Agenda Item 5.1

Implementation of the Triennium Work Plan (2010-2012) – Other Issues
Review of New Information on Population Size, Distribution, Structure and Causes of Any Changes

Document 5-06

ICES 2010: Report of the Working Group on Marine Mammal Ecology

Action Requested

- Take note

Submitted by

United Kingdom

NOTE:
IN THE INTERESTS OF ECONOMY, DELEGATES ARE KINDLY REMINDED TO BRING THEIR OWN COPIES OF DOCUMENTS TO THE MEETING

ICES WGMME REPORT 2010

ICES ADVISORY COMMITTEE

ICES CM 2010/ACOM:24

Report of the Working Group on Marine Mammal Ecology (WGMME)

12–15 April 2010

Horta, The Azores

International Council for the Exploration of the Sea

Conseil International pour l'Exploration de la Mer

International Council for the Exploration of the Sea
Conseil International pour l'Exploration de la Mer

H. C. Andersens Boulevard 44–46
DK-1553 Copenhagen V
Denmark
Telephone (+45) 33 38 67 00
Telefax (+45) 33 93 42 15
www.ices.dk
info@ices.dk

Recommended format for purposes of citation:

ICES. 2010. Report of the Working Group on Marine Mammal Ecology (WGMME), 12–15 April 2010, Horta, The Azores. ICES CM 2010/ACOM:24. 212 pp.

Contents

Executive summary

The Working Group on Marine Mammal Ecology (WGMME) met at the University of the Azores in Horta, The Azores from 12 April to 15 April 2010. Sinéad Murphy chaired the meeting of 25 participants, representing twelve countries.

Seven different ToRs were assessed, covering a wide range of issues, including reviewing the effects of wind farm construction and operation on marine mammals and assessing the current contaminant loads in marine mammals within the ICES Area. Other topics included reviewing population abundance, structure and status of marine mammals off the Azores, further development of a framework for surveillance and monitoring of marine mammals, evaluating the scope for a European marine mammal tissue bank, assessing the current status of the Saimaa ringed seal population, grey seal prey consumption in UK waters, and further development of the ICES seal database. The European Commission also requested an assessment of the population status of cetaceans concerned by EC Regulation 812/2004.

The WG outlined and reviewed the potential negative impacts of wind farms (construction and operation) on marine mammals and provided advice on research needs, monitoring and mitigation schemes. The WG made a number of recommendations with regard to wind farm developments, including the establishment of means for efficient dissemination of results of common interest and means of making available previous EIA reports and previously collected baseline data for subsequent studies and assessments. The WG also recommended that multinational studies should be undertaken, and management decisions regarding offshore wind farms should be based on appropriate populations and/or management units for the relevant marine mammal species, irrespective of national borders. Further, development of methods to assess the cumulative effects on marine mammals of the underwater noise level caused by the simultaneous wind farm construction and operation at nearby sites. Other recommendations relate to improving our understanding of the characterization sources of underwater noise associated with the construction and operation of offshore wind farms, establishing common accepted tolerance limits for acute noise exposure in marine mammals and the development of common guidelines for mitigation in relation to pile driving.

An overview on current contaminant loads in marine mammals inhabiting the ICES Area is presented within this report, and highlights (regions and) marine mammal populations at highest risk from environmental exposure. Further, the cause–effect relationships between contaminants and health status, and the population-level effects of environmental impacts were also assessed. Despite being banned for two to three decades, polychlorinated biphenyls (PCBs) still occur at concentrations that exceed proposed thresholds for mammalian toxicity in some marine mammal top predator species, including bottlenose dolphins, killer whales and polar bears. Compared with many other legacy pollutants, PCBs are declining only very slowly in many geographic regions (e.g. harbour porpoises in UK waters). Given the high levels of PCBs in marine mammals (compared with proposed toxicity thresholds), the resistance of PCBs to environmental degradation and their relative toxicity, PCBs undoubtedly continue to pose the greatest toxicological threat to some marine mammal species within the ICES Area. The WG recommended that research is needed to assess trends in contaminant exposure (PCBs and newer contaminants) and to conduct risk assessments for health and reproductive effects from contaminant exposure in species of highest risk (e.g. killer whales, St Lawrence belugas, polar bears, bottlenose dolphins, and Baltic marine mammals).

As part of this year's meeting, a full assessment of the cetaceans inhabiting waters off the Azores was undertaken. This was the first of its kind for this region, and incorporated information on population structure, abundance, habitat use, seasonal movements, and the potential impacts to local populations from whale watching activities and incidental capture. Based on work undertaken by the University of the Azores, and others, a regulation was developed for whale watching, which was implemented into law in 1999. The regulation stipulates the types of manoeuvres and boat speeds that are permissible around cetaceans, and also currently limits the number of whale watching licences by zone. However, the effectiveness of the regulation may be compromised by a lack of law-enforcement and the WG strong recommends the implementation of an efficient law-enforcement scheme. Although low rates of accidental death due to interactions with fishing gear have been reported for marine mammals in Azorean fisheries, since the opening of waters beyond 100 nm to European deep-water fleets in 2003, the actual bycatch rate in the region may be higher. The WG recommended the implementation of bycatch monitoring of European-deep-water fleets in this region to establish the bycatch rate.

The WG discussed extensively the development of Europe-wide networks for monitoring (e.g. for strandings, sightings, and bycatch), as well as the establishment of common databases (for strandings, sightings and bycatch data) and sample banks. One such initiative that was discussed at the meeting was the development of a European Marine Mammal Tissue Bank (EMMTB). This would entail (a) identifying laboratories and institutions involved in the post-mortem investigation (full necropsy and tissue sampling) of marine mammals in the North-east Atlantic, (b) collating information on the availability and location of samples, (c) developing bilateral collaborations between laboratories and institutes to fulfil the objectives of a tissue bank, including the establishment of a steering committee to manage sample loans and data exchanges, and (d) developing a website and meta-database for the EMMTB, with links to national websites and databases. Further, as part of this initiative, and recommended in a few other ToRs, there is a need for standardization of marine mammal stranding network protocols for conducting necropsies, storing samples and conducting contaminant analyses across the ICES Area. Other initiatives include the ICES seal database for harbour and grey seals which is currently being populated by members of the WG with data from seal population monitoring programmes throughout the Northeast Atlantic.

Establishment of these initiatives would enable the 'unit of monitoring' to be the natural population or (minimally) broad-scale spatial divisions that take into account the transboundary nature of most marine mammal populations; rather than national waters which are currently used in the Habitat's Directive Favourable Conservation Status assessment reports. This was expanded upon in further detail in the framework for surveillance and monitoring of marine mammals within the ICES Area. In addition to the establishment of a Steering Group composed of representatives from all relevant bodies, and assessing marine mammals at the natural population and/or management unit level, it was proposed in the framework to undertake an adaptive monitoring and surveillance approach, under which objectives, monitoring and outcomes are regularly reviewed and updated by the Steering Group. The WG advocated that this approach would improve the mechanisms for translating monitoring findings into appropriate management actions for marine mammals.

1 Opening of the meeting

The Working Group on Marine Mammal Ecology (WGMME) met at the University of The Azores in Horta, The Azores, from 12 April to 15 April 2010. The list of participants and contact details are given in Annex 4.

The meeting was opened by the Secretary of the Environment of the Azores.

The Working Group thanks the University of the Azores for their invitation to conduct the meeting in Horta. The Working Group gratefully acknowledges the support given by several additional experts that kindly provided information and/or reports for use by WGMME and reviewed parts of the Report. The Chair also acknowledges the diligence and commitment of all the participants before, during and after the meeting, which ensured that the Terms of Reference for this meeting were addressed.

2 Acknowledgements

The Working Group also gratefully acknowledges the support given to us by Phil Hammond, Michael Fontaine, Jonathan Gordon, Alex Aguilar, Ana Cañadas, Rob Deaville and Kees Camphuysen who kindly provided unpublished data, text and/or reports for use by the WGMME. The Working Group also thanks Michelle Cronin, Wayne Ledwell, Nynke Osinga, Gisli Vikingsson and Fabian Ritter for providing information on their marine mammal monitoring programmes in their respective countries.

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

3 Adoption of the agenda

The following Terms of Reference and the work schedule were adopted on April 12th.

a) Review the effects of wind farm construction and operation on marine mammals and provide advice on monitoring and mitigation schemes;

b) Review the current contaminant loads reported in marine mammals in the ICES area, the cause–effect relationships between contaminants and health status, and the population-level effects of environmental impacts;

c) Further development of the framework for surveillance and monitoring of marine mammals applicable to the ICES Area;

d) Review and report on any new information on population sizes, population/stock structure and management frameworks for marine mammals;

e) Provide information on abundance, distribution, population structure and incidental capture of marine mammals off the Azores;

f) Review of the scope, objectives and technical issues of the initiative for a European Marine Mammal Tissue Bank;

g) Update on development of the ICES seal database, status of intersessional work.

WGMME will report to the attention of the Advisory Committee (ACOM) by 29 April 2010.

Supporting Information: Scientific Justification and relation to Action Plan:

Resource requirements:	No specific requirements beyond the needs of members to prepare for, and participate in, the meeting.
Participants:	The Group is normally attended by some 20–25 members and guests.
Secretariat facilities:	None.
Financial:	No financial implications.
Linkages to advisory committees:	WGMME reports to ACOM
Linkages to other committees or groups:	
Linkages to other organizations:	

4 ToR a. Review the effects of wind farm construction and operation on marine mammals and provide advice on monitoring and mitigation schemes

4.1 Introduction

Significant gaps exist in our knowledge of the possible impacts on the environment from the construction and operation of offshore windfarms. Given the number of windfarms being constructed or planned for realization in the near future, many research projects are currently assessing possible and actual effects of windfarm construction and operation on the different components of ecosystems, such as marine mammals. Also in the near future, developments in tidal turbines and wave generators are likely to increase and although some issues relating to marine mammals may be different from those of offshore windfarms, the general issues of concern remain the same.

During the construction and operation of offshore windfarms many activities can be identified which may, due to their noise emissions, have an effect on marine mammals; these are among others bottom profiling, ship traffic, pile driving and other construction activities, and the operation itself. The impacts mentioned here are not comprehensive, but are considered of the greatest concern at the present point in time. This report primarily focuses on the impacts due to increased underwater noise during the construction (especially from pile driving) and operation phases. It should be noted however that there may be some positive effects of windfarms, such as no take fishing areas, but these effects will not be discussed here in detail.

The harbour porpoise is the most abundant marine mammals in the continental shelf waters of the North-east Atlantic Ocean, including the North Sea. As it is considered a species which is sensitive to human generated underwater noise (e.g. Bain and Williams 2006; Cox *et al*., 2001; Thompson, 2000; Verboom and Kastelein, 2005), it is natural that up to now most impact studies have focused on this species. A more limited number of studies have focused on the effects on other species, such as seals and the bottlenose dolphin. However, as windfarms are gradually being constructed further offshore and in areas where they were not considered before, other species may be affected, e.g. the minke whale and white-beaked dolphin.

The different environmental, technical and legal aspects of underwater noise from windfarms have recently been the subject of international workshops, among others in Hamburg (TPWind - Underwater noise and offshore windfarms, 2–3 June 2009), London (Underwater Sound Forum - Assessing and managing the potential impact of marine piling noise within the evolving regulatory framework, 24 February 2010), and Stralsund (ECS/BSH - Pile driving in offshore windfarms: effects on harbour porpoises, mitigation measures and standards, 21 March 2010).

It is not possible to present here an in-depth review of the studies that have been made, or which are ongoing. Also, recent in-depth reviews on the effects of offshore windfarms on the ecosystem, which summarize the results of a large number of research projects have been, or are being, produced; such as OSPAR (2008; 2009a; 2009b). Therefore this report will provide an overview of the currently available studies on effects of offshore wind farms and focus on highlighting current research needs and important issues of regulation and management to be addressed in the coming years.

4.2 Effects of wind farm construction and operation on marine mammals

The construction and operation of offshore windfarms should ultimately be evaluated in terms of their effects on marine mammal populations (or relevant management units). Negative impacts could be identified by changes in parameters such as fecundity, calf/pup survival, and juvenile and adult mortality. As the cumulative impacts of offshore windfarms may lead to a decrease in population size, regulations must be population based and take on-board that marine mammals are, on the whole, migratory species. However, to date research has been limited to within national boarders and therefore it has been difficult to assess the cumulative impacts.

When evaluating the impact of windfarms it is useful to separate the assessment into the three phases: construction (including site surveying prior to construction), operation and decommissioning. Decommissioning is fundamentally similar to the removal of other types of offshore structures, such as oil and gas platforms, and will not be covered here; except for mentioning that offshore wind farm developers and licensing authorities should be encouraged to consider decommissioning within the design phase.

To date, direct impact studies have been conducted during the construction and/or operation phases in several offshore wind farms. Details of six of these are summarized in Table 1. In addition to these, measurements of noise were obtained from pile driving and turbines in operation from a number of wind farms in Denmark, Sweden, Germany, Netherlands, Belgium and the UK. However, in most cases noise measurements were not coupled directly to measurements of effects on marine mammals and are thus not included. An overview of sources of impact, relevant impact studies, research needs and mitigation measures are provided in Table 2. More detailed discussion on mitigation measures and monitoring are presented later.

