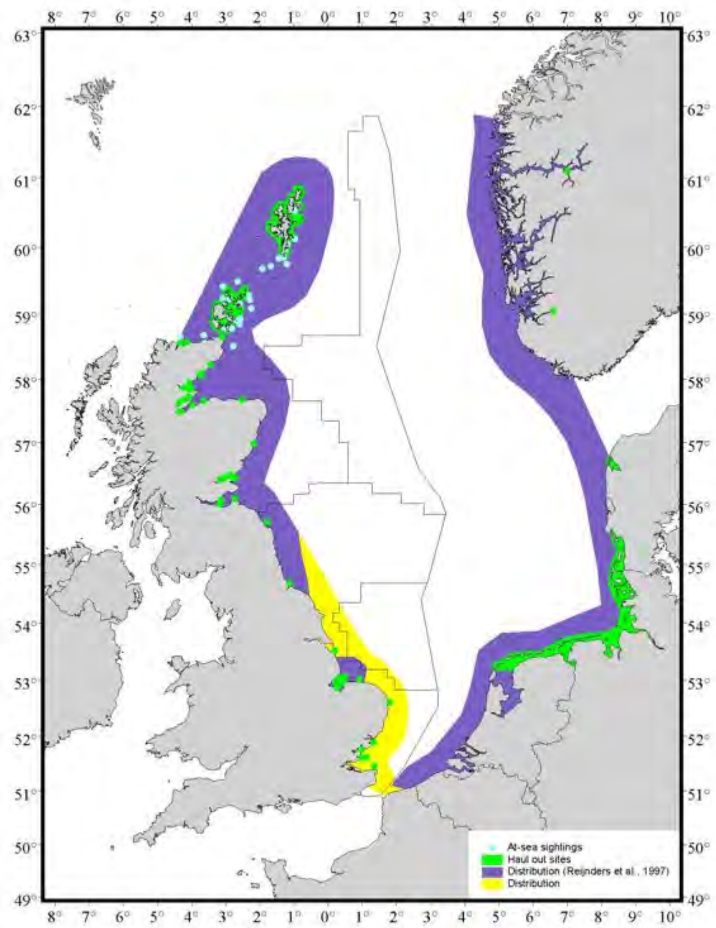

Figure 7.10. Distribution of harbour seals in the North Sea. Map extended from Reijnders *et al.* (1997) to take into account additional known haul-out sites in the southwestern North Sea. At-sea sightings from Pollock *et al.* (2000) are also shown. Source: DTI, 2002.

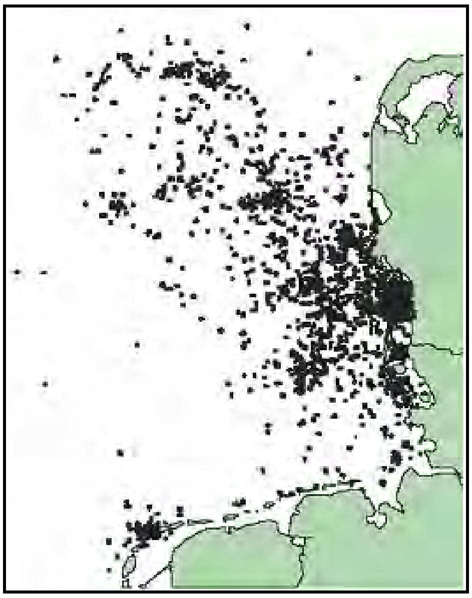

Figure 7.11. Telemetry data from harbour seals tagged at Horns Rev (http://www.hornsrev.dk/).

7.8 Grey seal

7.8.1 Population size

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

7.8.1.1 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.

7.8.1.2 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 7.1).

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 7.1 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

7.8.1.3 Germany

Relatively few grey seal pups are born on German coasts (Table 7.2). There is a gradual increase in numbers, but note the large inter-annual fluctuations (SCOS 2003).

Table 7.2 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

7.8.1.4 Total numbers of grey seals breeding in the North Sea

Table 7.3 shows current estimates for total numbers in the North Sea.