Table 1. A summary of impact studies undertaken during the construction and operation phases of offshore wind farms.

Name	Location	Construction year	Foundation type	Pile diameter	Water depth	No of turbines	Turbine type	Monitoring	Comments
Horns Reef I	Danish North Sea	2002	Monopiles	3.8 m	6–12 m	80	2 MW	Construction and operation	
Nysted	Western Baltic, Denmark	2002–2003	Gravitational foundations	n.a.	5–10 m	72	2.2 MW	Construction and operation	Sheet piling conducted during construction
Beatrice	Outer Moray Firth, Scotland	2006	4-legged jacket	1.8 m	42 m	2	5 MW	Construction	4 piles per foundation
Egmond aan Zee	Dutch North Sea	2006–2007	Monopiles		18–20 m	36	3 MW	Operation	
Horns Reef II	Danish North Sea	2008	Monopiles	4 m	5–15 m	95	2.3 MW	Construction	Located 15 km from Horns Reef I
Alpha Ventus	German Bight	2009	4-legged jacket and tripod	2.6 m	25 m	12	5 MW	Construction and operation	Transformer platform

Table 2. An overview of sources of impact, relevant impact studies, research needs and mitigation measures.

Impact	Pile driving	Construction in general	Operation	Service activities	Changes to habitat
Observed effects	Harbour porpoises: Decrease in acoustic activity out to at least 20 km (2; 3; 4; 10) Decreased abundance well beyond construction site in visual surveys during pile driving (6). One study showed decreased porpoise acoustic activity at the piling site (Beatrice), but no significant change at a control site 40 km away (8). Seals: Decreased numbers at a nearby haulout site during piling (5). Indication of avoidance out to 40 km by animals fitted with SRDLs tags (Egmond aan Zee wind farm)	Harbour porpoises: Decreased abundance during construction phase (2; 3; 9; 11). Seals: Limited information. No general effect of construction on haul out behaviour, except a partial displacement to alternative haulout sites in the pupping season (e.g July in harbour seals) (5).	Harbour porpoises: Three studies indicate no negative effect during operation (1; 11; 13). A study from Nysted windfarm demonstrated decreased abundance two years after construction (3). However, a subsequent study did not report variations in abundance between the Nysted windfarm site and adjacent areas (1). Seals: No effect detected in satellite tagged animals, though very few animals were tagged (Egmond aan Zee wind farm)	No evidence of effect but limited information available	Limited information available. One study (13) observed increased harbour porpoise abundance inside an operating windfarm, which may be related to exclusion of fisheries and/or ships.
Significance of impact	Significant risk of hearing damage to seals and harbour porpoises, even under current mitigation schemes. Nature of behavioural impact is unknown, but could be significant.	Partial or complete habitat loss during period of construction. Significance depends on scale of project, abundance of animals and nature of surrounding habitats. Impact beyond the construction site is possible if migration routes are affected but no studies are available on this. Indirect effects through altering local prey abundance have not been assessed to date.	Significance for small cetaceans likely to be low (7; 12). Significance for other species with better low frequency hearing (e.g. baleen whales and seals) is unknown, though could be greater. Impact could be significant if migration routes are affected.	By nature similar to impact from other ship and boat traffic activities. Cumulative effects should be considered, i.e. taking into account other non-construction boat traffic.	Introduction of hard substrata will change prey species composition. Reduction of fishing activities will affect prey abundance and size distribution. Effects on marine mammals have not been assessed. Though significant changes to ice habitats (Baltic Sea) may occur due to foundations and service vessel traffic. This may affect the distribution and abundance of seals.

Impact	Pile driving	Construction in general	Operation	Service activities	Changes to habitat
Research needs	Cumulative effects of several simultaneous pile driving operations in the same area. Elucidation of the nature of behavioural response of seals and cetaceans. Establishment of links between behavioural response and impact on fitness (reduced survival and/or fecundity). Determination of possible links between spectral properties of noise and size of impact area.	Determination of population level effects by temporary habitat loss. Assessment of effects from individual activities during construction.	Determination of extent of habitat loss (if any). Assessment of effect on migration routes (if relevant). Determination of population level effects of partial habitat loss	Establishment of link s between service activities and alterations in abundance /behaviour. Determination of population level effect of disturbance.	Investigation of fine-scale habitat use inside the wind farm to address whether marine mammals exploit the artificial reefs. Determination of net population level effects (positive or negative) of changes in habitat.
Mitigation (if required)	Visual observers only detect some animals and therefore this method alone is not efficient. Ramp up/acoustic deterrent devices partially address acute hearing damage. Reducing impact on behaviour can be undertaken by reducing radiated energy at relevant frequencies or by limiting installation to periods with low marine mammal abundance and/or by changes in methodology.	Construction should occur during periods with low abundance. Further, noise emission from other sources (e.g. ships, boats etc.) should be reduced.	Modification of turbines and foundations to reduce noise emission at relevant frequencies.	Selection of service vessels based on minimizing impact. Larger maintenance operations should be located in periods with low marine mammal abundance.	Changes to design of foundations and scour protection.

Impact	Pile driving	Construction in general	Operation	Service activities	Changes to habitat

1. Blew, J., Diederichs, A., Grünkorn, T., Hoffmann, M., and Nehls, G. (2006) Investigations of the bird collision risk and the responses of harbour porpoises in the offshore wind farms at Horns Rev, North Sea and Nysted, Baltic Sea, in Denmark. Status report 2005 to the Environmental Group. Hamburg, BioConsult SH.
2. Brandt, M. J., Diederichs, A., and Nehls, G. (2009) Harbour porpoise responses to pile driving at the Horns Rev II offshore wind farm in the Danish North Sea. Final report to DONG Energy. Husum, Germany, BioConsult SH.
3. Carstensen, J., Henriksen, O. D., and Teilmann, J. (2006). Impacts on harbour porpoises from offshore wind farm construction: Acoustic monitoring of echolocation activity using porpoise detectors (T-PODs). Mar.Ecol.Prog.Ser. 321, 295-308.
4. Diederichs, A., Brandt, M. J., and Nehls, G. (2009) Auswirkungen des Baus des Umspannwerks am Offshore-Testfeld „alpha ventus" auf Schweinswale. Husum, Germany, BioConsult SH.
5. Edrén, S. M. E., Andersen, S. M., Teilmann, J., Carstensen, J., Harders, P. B., Dietz, R., and Miller, L. A. (2010). The effect of a large Danish offshore wind farm on harbour and grey seal haul-out behaviour. Mar.Mammal Sci. In press.
6. Lucke, K. Potential effects of offshore windfarms on harbour porpoises - the auditory perspective. Talk at Pile driving workshop at the European Cetacean Socity meeting, Stralsund, 21th March 2010. 2010.
7. Madsen, P. T., Wahlberg, M., Tougaard, J., Lucke, K., and Tyack, P. L. (2006). Wind turbine underwater noise and marine mammals: Implications of current knowledge and data needs. Mar.Ecol.Prog.Ser. 309, 279-295.
8. Thompson, D., Lusseau, D., Barton, T., Simmons, D., Rusin, J., and Bailey, H. (2010). Assessing the responses of coastal cetaceans to the construction of offshore wind turbines. Marine Pollution Bulletin In press.
9. Tougaard, J., Carstensen, J., Bech, N. I., and Teilmann, J. (2006) Final report on the effect of Nysted Offshore Wind Farm on harbour porpoises. Annual report to EnergiE2. Roskilde, Denmark, NERI.
10. Tougaard, J., Carstensen, J., Teilmann, J., Skov, H., and Rasmussen, P. (2009). Pile driving zone of responsiveness extends beyond 20 km for harbour porpoises (Phocoena phocoena, (L.)). J.Acoust.Soc.Am. 126, 11-14.
11. Tougaard, J., Carstensen, J., Wisz, M. S., Teilmann, J., Bech, N. I., and Skov, H. (2006) Harbour porpoises on Horns Reef in relation to construction and operation of Horns Rev Offshore Wind Farm. Technical report to Elsam Engineering A/S. Roskilde, Denmark, National Environmental Research Institute.
12. Tougaard, J., Henriksen, O. D., and Miller, L. A. (2009). Underwater noise from three offshore wind turbines: estimation of impact zones for harbour porpoises and harbour seals. J.Acoust.Soc.Am. 125, 3766-3773.
13. Tougaard, J., Scheidat, M., Brasseur, S., Carstensen, J., Petel, T. v. P., Teilmann, J., and Reijnders, P. Harbour porpoises and offshore development: increased porpoise activity in an operational offshore wind farm. Proceedings of the 24th conference of the European Cetacean Society. 63. 2010. Stralsund, Germany, European Cetacean Society.

Each wind farm is unique. The number and arrangement of turbines and the physical characteristic of the site (e.g. sediment type, water depth) vary considerably between projects. They also occur in areas with different populations and densities of marine mammals. Different foundation types require different construction operations producing different types and levels of noise, and levels of turbation or pollution. These factors all have implications for environmental impact and underline the need for a case by case evaluation of projects until a more general understanding of effects is available.

4.2.1 Construction

Among the methods currently used for construction, there is little doubt that pile driving constitutes the single most important source of impact and hence is treated separately in this section. The majority of offshore turbines are monopiles. The foundation is usually a steel tube of 2 to 5 m in diameter (with larger diameter piles being planned for future farms) which is driven into the seabed. Occasionally, alternative constructions such as tripod, jacket or gravity foundations are used. Piles are driven into the bottom by some thousand strokes of strong hydraulic hammers, produced at a rate of 30–60 pulses per minute. The ramming operation lasts from less than one hour to a few hours per pile, depending on the seabed type. The levels of noise emissions depend on a variety of factors including pile dimensions, seabed characteristics, water depth, as well as impact strengths and duration (Diederichs *et al*., 2008).

4.2.1.1 Pile driving-cetaceans

Studies were undertaken during the construction phase of both **Horns Reef I** and **Horns Reef II windfarms** in the Danish North Sea (Tougaard *et al*., 2009; Brandt *et al*., 2009). Both studies measured the acoustic activity of harbour porpoises using passive acoustic detectors (T-PODs) located within the windfarm sites and at stations situated at various distances from the piling events. Both studies demonstrated a decrease in acoustic activity following an individual pile driving event at all stations, including stations located up to 20–25 km from the piling event. The duration of the impact was assessed differently in the two studies and thus may not be directly comparable. For **Horns Reef I** the impact persisted for up to c.6 hours following the completion of an individual pile driving (Tougaard *et al*., 2009), whereas longer-term impacts of up to c.48 hours were detected at **Horns Reef II** (Brandt *et al*., 2009). These results were corroborated by a T-POD study undertaken at the **Alpha Ventus** test field in the German Bight, which demonstrated an effect extending to c. 20 km from the windfarm site, and lasting for 1–2 days after the completion of each individual pile driving event (Diederichs *et al*., 2009). The large impact area was confirmed by aerial surveys conducted before and during pile driving (Lucke, 2010). A smaller study in Moray Firth, Scotland (**Beatrice offshore wind farm**) demonstrated a decrease in acoustic activity of harbour porpoises and also dolphins (bottlenose dolphins and common dolphins) during the month when pile driving was undertaken, compared with periods without pile driving (Thompson *et al*., 2010). This study did not evaluate the effects of individual pile driving events and the temporal extent of the impact of each pile driving was thus not established.

The study at **Beatrice** had only two stations, one very close to the piling site, the other 40 km away. No reduction in the acoustic activity of small cetaceans was observed at the far station, indicating that the extent of the impact zone was less than 40 km (Thompson *et al*., 2010). There seems little doubt that pile driving of turbine foundations affects the behaviour of harbour porpoises at distances of at least 25 km from the piling site (Tougaard *et al*., 2009; Brandt *et al*., 2009; Diederichs *et al*., 2009). To date, the extent of the impact zone is thus unknown, but among other factors is likely to be related to the emitted noise energy, which is strongly correlated with pile diameter (Betke, 2010). The piles used at **Beatrice** are among the smallest at 1.8 m in diameter, followed by **Alpha Ventus** at 2.5 m and Horns **Reef I and II** at c. 4 m.

While the existence of a behavioural reaction to pile driving noise is well documented for porpoises (i.e. a reduction in echolocation clicks recorded), no work so far has addressed the important questions of what the nature of this behavioural reaction is, and what the consequences may be for the long-time survival of individuals. It is thus relevant to elucidate for example the energetic consequences of the disturbance. Pile driving can disturb animals during their feeding activities, and therefore the degree to which their food intake, and ability to nurse calves, declines during the construction period will determine the true energetic cost of the impact. Even though the disturbance itself, i.e. a single pile driving event, is fairly short term (in the order of maximum 2 hours), it may take 1–2 days following an individual pile driving event before porpoises gradually return to the impact area. However this depends on the number of foundations being piled, and also the intervals between piling.

4.2.1.2 Pile driving-seals

Only one study has directly addressed the impact of pile driving on seals. This study was conducted during the driving (by vibration and not impact driving) of sheet piles in connection to the installation of gravitational foundations at **Nysted offshore windfarm** in the Baltic Sea (Edrén *et al.*, 2010). Daily counts of hauled out seals made by remotely operated video cameras showed that 20–60% fewer grey and harbour seals hauled out on days when pile driving was conducted, compared with days without piling. Furthermore, the proportion of the seals in the region which hauled out on the nearby sandbank during the harbour seal pupping period in July (coinciding with pile driving) was significantly lower than both the preceding year and the following year. The most likely explanation is that seals were partly displaced to other haulout sites in the region during pile driving (Edrén *et al.*, 2010). Construction coincided with the outbreak of a phocine distemper epizootic. However, the harbour seal population in the western Baltic was not severely affected (Harkonen *et al.*, 2006) and because all haulout sites in the management area were surveyed, this additional factor was taken into account in the analysis.

Research undertaken on the **Egmond aan Zee offshore windfarm** fitted seals with satellite-relayed data loggers (SRDLs), and results indicated an effect from pile driving. During the construction period seals did not approach within 40 km of the windfarm area, whereas they were recorded within the windfarm area both before and after construction.

4.2.1.3 Acute damage from pile driving noise

Noise levels emitted during pile driving are very high, with sound pressures reaching 200 dB re. 1 μPa peak-peak at 100 m and sound exposure levels of single pulses reaching 180 dB SEL 100 m from the foundation. Such high levels have the potential to inflict temporary or permanent damage to the auditory system of marine mammals (Nachtigall *et al.*, 2003; Kastak *et al.*, 2005; Finneran *et al.*, 2005; Lucke *et al.*, 2009). There are no commonly adopted exposure criteria for marine mammals and thus no consensus on which exposure levels are considered safe. The criteria suggested by Southall *et al.*, 2007 are based on permanent threshold shifts (PTS) and levels are thus higher than what others have suggested. Nevertheless, modelling of cumulated sound exposure over the duration of a single pile driving event suggests that levels sufficient to elicit PTS could be reached for both seals and porpoises at distances of around 1 km from the piling site (Brandt *et al.*, 2009). For this reason mitigation measures in the form of ramp up (soft start) procedures and use of acoustic deterrent devices (pingers and seal scarers) immediately prior to piling have been introduced in order to deter animals out of the impact area before piling commences.

The exposure criteria of Southall *et al.* (2007) did not include information about harbour porpoises as this was not available at that time. However, recent results indicate that harbour porpoises may be more susceptible than other odontocetes tested and have significantly lower thresholds for eliciting TTS (temporary threshold shift, Lucke *et al.*, 2009).

4.2.1.4 Other construction activities

During the entire construction phase at **Horns Reef I**, **Horns Reef II** and **Nysted offshore wind farms** there was a pronounced general decrease in abundance of harbour porpoises (Carstensen *et al.*, 2006; Tougaard *et al.*, 2006b; Brandt *et al.*, 2009). However, no attempts were made to assess the effects of other construction activities, which included acoustic bottom profiling, dredging, deposition of boulders for scour protection and installation of turbines. The disturbance caused by the installation of gravitational foundations without associated pile driving is thus not known. Neither have any studies documented effects of ship noise (due to increased boat traffic associated with construction) in general on the abundance and behaviour of harbour porpoises.

4.2.2 Operation

Operational effects of offshore windfarms on harbour porpoises have been studied in three wind farms: **Horns Reef I** (Blew *et al.*, 2006; Tougaard *et al.*, 2006b), **Nysted** (Blew *et al.*, 2006; Tougaard *et al.*, 2006a) and **Egmond aan Zee** (Tougaard *et al.*, 2010). As these three windfarms are dissimilar in a number of characteristics, it would be expected that results and conclusions may differ between impact studies.

The **Horns Reef I offshore wind farm** is located in shallow waters in the Danish North Sea and consists of 80 turbines mounted on monopile foundations. Studies undertaken using T-PODs (Tougaard *et al.*, 2006b) monitoring porpoise acoustic activity before (baseline), during and after construction showed a clear decrease in acoustic activity inside the windfarm site during the construction phase. This was followed by a full recovery to baseline levels during the first year of operation. The results of this study were subsequently supported by a second fine-scale study by Blew *et al.* (2006) where a possible gradient in acoustic activity across the edge of the wind farm was investigated using T-PODs during the second year of operation. Results from Blew *et al.* (2006) suggested no evidence of such a gradient.

The **Nysted offshore wind farm** is located in the Baltic Sea in an area with comparatively low harbour porpoise abundance. It consists of 72 turbines mounted on gravitational foundations. Tougaard *et al.* (2006a) compared porpoise acoustic activity using T-PODs inside the windfarm site with a reference area located 10 km away. Data from this study showed a significant decrease in acoustic activity (and hence possibly porpoise abundance) during construction in both the windfarm and the reference area. During the second year of operation (2005) the acoustic activity in the reference area had attained baseline levels whereas acoustic activity inside the windfarm site was still significantly below baseline. However, in contrast to this are the results of a second study by Blew *et al.* (2006) where a gradient in porpoise acoustic activity (and abundance) was investigated by placing a number of acoustic loggers (T-PODs) inside and immediately outside the wind farm. This study did not demonstrate a gradient in acoustic activity (and possibly abundance) across the edge of the wind farm. It should be noted though that both studies were only partially overlapping in time (Blew *et al.*, 2006 conducted in 2005–2006) and they were looking at porpoise acoustic activity/abundance at two different scales (possible gradient over a few hundred meters vs. difference to a reference area 10 km away). Underwater noise measurements from Nysted did not indicate noise levels or spectral properties significantly different from what has been measured in other offshore wind farms (110 dB re. 1 µPa rms @ 100 m, dominant frequency 135 Hz, Blew *et al.*, 2006).

The **Egmond aan Zee offshore wind farm** is located in the Dutch North Sea and consists of 36 turbines mounted on monopile foundations. A study using T-PODs located inside the windfarm site and at two nearby reference (or control) sites, reported that after construction, i.e. during the operational period, a significant increase in harbour porpoise acoustic activity was noted inside the windfarm site relative to baseline levels (Tougaard *et al.*, 2010). The underlying cause of this increased acoustic activity (and possibly abundance) inside the operating windfarm site is unknown. It may be related to increased prey availability due to the artificial reefs created by the foundations or it may simply be due to the windfarm site providing shelter from other disturbing factors; as ships and trawling are not allowed inside the windfarm site.

4.3 Overview of each country's guidelines on monitoring and mitigation

National and international guidelines and regulations exist for monitoring and mitigation of the effects of windfarms. Recommendations, guidelines and regulations with relevance to effects on the environment of underwater noise and/or offshore windfarms, have been prepared by many international fora, such as the European Commission, the US Marine Mammal Commission, OSPAR, UNCLOS, CMS, ASCOBANS and IWC. They are relevant given that they can, are, or should be taken up at the national level.

4.3.1 International recommendations, guidelines, regulations

4.3.1.1 EIA directive

The European EIA Directive on Environmental Impact Assessment of the effects of projects on the environment (Directive 85/377/EEC 1985; amended 1997/2003) sets out rules on what information an EIA must provide.

4.3.1.2 European habitats directive

The Habitats Directive is relevant in the framework of offshore windfarms in several aspects:

1) For species listed in Annex II of the Directive (harbour porpoise, bottlenose dolphin, harbour, grey and ringed seal), Member States have to establish Special Areas of Conservation (SAC) (Article 6).

All cetaceans and some seal species are listed in Annex IV (Animal and Plant Species of Community Interest in Need of Strict Protection), and the grey, harbour and Baltic ringed seal Phoca *hispida botnica* in Annex V (Animal and plant species of Community interest whose taking in the wild and exploitation may be the subject of management measures) of the European Commission's Habitats Directive. Under Article 12 Member States shall take the requisite measures to establish a system of strict protection for the animal species listed in Annex IV(a) (all cetaceans, the Samaii ringed seal, *Phoca hispida* ssp. *Saimensis*, and the Mediterranean Monk seal *Monachus monachus*) in their natural range, prohibiting: (a) all forms of deliberate capture or killing of specimens of these species in the wild; (b) deliberate disturbance of these species, particularly during the period of breeding, rearing and migration; and (d) deterioration or destruction of breeding sites or resting places.

2) The Habitats Directive also requires that Member States shall undertake surveillance of the conservation status of species of Community interest, with the aim to maintain or restore species at a favourable conservation status (FCS). The conservation status of species will be taken as 'favourable' when: (a) population dynamics data on the species concerned indicate that it is maintaining itself on a long-term basis as a viable component of its natural habitat; (b) the natural range of the species is neither being reduced nor likely to be reduced for the foreseeable future; and (c) there is, and will probably continue to be, a sufficiently large habitat to maintain its populations on a long-term basis.

These Articles are subject to interpretation, and have indeed been interpreted differently among Member States of the European Union. Some Member States have prepared interpretation manuals that clearly outline the views of their Government. It is not possible to present a comprehensive overview detailing how the Habitats Directive has been interpreted in the development of offshore wind energy production, nor is it opportune given that it is in some cases more a political issue than a scientific one. Some examples of the different interpretation are given below, and in Table 3.

As Article 6 specifically requires Member States to take appropriate steps to avoid disturbance of protected species within SACs, some Member States have *a priori* excluded offshore windfarm projects in SACs - even SACs not specifically established for marine mammals. Other Member States have not *a priori* excluded offshore windfarms in SACs.

The protection of cetaceans from the impact of anthropogenic noise can form part of the strict protection awarded to them. There is increasing consensus on the view that noise should be considered a form of pollution and as such thus already covered in general terms in current international legislation regulating the emission of energy into the marine environment. According to this generally accepted view deliberate or incidental emission of noise is clearly an issue in cases where it would likely be significant in relation to the objectives of the Directive, which include the *maintenance of the protected species at a favourable conservation status*.

4.3.1.3 European integrated maritime policy

One of the products of the Integrated Maritime Policy, launched by the EC in October 2007, is the *Roadmap for Maritime Spatial Planning: achieving common principles in the EU*. According to Gilliland and Laffoley (2008), marine spatial planning is an essential tool for delivering an ecosystem approach if based on a clear set of principles with a sustainable development purpose.

One of the applications mentioned in the EU Directive 2002/49/EC for noise in air is generating strategic noise maps, which are useful for spatial planning in relation to sound exposure. As recognized and suggested by the Task Group 11 in the process of developing a framework for underwater noise for the implementation of the MSFD, noise mapping on a regional basis should be used to analyse noise budgets of the oceans and regional sea areas. This can be done by acoustic measurements and modelling based on data and information gained through the application of the suggested indicators for descriptor 11 (see below).

4.3.1.4 Marine strategy framework directive

The MSFD requires Member States to develop marine strategies that apply 'an ecosystem-based approach to the management of human activities while enabling a sustainable use of marine goods and services, priority should be given to achieving or maintaining Good Environmental Status (GES) in the Community's marine environment, to continuing its protection and preservation, and to preventing subsequent deterioration'.

One of the main objectives is to achieve a GES for European marine waters by 2020. For achieving GES eleven descriptors were provided, among which descriptor 11 which states that *introduction of energy, including underwater noise, is at levels that do not adversely affect the marine environment*. Criteria to attain this are currently being prepared. Guidance on methodological standards will follow soon. The draft version (March 2010) of the criteria for descriptor 11 reads:

Indicator 1: Distribution in time and place of loud, low and mid frequency impulsive sounds

Proportion of days within a calendar year, over areas of a determined surface and their spatial distribution, in which anthropogenic sound sources exceed either of two levels, [159–183] dB re 1μPa²s (i.e. measured as Sound Exposure Level, SEL) or [180–224] dB re 1μPa$_{peak}$ (i.e. measured as peak sound pressure level) when extrapolated to one metre, measured over the frequency band 10 Hz to 10 kHz (11.1).

Indicator 2: Continuous low frequency sound

Ambient noise level, as measured by a statistical representative sets of observation stations, where noise within the 1/3 octave bands 63 and 125 Hz (centre frequency) (average noise level in these octave bands over a year) (11.2).

4.3.1.5 Agreement on the conservation of small cetaceans of the Baltic, North East Atlantic, Irish and North Seas (ASCOBANS)

ASCOBANS has prepared a number of Resolutions with recommendations on underwater noise. In the Resolution on *Adverse Effects of Underwater Noise on Marine Mammals during Offshore Construction Activities for Renewable Energy Production*, adopted by ASCOBANS Parties in 2009, Parties recommend that:

- A strategic approach in marine renewable developments should be taken;
- The precautionary approach should be followed;
- Guidelines should include an appropriate location of devices, measures for avoiding construction activities with high underwater noise source levels during the periods of the year with the highest densities of small cetaceans, measures for avoiding construction activities with high underwater noise source levels when small cetaceans are present in the vicinity of the construction site, measures for alerting small cetaceans to the onset of potentially harmful construction noise, and technical measures for reducing the sound emission during construction works.

ASCOBANS further promotes the development of effective mitigation measures, guidelines and technological adaptations, an assessment of the effectiveness of guidelines, a continued monitoring of effects and the exchange of information.