Table 7.3. Current estimates of abundance of grey seals in North Sea waters (SCOS, 2003).

Region	Year	Estimate of abundance
UK	2002	54,600
Germany	1998	100
The Netherlands	2000	500
France		>80
Norway	2003	35 (pup count, not extrapolated)

7.8.2 Population distribution

The distribution of grey seal pupping (North Sea) and moult haul-out is well-known and included in Tables 7.1, 7.2, and in parts of Section 7.8.1.

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 7.12). 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 7.13).

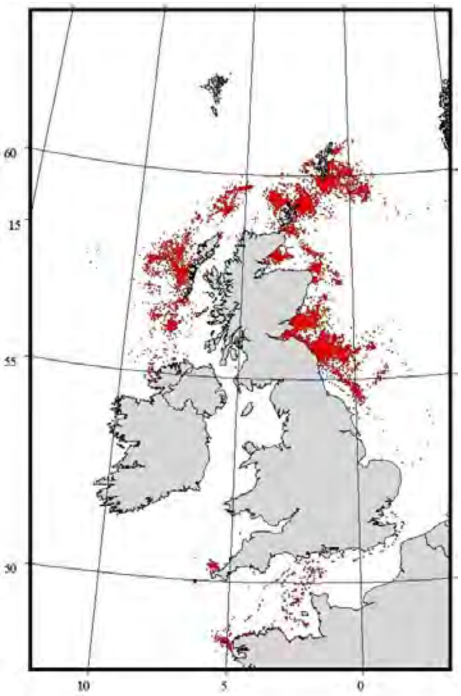

Figure 7.12. Locations of 108 grey seals fitted with satellite-relay data loggers over a period of about ten years (McConnell *et al.*, 1999).

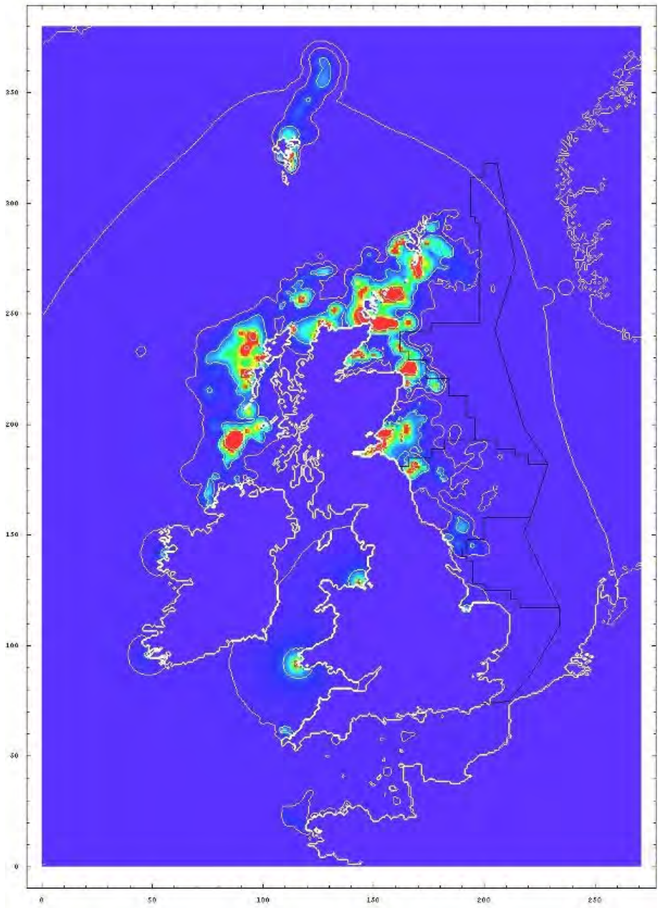

Figure 7.13. Distribution of grey seals foraging around the British Isles (predicted by a spatial model using the satellite-linked telemetry data from Figure 15 and other SMRU unpublished data). Source: Matthiopoulos *et al*. (in press) cited in: DTI, 2002.