4.3.1.6 Convention on migratory species (CMS)

CMS Resolution 9.19 on adverse anthropogenic marine/ocean noise impacts on cetaceans and other biota, adopted by the 9th Meeting of the Conference of the Parties in 2008, noted that in any case of doubt the precautionary approach should be applied. Parties are further encouraged to facilitate:

- Regular collaborative and coordinated temporal and geographic monitoring and assessment of local ambient noise (both of anthropogenic and biological origin);
- The compilation of a reference signature database, to be made publicly available, to assist in identifying the source of potentially damaging sounds;
- Characterisation of sources of anthropogenic noise and sound propagation to enable an assessment of the potential acoustic risk for individual species in consideration of their auditory sensitivities;
- Studies reviewing the potential benefits of "noise protection areas", where the emission of underwater noise can be controlled and minimized for the protection of cetaceans and other biota.

4.3.2 National guidelines on monitoring and mitigation

There are important differences in the monitoring and mitigation guidelines among parties. Sometimes guidelines are clearly described in nationally accepted documents, and sometimes they are issued on a project basis. Some guidelines and conditions are described for EIA requirements, and others are focused on the construction and operational phases of offshore windfarm developments.

The Group considered that it was not in a position to make an overview of national guidelines on monitoring and mitigation relevant to offshore windfarm construction and operation, and even less in a position to compare and review these. The number of relevant guidelines is often very large, and it was not opportune to make a selection, which inevitably would be incomplete and biased.

Some relevant documents, including national guidelines, are for Germany: BSH (2007a; 2007b; 2008), for the UK: Cefas (2004), DEFRA (2005), JNCC (in consultation), and for The Netherlands: Prins *et al*. (2008).

Examples of national guidelines for mitigation and prevention are taken up in the table below (Table 3). They illustrate differences in the guidelines for the construction of offshore windfarms applicable in different nations' waters.

Table 3. Examples of guidelines in some countries for preventing and/or mitigating negative effects on marine mammals in the framework of the construction of offshore windfarms.

	Use of Acoustic Deterrent Devices required during pile driving	Marine mammal observers required before and during pile driving	Seasonal restrictions for pile driving	Soft start – ramp up procedure for pile driving	Offshore windfarms in NATURA 2000 areas allowed?	Examples of other guidelines
Belgium	Yes, taken up in the permit	No	Yes, but only in the advice: no piling between 1 January and 30 April	Yes, taken up in the permit, and not standardized	Not *a priori* forbidden, but currently no NATURA 2000 areas are considered for windenergy production	
Denmark	Yes	No	Currently not	Yes, but not standardized	Yes, conditions apply	
Germany	Yes	No	Currently not	Yes	No, since the establishment of marine spatial planning regulations	Noise limitation from 750 m from the piling onwards: 160 dB SEL and 190 dB SPL*
The Netherlands	Yes, general guideline	No	Yes, no piling between 1 January and 1 July		Not *a priori* forbidden	There cannot be more than one construction activity in which piles are driven ongoing at any time
United Kingdom	Case by case basis as a condition of the consent	Yes, and/or real-time acoustic monitoring	Yes, in relation to spawning fish (some of which are prey items)	Yes	Not *a priori* forbidden	Depending on work being undertaken, requirement for amonitoring zone prior to piling. The size of which is defined by the area over which injury may occur

***: The German Federal Environment Agency (UBA) has defined `injury` as Temporal Threshold Shift (TTS) based on data provided by Lucke *et al*. (2009). A threshold consisting of a dual criterion of 160 dB re 1 mPa^{2}· s SEL (Sound Exposure Level) and 190 dB re 1 µPa SPL (Sound Peak Pressure Level) should not be exceeded at a distance of 750 meters around the piling site. The threshold is based on a TTS found in a harbour porpoise at 164 dB re 1mPa2 · s SEL and 199 dB re 1 µPa SPL. Thus the chosen values include some safety adjustment. This threshold is part of the licence, and therefore legally binding.**

4.4 Present information on the distribution and scale of wind farm developments in ICES waters

An overview of the current distribution of windfarms in the North-East Atlantic can be obtained from the OSPAR Database on Offshore Wind-farms. This database is annually updated by the OSPAR Working Group on the Environmental Impact of Human Activities (EIHA), and is available on the OSPAR website (www.ospar.org). Figure 1 presents an overview of the operational, authorized and planned windfarms in the OSPAR maritime area (correct as of July 2009). An overview of the status of offshore wind farms in the Baltic Sea is presented in Figure 2. Within western and northern European waters, by the end of 2009, 36 windfarms were operational in 9 countries, with a total of 796 wind turbines (see Table 4).

An overview of plans for future possible developments of offshore windfarms is not presented here, given that this is rapidly evolving. The OSPAR database or the European Wind Energy Association regularly provides updates on these. An assessment of the potential of offshore windfarms (EEA 2009) indicated that it is restricted by, among others, shipping lanes, anchoring areas, military areas and oil and gas platforms. Apart from expanding to areas where little or no projects exist, such as the USA and Scotland (see Figure 3a), one development, however, is obvious. Windfarms will be built further offshore and in deeper waters, as can be illustrated with the Round 3 proposed sites for wind farm leasing in UK waters (Figure 3b).

Table 4. Operational offshore wind-farms in Europe in 2009. See www.ospar.org or www.ewea.org for detailed information on individual windfarms. Source: EWEA. UK – United Kingdom, DK – Denmark, SE – Sweden, NL – The Netherlands, DK – Denmark, BE – Belgium, IE - Ireland, FI - Finland, and NR – Norway.

COUNTRY	UK	DK	SE	NL	DK	BE	IR	FI	NR	TOTAL
No of farms	12	9	5	2	4	1	1	1	1	36
No of turbines	287	305	75	98	9	6	7	8	1	796

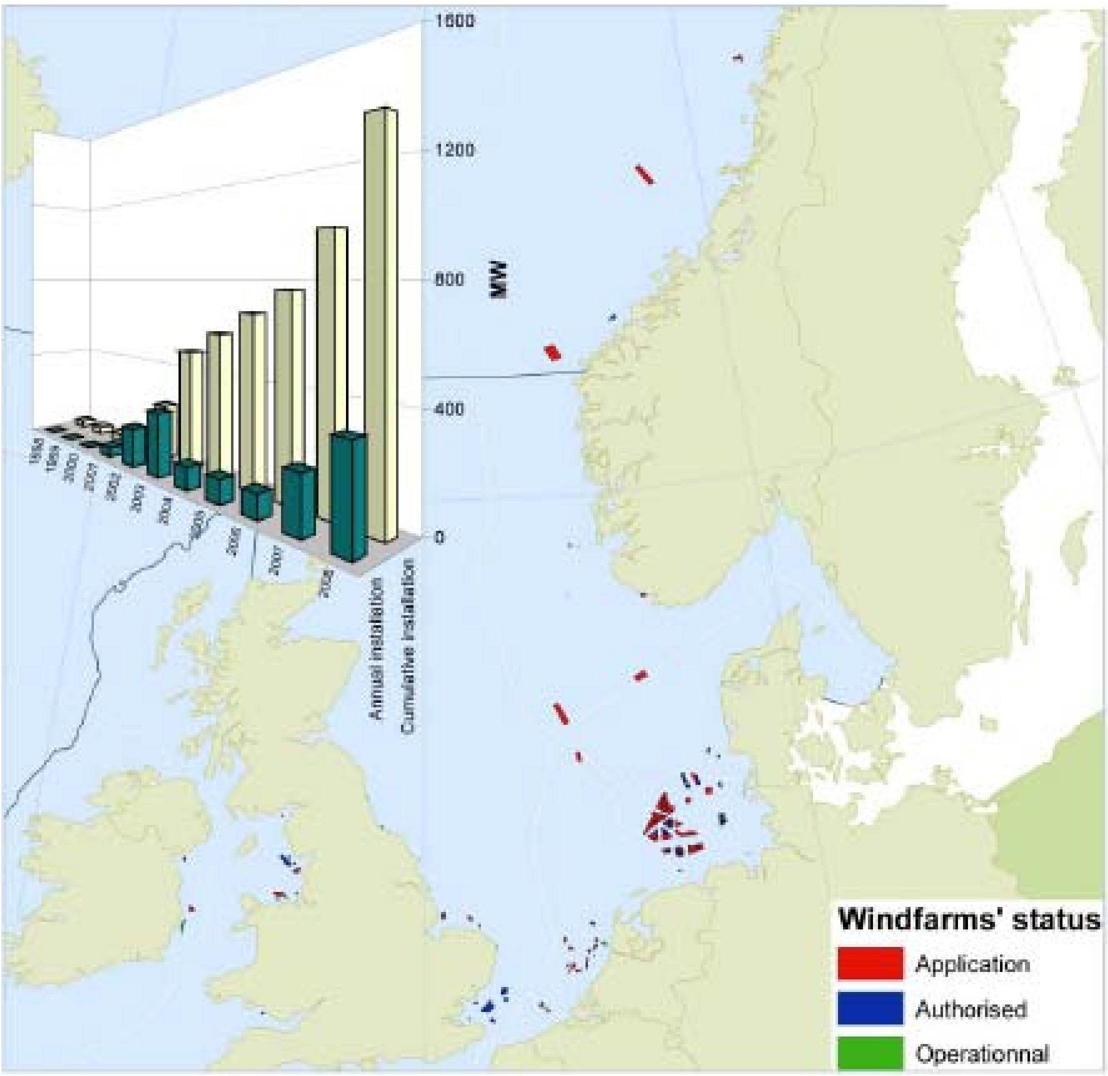


Figure 1. Location of operational authorized and planned wind-farms in the OSPAR maritime area (July 2009). The inset shows trends in the development of wind power since the late 1990s. Source: OSPAR database on offshore wind-farms. OSPAR website, accessed 15 April 2010 (www.ospar.org).

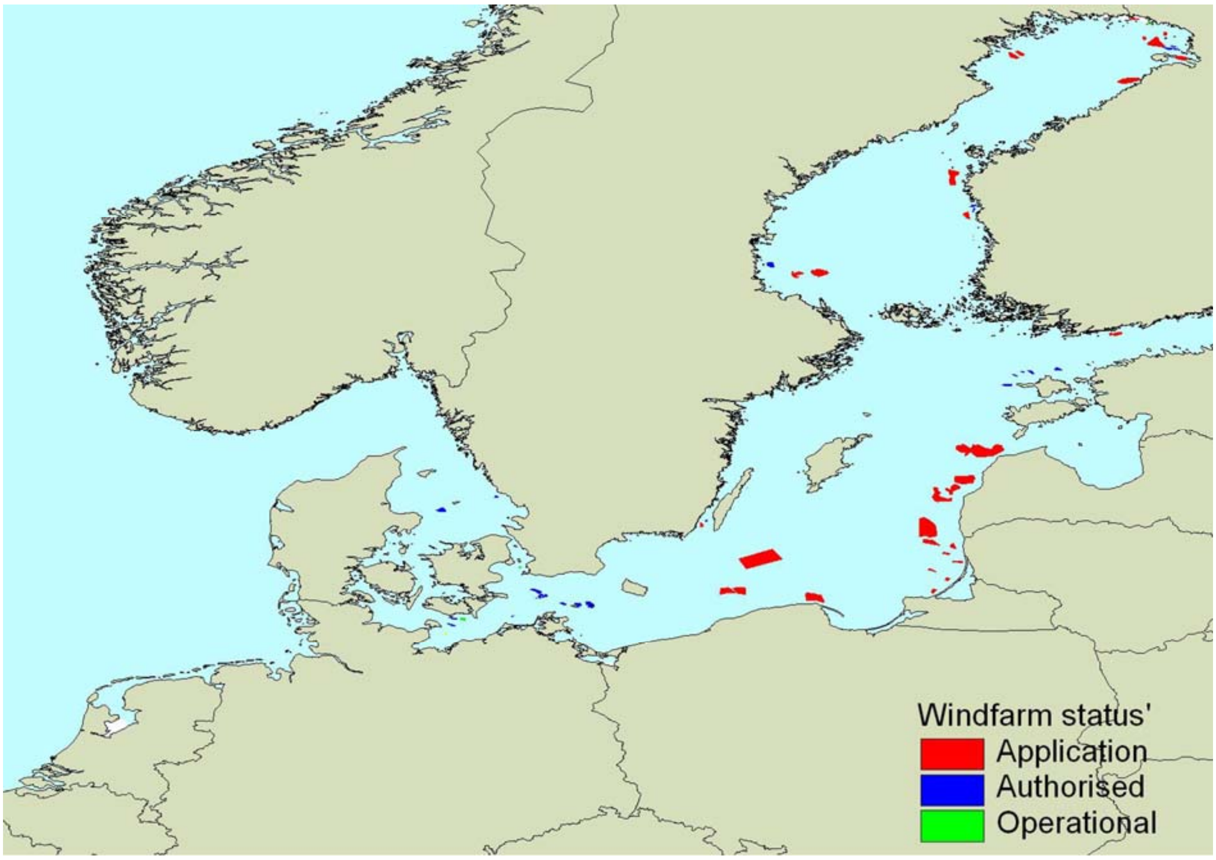


Figure 2. Location of proposed, planned and operation wind farms within the Baltic Sea. Data provided by HELCOM.

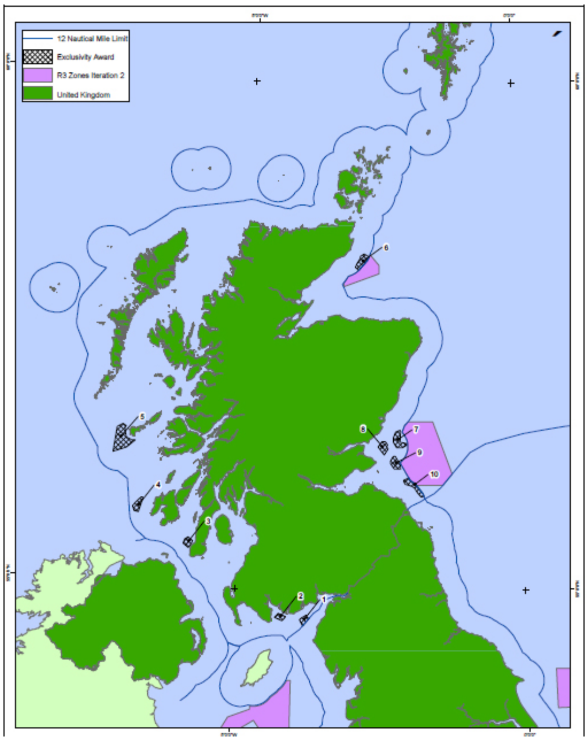


Figure 3a. Location of exclusivity awarded areas for windfarm construction in Scottish waters, 2009. Source: The Crown Estate (www.thecrownestate.co.uk).

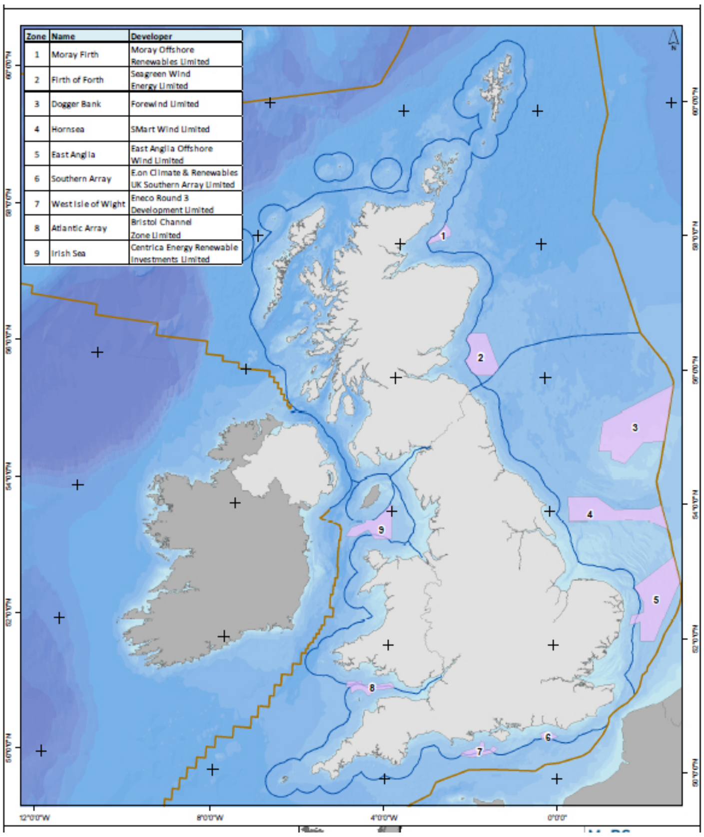


Figure 3b. Location of the so-called round 3 wind-farms planned in UK waters, till 2020. Source: The Crown Estate (www.thecrownestate.co.uk).

Table 5. A list of national sources for obtaining most recent information.

	WEBSITE
OSPAR	http://rod.eionet.europa.eu/obligations/448
Baltic Sea HELCOM	www.helcom.fi
UK	http://www.thecrownestate.co.uk
INTERREG III Baltic Sea	http://www.windenergy-in-the-bsr.net/index.html
Belgium	www.mumm.ac.be
Poland	www.ptew.pl
Sweden	http://www.energimyndigheten.se/
Denmark?	http://www.ens.dk/EN-US/SUPPLY/RENEWABLE-ENERGY/WINDPOWER/OFFSHORE-WIND-POWER/Sider/Forside.aspx

4.5 Provide advice and recommendations on monitoring and mitigation schemes

Monitoring in connection to offshore wind farms can be divided into two phases: baseline data collected prior to construction (often as part of the EIA process) and impact data collected during construction and operation of the offshore wind farm.

4.5.1 Baseline monitoring

The aim of baseline monitoring can be twofold. First and foremost it is to establish abundance patterns of marine mammals in the proposed construction area and thus provide important information for the decision process of the EIA. Second the baseline monitoring should collect baseline data for later impact studies, given that such are undertaken. Some countries (e.g. Germany) always require baseline data to be collected during the EIA, whereas most other countries only require collection of new data if other relevant data are not available. With regard to baseline monitoring, the Group advises the following:

1) With regard to wind farm developments, establishment of means for efficient dissemination of results of common interest and means of making previous EIA reports and previously collected baseline data available for subsequent studies and assessments.

The risks and potential impacts of many offshore developments are similar. It is obviously inefficient that EIAs are carried out entirely independent from each other because this will result in the duplication of effort and repetition of the same mistakes. The Aarhus Convention of the EU (ECE/CEP/43: http://ec.europa.eu/environment/eia/full-legal-text/aarhus_en.pdf) along with the Convention on Biodiversity (http://www.cbd.int/doc/legal/cbd-un-en.pdf) recognize this. The latter set up a Clearing House Mechanism to promote the sharing of information. There are currently two limitations to using this approach: awareness of the existence of data and the availability of publically owned data. While the release of commercially sensitive data has to be subject to delay, this needs to be balanced against the benefits of its use in evaluating other applications and thus also data collected by private companies should be made available. Shifting the balance towards more rapid dissemination could reduce the amount of new information, and experimental studies, required and, improve the assessment of new projects.

2) Encourage multinational studies and encourage management decisions regarding offshore wind farms to be based on appropriate populations and/or management units for the relevant marine mammal species, irrespective of national borders.

Consent for development, and assessment of impacts are matters for individual Governments. Many marine mammals are wide ranging and occur in populations that regularly move and mix across national boundaries. This means that assessments of impacts cannot be carried out entirely within territorial boundaries, and that consents given by one Government can affect the acceptability of potential developments within a neighbouring jurisdiction. Some recognition of this is essential in decision-making. In some cases coordination can occur within existing frameworks, as for example the Common Wadden Sea Agreement and ASCOBANS. In other cases new fora for coordination must be created. Increased cooperation between EU Member States will be required by the Marine Strategy Framework Directive through the application of an ecosystem-based approach to the management of human activities.

3) As the development of offshore wind farms extends further offshore and into new waters, monitoring should be extended to include all commonly occurring marine mammal species and marine mammal species of particular concern.

Most impact studies and assessments so far have focused on harbour porpoises and harbour seals. These are the most 'accessible' species: they are the most common species in the coastal waters where wind farms are currently being constructed, methods to study them have been well developed, and captive animals are accessible. However, as wind farms are planned further offshore and extend into new waters, such as the English Channel, the northern North Sea and the Baltic Proper, other species become increasingly important and should be included in assessments and impact studies. For the North Sea this includes species such as the white-beaked dolphin, common dolphin, minke whale and killer whale, while for the Baltic Proper the ringed seal becomes relevant. Offshore wind farm developments on the east coast of the US and Canada will possibly interact with other species, most importantly right whales and belugas.

4) Geographical location of offshore wind farms should consider the distribution of marine mammals throughout the year, time of day and under typical weather and hydrographical conditions.

For most species of marine mammals the information available on distribution comes from limited sources and there is thus in several cases a strong bias in the information towards times of the year and weather conditions where for example surveys can be conducted. One example is SCANS-II survey which assessed the abundance of harbour porpoises throughout the North Sea. The results of this survey regarding the distribution of porpoises reflect a single moment in time (summer 2005), and they do not provide for information about migration, and for instance on the distribution of porpoises during winter. Also, important shifts in the distribution of marine mammals have occurred throughout recent years, and therefore regular monitoring activities should be undertaken with appropriate methods. Evidently, one single method cannot cover all species, so the most appropriate method must be used for each of the species in question.

5) Increase efforts to develop common measurement standards for both noise and marine mammal abundance.

In clear accordance with the identified need for increased availability of data from different studies, there is a need for standardized methods for collection and analysis of data. Furthermore there is evidently a need for common or at least compatible data storage formats across projects and countries. Surveys with ships and airplanes are already covered to a large degree by de facto standards through the very widespread use of distance sampling methods and analysis by means of the associated software (Buckland *et al.*, 2001). In contrast to this are stationary visual surveys (from land or fixed platforms) where no standards are available.

Some work has been conducted in the field of common standards for using passive acoustic monitors (T-PODs and C-PODs) (Teilmann *et al.*, 2001; Teilmann *et al.*, 2002; Anon., 2009), but there is clearly a need for more work in this direction. Currently there is de facto only one instrument available for high-frequency species (the C-POD), but competing designs are beginning to appear, which increases the need for intercalibration and common standards. For low-frequency species as well as noise in the range up to 20 kHz a range of dataloggers are available, making the need for standards urgent.

4.5.2 Impact monitoring

Impact monitoring deals with determining actual effects of construction activities and/or habitat loss connected to the operating offshore windfarm. In addition it also includes quantifying the source of the impact, if this is known. The most prominent example of the latter would be measurements of underwater noise from construction activities and operating turbines.

Significant evidence has been collected on the effects of underwater noise due to pile driving (see Section 4.2), and there is little doubt that this activity can have significant negative effects on marine mammals. Therefore a focus is put on recommendations concerning this activity. Comparatively less is known about the levels and possible impact of underwater noise in general during the construction phase, and as such this also should receive attention.

Next to the recommendations related to the direct effects of underwater noise on marine mammals, there are possibly indirect effects, through effects on the main prey species of marine mammals.

The Group recommends to:

6) Increase the effort to characterize sources of underwater noise related to the construction and operation of offshore wind farms. As part of this, common standards for measurement and characterization of underwater noise should be developed.

Given the many factors influencing the underwater noise emissions and transmission, monitoring of underwater noise should be undertaken whenever there are reasons to believe that results from research at other wind farms cannot be extrapolated. It should be emphasized that at present, there is limited knowledge of the general patterns of noise generation from offshore wind turbines, meaning that emitted noise characteristics cannot be predicted. Transmission loss models for the relevant areas should be developed and used to map the predicted noise impact, based on actual noise measurements.

Underwater noise is now described in different ways, which makes it difficult to compare data. Standards should be chosen in a way to facilitate the monitoring of the effects. Standards for expressing noise have been proposed by Southall *et al.*

(2007) and de Jong *et al*. (2010). Southall *et al*. (2007) put the main focus on measures relevant to effects, whereas de Jong *et al*. (2010) put the focus on the physical description of noise. A common best practice of measuring, analysing and presenting underwater noise should be adopted, including methods to quantify the particle motion part of the sound field in addition to the pressure field which is normally the only component measured[1].

7) Develop methods to assess cumulative effects on marine mammals of the underwater noise level caused by the simultaneous construction and operation at nearby sites.

Currently a lot of data are lacking, which prevents us from assessing the impact of the construction and operation of offshore wind farms on marine mammals, both on individual animals and on populations. Effects should be assessed on a short-term and long-term level, and during the construction and operation phases of the projects. Evaluation of cumulative effects should not be limited to offshore wind farms but must include all other anthropogenic impacts in the area (such as other construction work, shipping, fishing, and oil and gas activities). Noise mapping (see Section 4.3.1.3.) could act as a tool to account for cumulative impacts of the construction and operation activities of owfs as well as other influencing noise sources.

The Group further advises to:

8) Step up research on the behaviour of marine mammals as a consequence of increased underwater noise levels, in particular on how changes ultimately affect population parameters.

Impact studies have demonstrated behavioural reactions of harbour porpoises towards pile driving noise. Although it is clear that the impact area can be extensive and extends out to at least 20 km from the piling site, the implications of this reaction for the fitness of the affected animals is unknown. It remains important to address this question and establish for example which consequences the reaction has on metabolic intake and ultimately on population parameters such as fecundity and survival. While the individual response of the animals can be measured, the impact should be assessed at the population level; the response of the animals should therefore be translated into a meaning of the effect on the (local) population. Cumulative effects on populations of marine mammals, due to the simultaneous construction and operation of different wind farms, should likewise be assessed.

9) Increase efforts to characterize fundamental properties of the auditory system of marine mammals and the way noise affects physiology and behaviour.