7.8.3 Population status

Trends in pup production in UK and German North Sea colonies over the past twenty years are shown in Tables 7.1 and 7.2, summarised in Table 7.4. Pup production remained nearly static between 2000 and 2001 and showed a small decline in 2002.

Table 7.4. 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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8 GREY SEALS IN THE BALTIC

8.1 Grey seal abundance in the Baltic

Seal researchers from Estonia, Finland, Russia, and Sweden met at the Finnish Game and Fisheries Research Institute on 23 February 2004 to summarize the results of the grey seal surveys carried out in these countries in 2003.

The waters of these countries contain almost the entire present distribution of the grey seal in the Baltic Sea. In constructing an international summary over the whole sea area, only censuses performed during a common two-week period at the end of May/beginning of June were used to minimize the possibility of double-counting due to seal movement.

At this time of the year, maximum numbers of grey seals haul out of the water to moult and can be counted on land or on ice. In Finland, aerial censuses aided by aerial photography are used, whereas counts from boats and from land are used in the other countries. Census results are presented by sea area (Table 8.1), rather than by country. It is likely that real population size is larger than the total number counted as some animals will not be hauled out at the time of the surveys.

Seals can travel long distances in a short period of time, so the possibility exists of double-counting or under-counting, in spite of the common census period of two weeks. In 2003, the almost perfect synchronisation of censuses in adjacent areas with high numbers of seals minimised these risks.

The need for synchronisation is important in future censuses, especially in the core area of the Baltic grey seal distribution in the archipelagos off central Sweden, southwestern Finland, and western Estonia, where a total of about 13,000 grey seals were counted in 2003.

Table 8.1 Grey seal abundance estimates in the Baltic, 2003 (Finnish Game and Fisheries Research Institute, unpublished).

Region	**Estimate of abundance**
Bothnian Bay and North Kvark	710
Bothnian Sea excluding the Åland Archipelago	855
SW Finnish archipelago including the Åland Archipelago	6880
Swedish Baltic between Gulf of Bothnia and 58°N (northern tip of Gotland)	3980
Gulf of Finland	490
Western Estonia	2700
Swedish Baltic south of 58°N	335
Total	15,950

8.2 Population trends of grey seals in the Baltic

The 2003 abundance estimate (15,950) is the highest in recent history, and is the fourth consecutive increase since 2000 (i.e., 9,700 in 2000, 10,300 in 2001, and 13,100 in 2002). It is likely that improvements in census methods have caused some of the apparent increase in 2003, as large-scale immigration to the Baltic seems unlikely, and seal populations do not grow at this rate (approximately 17% from 2000 to 2003).

8.3 Seal sanctuaries in the Baltic

Seal sanctuaries in Finland have been established principally for grey seals, although other species such as the ringed seal (*Phoca hispida botnica*) may also benefit to some extent. The first sanctuary was established in the southeast Åland Archipelago in 1998 (E. Helle, Finish Game and Fisheries Research Institute, pers. comm.) Seven additional areas were established in 2001 in mainland Finland, covering the entire Finnish Baltic coast but concentrating on the southwestern archipelago of Finland, where the population size of the grey seal is highest.

The seal sanctuaries have been established in the haul-out sites favoured in the longer term. In the 2002 and 2003 censuses in late May/early June, 50.3% of grey seals were found in the sanctuaries off the coast of mainland Finland and 25.5% on the Åland Islands.

9 FUTURE WORK OF THE WGMME AND RECOMMENDATIONS

9.1 Future Work of WGMME

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.

9.2 Recommendation for Future Meeting

WGMME (Chair: Gordon T. Waring, USA) agreed that the best dates for the next annual meeting will be 30 May – 2 June 2005 at Savonlinna, Finland.

WGMME recommended that activities for the 2005 meetings include:

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

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

Justification:

a) 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, bycatch, and stranding. This addresses Goal 1, 2 and 5 in the ICES Strategic Plan.

b) This work is required in relation to a request from the European Commission. This also addresses Goal 1 of the ICES Strategic Plan.

c) 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.

10 RESOLUTION

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.

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