Assessment of the impact requires fundamental knowledge of the way marine mammals perceive and use sound. For a few species, such as bottlenose dolphins and harbour seals a great deal is known, for others such as harbour porpoises the knowledge is more limited, and for still other species such as grey seal, ringed seal, common and white-beaked dolphins and baleen whales next to nothing is known about hearing physiology. As there can be large and unexpected differences between even closely related species, it is important to have information about parameters such as

---

1 Many species of fish, in particular species without swim bladders are mainly sensitive to the particle motion part of the sound field, whereas marine mammals hear only the pressure part. Particle motion is thus primarily of interest concerning impact on fish but the possibility that marine mammals can perceive intense low frequency particle motion should not be excluded.

hearing range, critical bandwidths and TTS-susceptibility. Common for all species is that a fundamental assumption underlying the recommendations of Southall *et al*. (2007)-the loudness function, has only been described in a single mammalian species: humans. Extrapolation by means of robust models of the auditory function should be used to assess the impact on species for which limited information is available and for which it is unlikely that such information can be obtained in the near future.

Even though most odontocetes and to some degree seals have comparatively poor hearing at very low frequencies, it is also important to investigate to which degree intense low-frequency sound affects these species.

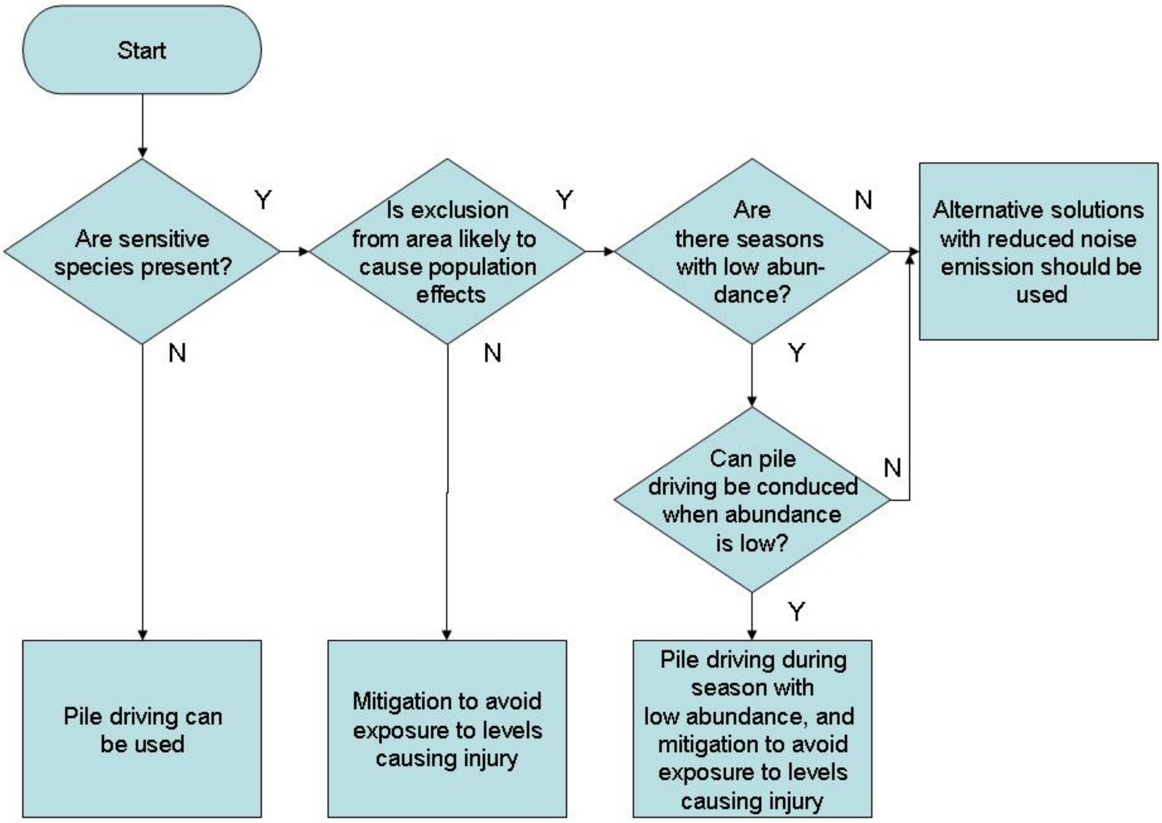
4.5.3 Mitigation

Given the number of offshore wind farm projects that are being constructed and planned, and the fact that several studies have indicated that effects of pile driving on marine mammals can occur beyond a distance of 20 kilometres from the construction site, the possible effects of this activity on marine mammal populations should be taken seriously. The group therefore considered means of reducing detrimental effects of intense noise during construction of offshore wind farms.

The decision whether to mitigate or to seek alternative solutions depends on a range of factors, most importantly the abundance of marine mammals in the area. As impacts of pile driving have been shown to extend at least 20 km from the pile driving site in case of harbour porpoises, it is relevant to consider not only abundance within the intended wind farm area but also in adjacent areas. This is particularly pertinent to constructions intended to take place immediately adjacent to protected sites such as Natura 2000 areas.

When considering reduction of impact from pile driving there is a distinction between the level of impact addressed. If a temporary exclusion from the construction site and adjacent areas impacted can be shown to be unlikely for the population in the relevant management area, then it may be appropriate to mitigate at the level of physical injury (TTS, PTS and non-auditory effects). This means that mitigation measures should ensure only that (ideally) no individuals are exposed to sound levels causing physical injury. If on the contrary, there is insufficient information available or direct concern that temporary habitat loss may affect the population, then mitigation must take place at the level of behavioural disturbance. This implies that the habitat loss should be minimized to a degree considered within acceptable levels.

The following diagram illustrates how a decision process could be organized:



In relation to mitigation the Group recommends:

10) With regard to marine mammals, to work towards common accepted tolerance limits for acute noise exposure and the development of common guidelines for mitigation in relation to pile driving.

The information regarding acute effects of underwater noise on marine mammals has increased considerably in recent years (reviewed among others by Southall *et al.*, 2007; OSPAR 2009a; 2009b) and has reached a level where it makes sense to start discussing the establishment of scientifically based tolerance limits. Such a development also falls along the lines of the requirements of the European Marine Strategy Framework Directive, which among other requires indicators for Good Environmental Status regarding underwater noise. Work in this direction is being, or has already been undertaken within different organizations, such as the US Marine Mammal Commission, ASCOBANS and ACCOBAMS. In line with recommendations of Southall *et al*. (2007), such exposure criteria should consider both unweighted and frequency weighted sound pressures as well as cumulated sound energy, both within single sounds and across multiple exposures. Exposure criteria should, as far as possible, be developed on a species by species basis.

Connected to the establishment of common tolerance limits, is the development of common guidelines for best practice and mitigation measures to be used to minimize the risk that marine mammals are exposed to sound levels exceeding the exposure criteria.

11) To undertake studies to develop better marine mammal acoustic deterrent devices, including realistic trials in the field to demonstrate their effectiveness.

One method for mitigating the risk of hearing damage from pile driving is to move vulnerable animals out of the danger zone by broadcasting aversive sounds, i.e. sounds which cause animals to move away without adding significantly to the animals' acoustic dose. If a method based on aversive signals could be developed and shown to be effective, it could have a number of advantages. As marine mammals are so difficult to detect at sea and mitigation zones are substantial, aversive signal mitigation could be more effective than current methods which relies on detecting animals within the impact zone followed by a temporary shut-down of piling. The use of deterrent devices would also allow construction to continue in poor weather conditions and at night and they should be very cost-effective. SMRU Ltd (2007) explored the potential advantages and problems of such a system and reviewed terrestrial examples where sound is used to move animals. Overall their conclusions were encouraging. They cited many examples of marine mammals moving considerable distances in response to sound. The authors mention two important caveats however. The first is that, to avoid habituation, whatever aversive signal might eventually be deployed, it should be quite different from other signals that animals might be routinely exposed to. For this reason existing acoustic deterrent devices such as fisheries pingers and "seal scarers" should not be used. By using a unique signal, which is coupled to something unpleasant (the pile driving noise that will follow) the risk of habituation is strongly reduced, as the animals are not reinforced for habituating to the signal as is the case with for example seals to seal scarers. Here the seal scarer is intended to deter animals from something they want to obtain (fish in fishing gear or in a fish farm).The other important caveat of deterrent devices is that these methods can only be relied upon once a substantial body of data has been collected to prove that they are effective on all the species of concern. These should be based on field trials in realistic field conditions, including on foraging grounds.

12) Attention should be given to improve efficient means of real-time detection of marine mammals during pile driving operations.

Visual observers and passive acoustic monitoring have been suggested as a mitigation measure during pile driving. Operators are asked to shut down the operation if marine mammals are observed inside a designated safety zone. The efficiency of such a procedure depends critically on the ability to detect the presence of marine mammals with sufficient reliability (low rate of misses, low rate of false alarms) within the entire relevant impact area (zone of injury), which could extend out to distances of several kilometres from the construction site (Gordon *et al.*, 2009).

4.5.4 Alternatives to pile driving

The most efficient way to reduce impact from widespread and extensive pile driving is to develop alternative methods for installing foundations with reduced noise emission during installation. Thus the Group recommends that:

13) Other measures than the above are taken to prevent that marine mammals are exposed to high levels of underwater noise. This includes limiting the radiated energy during pile driving and the development of alternative methods for installation.

The best approach to reducing impact from construction of offshore wind farms is to avoid pile driving altogether, such as through developing alternative methods for pile driving or the use of alternative types of foundation. This includes, but is not limited to use of gravitational foundations or suction piles, installation by water jet or by drilling, and in deeper waters use of floating platforms tethered to the seabed. Secondary solutions involve limiting the energy radiated from the pile driving into the water for example by using bubble curtains or pile sleeves (if feasible and if efficient).

4.6 References

Anon. 2009. Förderprogramm des BMU zu Forschung und Entwicklung im Bereich Windenergie. Projekt AMPOD: Anwendungs- und Auswerteverfahren für den Einsatz von T-PODs in WEA-Umweltverträglichkeitsprüfungen: Vergleichbarkeit und Entwicklung von Standardmethoden. Germany, Federal Ministry for the Environment and Reactor Safety.

Bain, D. E., and Williams, R. 2006. Long-range effects of airgun noise on marine mammals: Responses as a function of received sound level and distance International Whaling Commission IWC-SC/58E35. Cambridge, UK.

Betke, K. 2010. Messungen von Unterwasserschall beim Bau der Windenergieanlagen im Offshore-Testfeld "alpha ventus". Hamburg, ITAP.

Blew, J., Diederichs, A., Grünkorn, T., Hoffmann, M., and Nehls, G. 2006. Investigations of the bird collision risk and the responses of harbour porpoises in the offshore wind farms at Horns Rev, North Sea and Nysted, Baltic Sea, in Denmark. Status report 2005 to the Environmental Group. Hamburg, BioConsult SH.

Brandt, M. J., Diederichs, A., and Nehls, G. 2009. Harbour porpoise responses to pile driving at the Horns Rev II offshore wind farm in the Danish North Sea. Final report to DONG Energy. Husum, Germany, BioConsult SH.

BSH. 2007a. Standard - Design of offshore wind turbines. Federal Maritime and Hydrographic Agency (BSH).

BSH. 2007b. Standard - Investigations of the Impacts of Offshore Wind Turbines on the Marine Environment. Federal Maritime and Hydrographic Agency (BSH).

BSH. 2008. Standard – Ground investigations for offshore wind farms. Federal Maritime and Hydrographic Agency (BSH).

Buckland, S. T., Anderson, D. R., Burnham, K. P., Laake, J. L., Borchers, D. L., and Thomas, L. 2001. Introduction to distance sampling. Estimating abundance of biological populations Oxford University Press.

Carstensen, J., Henriksen, O.D., and Teilmann, J. 2006. Impacts on harbour porpoises from offshore wind farm construction: Acoustic monitoring of echolocation activity using porpoise detectors T-PODs. Marine Ecology Progress Series 321: 295–308.

Cefas. 2004. Guidance Note for Environmental Impact Assessment in Respect of FEPA and CPA Requirements. Centre for Environment, Fisheries and Aquaculture Science (Cefas) on behalf of the Marine Consents and Environment Unit (MCEU).

Cox, T. M., Read, A. J., Solow, A., and Tregenza, N. 2001. Will harbour porpoises (*Phocoena phocoena*) habituate to pingers? Journal of Cetacean Research and Management 3(1): 81–86.

DEFRA, 2005. Nature conservation Guidance on Offshore Windfarm Development: a Guidance Note for Developers Undertaking Offshore Wind farm Developments. Prepared by Department of Environment, Food and Rural Affairs.

Diederichs, A., Brandt, M. J., and Nehls, G. 2009. Auswirkungen des Baus des Umspannwerks am Offshore-Testfeld „alpha ventus" auf Schweinswale. Husum, Germany, BioConsult SH.

EEA. 2009. Europe's onshore and offshore wind energy potential. An assessment of environmental and economic constraints. EEA technical report No 6/2009.

Edrén, S.M.E., Andersen, S.M., Teilmann, J., Carstensen, J., Harders, P.B., Dietz, R., and Miller, L. A. 2010. The effect of a large Danish offshore wind farm on harbour and gray seal haul-out behaviour. Marine Mammal Science: In press.

Finneran, J. J., Carder, D. A., Schlundt, C. E., and Ridgway, S. H. 2005. Temporary threshold shift in bottlenose dolphins (*Tursiops truncatus*) exposed to mid-frequency tones. Journal of the Acoustical Society of America 118: 2696–2705.

Gilliland P.M. and Laffoley D. 2008. Key elements and steps in the process of developing ecosystem-based marine spatial planning. Marine Policy 32 (5): 787–796.

Harkonen, T., Dietz, R., Reijnders, P., Teilmann, J., Harding, K., Hall, A., Brasseur, S., Siebert, U., Goodman, S. J., Jepson, P. D., Rasmussen, T. D., and Thompson, P. 2006. The 1988 and 2002 phocine distemper virus epidemics in European harbour seals. Diseases of Aquatic Organisms 68: 115–130.

JNCC. In consultation. Statutory nature conservation agency protocol for minimising the risk of disturbance and injury to marine mammals from piling noise. Joint Nature Conservation Committee (JNCC), Marine Advice, Aberdeen.

de Jong, C. A. F., Ainslie, M. A., and Blacquiere, G. 2010. Measuring underwater sound: towards measurement standards and noise descriptors. TNO report TNO-DV 2009 C613. TNO.

Kastak, D., Southall, B. L., Schusterman, R. J., and Kastak, C. R. 2005. Underwater temporary threshold shift in pinnipeds: effects of noise level and duration. Journal of the Acoustical Society of America 118: 3154–3163.

Lucke, K. Potential effects of offshore windfarms on harbour porpoises - the auditory perspective. 2010. Pile driving in offshore windfarms: effects on harbour porpoises, mitigation measures and standards, European Cetacean Society meeting, Stralsund, 21st March 2010.

Lucke, K., Siebert, U., Lepper, P. A., and Blanchet, M.-A. 2009. Temporary shift in masked hearing thresholds in a harbour porpoise (*Phocoena phocoena*) after exposure to seismic airgun stimuli. Journal of the Acoustical Society of America 125: 4060–4070.

Nachtigall, P. E., Pawloski, D. A., and Au, W. W. L. 2003. Temporary threshold shifts and recovery following noise exposure in the Atlantic bottlenosed dolphin (*Tursiops truncatus*). Journal of the Acoustical Society of America 113: 3425–3429.

OSPAR. 2008. Assessment of the environmental impact of offshore windfarms. OSPAR Biodiversity Series, publication 385.

OSPAR. 2009a. Assessment of the environmental impact of underwater noise. OSPAR report 436.

OSPAR. 2009b. Overview of the impacts of anthropogenic underwater sound in the marine environment. OSPAR report 441.

Prins, T.C., Twisk, F., Van den Heuvel-Greve, M.J., Troost, T.A. and Van Beek, J.K.L. 2008. Development of a framework for Appropriate Assessments of Dutch offshore wind farms. IMARES report Z4513.

SMRU LTD. 2007. Assessment of the potential for acoustic deterrents to mitigate the impact on marine mammals of underwater noise arising from the construction of offshore windfarms. Commissioned by COWRIE Ltd (project reference DETER-01-07).

Southall, B.L., Bowles, A.E., Ellison, W.T., Finneran, J., Gentry, R., Green, C.R., Kastak, C.R., Ketten, D.R., Miller, J.H., Nachtigall, P.E., Richardson, W.J., Thomas, J.A., and Tyack, P.L. 2007. Marine Mammal Noise Exposure Criteria. Aquatic Mammals 33: 411–521.

Teilmann, J., Henriksen, O. D., and Carstensen, J. 2001. Porpoise detectors (PODs) as a tool to study potential effects of offshore wind farm on harbour porpoises at Rødsand. Roskilde, Denmark, NERI. p. 41.

Teilmann, J., Henriksen, O.D., Carstensen, J., and Skov, H. 2002. Monitoring effects of offshore windfarms on harbour porpoises using PODs (porpoise detectors). Technical report to SEAS A/S. Roskilde, Denmark, NERI.

Thompson, D. (Ed.). 2000. Behavioural and physiological responses of marine mammals to acoustic disturbance – BROMMAD, Final Scientific and Technical Report. St Andrews, UK.

Thompson, P. M., Lusseau, D., Barton, T., Simmons, D., Rusin, J., and Bailey, H. 2010. Assessing the responses of coastal cetaceans to the construction of offshore wind turbines. Marine Pollution Bulletin.

Tougaard, J., Carstensen, J., Bech, N.I., and Teilmann, J. 2006a. Final report on the effect of Nysted Offshore Wind Farm on harbour porpoises. Annual report to EnergiE2. Roskilde, Denmark, NERI.

Tougaard, J., Carstensen, J., Teilmann, J., Skov, H., and Rasmussen, P. 2009. Pile driving zone of responsiveness extends beyond 20 km for harbour porpoises (*Phocoena phocoena,* (L.)). Journal of the Acoustical Society of America 126: 11–14.

Tougaard, J., Carstensen, J., Wisz, M.S., Teilmann, J., Bech, N.I., and Skov, H. 2006b. Harbour porpoises on Horns Reef in relation to construction and operation of Horns Rev Offshore Wind Farm. Technical report to Elsam Engineering A/S. Roskilde, Denmark, National Environmental Research Institute.

Tougaard, J., Scheidat, M., Brasseur, S., Carstensen, J., Petel, T. v. P., Teilmann, J., and Reijnders, P. 2010. Harbour porpoises and offshore development: increased porpoise activity in an operational offshore wind farm. Proceedings of the 24th conference of the European Cetacean Society. Stralsund, Germany, European Cetacean Society. p 63.

Verboom, W.C. and Kastelein, R.A. 2005. Some examples of marine mammal discomfort thresholds in relation to man-made noise. Proceedings UDT 2005, Amsterdam.

5 ToR b. Review the current contaminant loads reported in marine mammals in the ICES area, the cause–effect relationships between contaminants and health status, and the population-level effects of environmental impacts

5.1 Introduction

Marine mammals are exposed to a variety of anthropogenic contaminants mainly through their diet. Many marine mammals are top predators and are at particular risk from biomagnification of contaminants through the food chain. Most research has focused on two main groups of contaminants: the persistent organic pollutants (POPs) and the heavy metals. However, there is some information on other contaminants including the polyaromatic hydrocarbons (PAHs), the butyltins and most recently the perfluorinated chemicals.

5.1.1 Persistent organic pollutants

This group of chemicals includes the organohalogenated compounds (such as the polychlorinated biphenyls - PCBs), the dichlorodiphenyltrichloroethanes (DDTs), polybrominated biphenyls (PBBs), polybrominated diphenyl ethers (PBDEs), chlordane, toxaphene, the cyclodienes (such as aldrin and dieldrin), and polychlorinated terphenyls (PCTs). Of these the occurrence and potential effects of the organochlorine compounds (OCs) are by far the best investigated. Many chlorinated pesticides are also included in this group. The significance of these compounds for marine mammals is that:

- they are highly lipophilic and hydrophobic;
- they bioaccumulate sometimes to high concentrations in lipid-rich tissues like marine mammal blubber;
- they are chemically very stable and persistent, many compounds being resistant to metabolic degradation;
- they are present as many different isomers and congeners, and comprise hundreds of different chemical formulations which may have different behaviours and toxicities;
- they have reproductive and immunosuppressive effects, and many are 'endocrine disrupters' - acting as hormone agonists or antagonists;
- animals are exposed to complex mixtures of compounds that may have additive or synergistic effects on various target organs and systems.

In marine mammals most of these compounds are sequestered into the blubber so much of the determination of POP residues has concentrated on this tissue. Between 90 and 95% of the total burden of many POPs, particularly PCBs and DDTs, are found in the blubber because of its high lipid content (Aguilar *et al*., 2002). The compounds are essentially bound away in this tissue until the lipid store is mobilized for energy requirements or for the production of milk. This aspect of the life cycle of marine mammals means they may be re-exposed to the contaminants when they call upon their blubber reserves during periods of natural fasting. Many factors can affect the occurrence and distribution of POPs in marine mammals. These include diet, foraging strategy, age, species, sex, and nutritional condition. These confounding variables need to be considered when interpreting the significance of reported tissue concentrations (Aguilar *et al*., 1999). This is particularly the case for animals that do not feed during the breeding season, and also means that females can offload a large proportion of their contaminant burdens to their offspring (Debier *et al*., 2003).

The production of PCBs and DDTs has been limited or completely banned since 1970s in most developed countries. However, organochlorine compounds including PCBs are still being released into the environment by (1) use, disposal or accidental release from previously produced material, (2) volatilization of previously released material, and (3) creation of PCBs and dioxins during combustion processes (Toft *et al*., 2004, and ref. therein). Nearly 97% of the historical use of PCBs was estimated to have occurred in the northern hemisphere (Breivik *et al*., 2007). Tanabe *et al*. (1988) calculated that only 30% of the produced PCBs have dispersed in the environment. PCBs have an environmental half-life of 20–40 years (Erickson, 1986) are declining only very slowly in most ecosystems globally. Several factors including accidental release, improper storage, inadequate disposal and ongoing use in materials and products contribute to its continuing presence in the environment (Tanabe *et al*., 1994; Aguilar *et al*., 2002; Breivik *et al*., 2007). The most recent predictions of global trends in PCBs suggest that PCB levels will not decline until around 2050 onwards (Breivik *et al*., 2007).

Newer POPs such as the polybrominated diethylethers (PBDEs), hexabromocyclododecane (HBCD) and tetrabromobisphenol A (TBBP-A) used as flame retardants are now found in the blubber of seals and cetaceans from UK and other waters raising concerns about their potential for toxics effects (Allchin *et al*., 1999). The deca-product mixture is still in use, whereas the penta and octa- mixtures containing the lower brominated compounds (de Wit, 2002) have been banned in Europe. In the US the penta and deca-mixtures are both still legally used in many industries but some States have now passed laws to phase out use of the penta and octa BDEs.

5.1.2 Heavy metals

The heavy metals are a heterogeneous group of compounds. Some are bioaccumulative (such as mercury) whereas others appear not to (such as chromium, nickel and copper). Data on zinc and lead in various species in the marine foodweb are equivocal (Muir *et al*., 1992). The liver, kidney and bone are the main target organs for heavy metals and levels can vary widely depending on the geographical location of the species. Marine mammals appear be protected against the effect of many heavy metals because of the presence of metallothioneins (Bowles, 1999). These are proteins whose production is induced by the occurrence of divalent cations such as Hg++, Cd++, Cu++ and Zn++. Metallothioneins have a high affinity for binding such cations, and they sequester the metals to form biochemical complexities with reduced toxicities. In addition mercury forms complexes with selenium, producing insoluble tiemannite granules (Nigro *et al*., 2002). This is an important mechanism, complementary to excretion, and allows many species to cope with a relatively high dietary exposure to mercury (Dietz *et al*., 1996). High levels of liver cadmium have been reported in a number of cetacean species and this probably also reflects dietary preferences (Bustamante *et al*., 1998). High concentrations of cadmium are accumulated in the liver and gonads of cephalopods (Hamanaka *et al*., 1982), the prey species of many cetaceans.

5.1.3 Polyaromatic hydrocarbons

The potential for the biomagnification of polyaromatic hydrocarbons (PAHs) is low, because fish (the main food of marine mammals) are good metabolizers of PAHs compared with molluscs and other invertebrates (Law and Whinnett, 1992). Bioaccumulation or exposure to these compounds will be lower in fish-eating marine mammals than those that feed on cephalopods or small crustaceans and plankton (such as the mysticete whales). Seals and cetaceans also have a detoxification enzyme system in the liver, which is induced in response to various xenobiotic compounds, including PAHs. This system (known as the mixed function oxidase, MFO or cytochrome P450 system) can convert parent compounds into excretable metabolites, largely by the addition of a hydroxyl group (Sipes and Gandolfi, 1991). This biotransformation of compounds may, however, be toxic if the metabolites produced are bioactive. In addition the rate at which transformation occurs is critical. If the non-toxic pathway is saturated, minor pathways, which produce further toxic intermediates, become involved. PAHs (specifically benzo-a-pyrene) has been associated with DNA adducts and a high prevalence of tumours (around 15%) in St Lawrence belugas (De Guise *et al.*, 1995a; De Guise *et al.*, 1994b).

5.1.4 Butyl tins (Tributyl tin (TBT), Dibutyl tin (DBT) and Monobutyl tin (MBT))

These groups of compounds were identified in liver samples of marine mammals, following knowledge of their toxicity and endocrine disrupting effects in invertebrates and fish (Iwata *et al.*, 1994). Results of analysis in liver samples from stranded animals have indicated a widespread contamination around the coasts of England and Wales; indeed TBT and DBT have been found in open ocean cetacean species, which indicates a wider contamination of the sea by these compounds (Law *et al.*, 1999). However, recent data on temporal trends of DBT, TBT and MBT in harbour porpoises from Norwegian (Berge *et al.*, 2004) and UK waters (Law *et al.*, 1999; Jepson, 2005) have found relatively low tissue concentrations following the restrictions on the use of TBT on small boats in the late 1980s.

5.1.5 Perfluorinated organochemicals

Perfluorinated organic compounds are widely used in the manufacture of plastics, electronics, textile and construction material in the garment, leather and upholstery industries. Recent studies have also found perfluorinated organochemicals (FOCs) in the tissues of marine mammals. Van de Vijver *et al.* (2003) measured the presence of FOCs in marine mammals, indicating a potential biomagnification of these compounds and their widespread occurrence. Liver, kidney and spleen appear to be the major target organs (Van de Vijver *et al.*, 2005). Among all the measured FOC compounds, PFOS (perfluorooctane sulfonate) was predominant in terms of concentration. The highest PFOS concentrations were found in the liver of harbour seal compared with white-beaked dolphin, harbour porpoise, grey seal, sperm whale, white-sided dolphin, striped dolphin, fin whale, and hooded seal. Harbour and grey seals and white-beaked dolphin, which displayed the highest trophic position, contained the highest PFOS levels, while offshore feeders such as sperm whales, fin whales, striped dolphin, and white-sided dolphin showed lower PFOS concentrations (Van de Vijver *et al.*, 2005). In UK waters, PFOS concentrations from <16 to 2420 ng/g wet weight were detected in harbour porpoise livers but perfluorooctanoic acid (PFOA) levels were not detected (Law *et al.*, 2008b).

5.1.6 Radionuclides

Few studies have been conducted on radioactivity levels associated with radionuclides (e.g. Cs137) in marine mammals. The few studies that have been conducted levels have generally shown that levels of radioactivity associated with radionuclides like Cs137 are typically low and usually emit significantly lower levels of radioactivity than naturally occurring radioisotopes such as K40 (e.g. Berrow *et al.*, 1998; Watson *et al.*, 1999).

5.1.7 Sources of data

There is a huge body of literature on contaminants in marine mammals worldwide. Reviews on the levels of contaminants found, the patterns of different compound groups in various species and the temporal changes in concentrations. The most comprehensive previous reviews are: Aguilar and Borrell (1997), Geraci and St Aubin (1990), Hall (2001), Law (1996), O'Shea (1999), Reijnders, Aguilar and Donovan (1999) and Aguilar *et al*. (2002).

The exposure to and effects of contaminants in marine mammals within the ICES area have not been formally reviewed for around a decade or so (ICES WGMMH and WGMMPD 2000; ICES WGMMPD 2001). In this review of contaminants and their toxic effects we therefore limited our research to scientific publications published from January 2000 to April 2010 inclusive. This report does not summaries all published literature on contaminant levels in marine mammals since 2000, but highlights the main studies and main regions and species of concern.

5.2 Assessing contaminant exposure within the ICES area

5.2.1 Arctic and adjacent waters

Although POPs are rarely used in the Arctic, they have been documented in Arctic wildlife since the beginning of the 1970s. Most POPs found in the Arctic are transported from distant industrial and agricultural sources by atmospheric and oceanic currents, as well as river discharges; through the circulation currents in the atmosphere, contaminants can be brought from the lower latitudes to the Arctic within days. The contaminants are then deposited and taken up mainly in the lipid rich food chains of the arctic marine ecosystem (Gabrielsen, 2007, and ref. therein). The Arctic Monitoring and Assessment Programme (AMAP) scientific reports provide an overview of POPs and heavy metals in Arctic species; their input into the environment and how they are eventually taken up by an organism, the regional and circumpolar levels and trends in these levels, temporal changes, and biological effects (AMAP 2004). Other studies (e.g. Gabrielsen and Henriksen, 2001; Gabrielsen 2007) provide an overview of what persistent organic pollutants (POPs), such as PCBs, DDTs, PBBs, PBDEs, PFAS, PFOS, and PCN, exist in Arctic species, which include fish, seals, whales, polar bears, birds, and the fox.

5.2.1.1 Cetaceans

Killer whales accumulate the highest levels of environmental POPs (including PCBs) of all marine mammals (and probably all species) (Ross *et al*., 2000). Wolkers *et al*. (2007) investigated the accumulation and transfer of contaminants in male killer whales (*Orcinus orca*) from northern Norway, in order to assess the degree and type of contaminant exposure and transfer in the herring–killer whale link of the marine foodweb. Results suggested that killer whales are one of the most polluted Arctic animals, with reported average levels of 25 000 ng/g lipid for total polychlorinated biphenyl (PCB) and chlorinated pesticides, and 500 ng/g for PBDEs. Compared with seals, other cetaceans and polar bears within the Arctic region, and even polluted beluga whales (*Delphinapterus leucas*) in the St Lawrence estuary, levels of compounds measured were higher in killer whales. Only killer whales from the Northeast Pacific showed comparable or higher halogenated organic contaminants (HOC) levels. The high levels reported in killer whales off northern Norway is possibly due to main their food source, herring, which showed 10 to 15 times higher HOC levels than, for example, polar cod from Svalbard; the dominant food of the beluga whales in the Arctic. On the whole, comparing the contamination of the killer whale's diet with the diet of beluga whales in the high-arctic waters, revealed six to more than 20 times higher levels in the killer whale diet. Consequently, levels in killer whales are between five and eight times higher than in beluga whales. There are specific concerns about the long-term health effects associated with these very high POP (especially

PCB) exposures (Hickie *et al*., 2007). A number of pathogens have been identified in killer whales, some of which cause abortions, reduced fecundity and/or increased mortality (Gaydos *et al*., 2004; Raverty pers. comm. 2007).

The COSEWIC (2008) assessment and status report on the killer whale evaluated different populations in Canada. The status for the North-west Atlantic/eastern Arctic population was considered of "Special Concern". One of the reasons given for the designated status was the threat of contaminants. Nothing is really known about e.g. food specialization of the Northwestern Atlantic/eastern Arctic population. In the eastern Canadian Arctic, killer whales have been reported to prey on bowhead whales (*Balaena mysticetus*), belugas, narwhals (*Monodon monoceros*), long-finned pilot whales (*Globicephala melas*), fin whales (*Balaenoptera physalus*), common minke whales (*Balaenoptera acutorostrata*), humpback whales (*Megaptera novaeangliae*), and seals (Higdon, 2007). Furthermore, there are no abundance estimates and no available information on trends in population size.

In beluga whales, the levels in recent measurements from the North-east Atlantic are 2–3 times lower than those made in the 1990s; which may be due to changes in feeding habits to almost exclusively krill after the collapse of the capelin (*Mallotus villosus*) stocks in 1986 (Gabrielsen, 2007). Higher POPs levels have been reported in eastern Canada and Svalbard beluga whales compared with southern Alaska (Wolkers, 2002). The levels of PBDEs in beluga whales from Svalbard were much lower than beluga whales from Canada and more southern latitudes (Wolkers *et al*., 2004a; Gabrielsen, 2007).

Kelly *et al*. (2009) made a comparative analysis of perfluoroalkyl contaminants (PFCs) and lipophilic organohalogens in a Canadian Arctic foodweb. Perfluorooctane sulfonic acid (PFOS) exposure in nursing Hudson Bay beluga whale calves (CI95 range: 2.7 × 10-5 to 1.8 × 10-4mg· kgbw-1 ·d-1), exceeded the oral reference dose for PFOS (7.5 × 10-5 mg· kg bw-1 ·d-1). PFC concentrations in the liver were generally equivalent to or higher than organochlorine concentrations, while PBDE concentrations are comparatively low. These results signal that legacy POPs (i.e. PCBs), banned nearly 40 years ago, remain a potential threat to wildlife, even in remote ecosystems (Kelly *et al*., 2009). PFOS and C8–C14 perfluorocarboxylic acids (PFCAs) are highly bioaccumulative in the Arctic marine foodweb. Unlike lipophilic POPs, perfluorinated acids (PFAs) exhibited no biomagnification in aquatic foodweb organisms. The observed foodweb-specific biomagnification of PFAs in this study can be explained by the anticipated phase partitioning behaviour of these recalcitrant proteinophilic compounds.

Dam and Bloch (2000) reported high levels of PCBs in pilot whales and other POPs were also comparably high; mean ∑PCB in the blubber was 11 900 ng/g wet weight, and all individuals were sampled from the Faroe Islands in 1997 (n=417). Van Bavel *et al*. (2001) reported that levels of brominated compounds in pilot whales are an order of magnitude higher than in other arctic marine mammals at that time (Gabrielsen, 2007).

In minke whales, regional variations in PCBs and organochlorine pesticide concentrations were examined using the blubber of 155 minke whales sampled in seven regions in the North Atlantic and European Arctic, including western and southeastern Greenland, the Norwegian Sea, the North Sea and the Barents Sea. Concentrations of major OC groups (ΣPCB, 89.1–22 800 ng/g lipid; ΣDDT, 65.3–6280 ng/g lipid; ΣCHL, 33.3–2110 ng/g lipid) generally increased from west to east, while HCH concentrations (ΣHCH, <1–497 ng/g lipid) showed the opposite trend (Hobbs *et al*., 2003).

Very high PCB levels (20 000–30 000 ng/g wet weight) were measured in harbour porpoise blubber samples from northern Norway. Berggren *et al*. (1999) stated that the levels of PCBs and DDTs were comparable with the Baltic Sea and North Sea and, at that time, the highest measured in any whale species from the Arctic. In contrast, levels in harbour porpoises from Greenland are much lower (Gabrielsen, 2007). The reason for the high PCB levels in harbour porpoise from northern Norway is not fully understood (Gabrielsen, 2007).

5.2.1.2 Pinnipeds

Muir *et al*. (2000) reported on a geographical study of PCBs and DDTs in ringed seal blubber, which showed higher levels in samples from the Yenisey Gulf in the Russian Arctic, Svalbard and eastern Greenland compared with western Greenland and the Canadian Arctic (Gabrielsen, 2007).

Bang *et al*. (2001) analysed blood samples from 12 adult ringed (*P. hispida*) and 11 bearded seals (*E. barbatus*) for organochlorines in Svalbard. The mean value of ∑PCB was 624.81 ng/g, which is considerably lower than levels recorded in pinnipeds in other more contaminated ICES regions. The highest concentrations were for ∑PCBs followed by ∑DDT, in both sexes. ∑PCBs and ∑DDT were higher in ringed seals that bearded seals, whereas ∑HCH (α,ß,γ) levels were higher in bearded seals. In ringed seals, females and males, ∑PCB was 337 ± 95 ng/g (n=6) and 625 ± 443 ng/g (n=6), whereas ∑DDT was 165 ±47 ng/g (n=6) and ∑621± 559 ng/g (n = 6), respectively. In bearded seal females and males, ∑PCB was 159 ± 132 ng/g (n=6) and 248 ± 93 ng/g (n = 5), whereas ∑DDT was 46 ±41 ng/g (n=6) and161± 71 ng/g (n = 5), respectively. Mono-ortho and di-ortho congeners contributed up to 65-70% to ∑PCB, and *p,p'*-DDE was the main DDT-related compound in both species consistent with other studies of marine mammals in the Northern hemisphere (Muir *et al*., 1992; Muir *et al*., 1999). The authors conclude that the OC-levels reported in ringed and bearded seals were in the lower range of previously reported concentrations in both these species at Svalbard. The higher concentration of most OCs in ringed seals compared with that of bearded seals is caused by the higher trophic position held by ringed seals in the Svalbard ecosystem. Interestingly, although concentrations of most OCs were significantly higher in males than in females, in ringed seals no significant inter-sex differences were found for any OCs. The observed sex differences in bearded seals are most likely related to the previously documented excretion of OCs during gestation and lactating in females.

Wolkers *et al*. (2004b) investigated the accumulation of PCBs and pesticides in harbour seals (*Phoca vitulina*) in Svalbard. Both PCB and pesticide levels reported in harbour seals were low compared with more southern harbour seal populations; animals from Svalbard contained 5–10 times lower contaminant levels, compared with seals from the Norwegian mainland, and 30 times lower concentrations than those of harbour seals from the Gulf of St Lawrence in eastern Canada (Bernt *et al*., 1999, Ruus *et al*., 2002, Wolkers *et al*., 2004b). Ringed seals from Svalbard have contaminant levels that are comparable with the harbour seals, probably because the diet, as well as the metabolic capacity, of the two species is similar at this location.

Observed ∑PCB (n=20) in harbour seals in the Wolkers *et al*. (2004) study are outlined below:

	FEMALE (N=4)	MALE (N=6)	PUP (N=4)	MILK (N=4)
∑PCB (ng/g)	458.8	2201.1	540.9	271.6

Levels of different toxaphene congeners, DDE, HCB, and HCH were also evaluated. The authors emphasized the difficulties in comparing different studies due to a lack of standardized sampling methods. Additionally, variation in contaminant levels can be due to age, sex, body condition, but also shifts in ocean currents, etc., so these factors must also be accounted for in order to make sensible comparisons.

The highest levels of toxaphene, a chlorinated pesticide, have been found in harp seal collected east of Svalbard. The levels of toxaphene (Tox 26 and Tox 50) were 20 times higher than in ringed seal samples west of Svalbard (Wolkers *et al.*, 1998; Wolkers *et al.*, 2000; Gabrielsen, 2007).

5.2.1.3 Other species

Gabrielsen (2007) provided a review of contaminant levels in polar bears within the Arctic region (*U. maritimus*); with information on geographic trends and health and reproductive effects from contaminant exposure.

Bernhoft *et al.* (2000) investigated blood samples from 56 polar bears (25 males, 31 females) from Svalbard for immunoglobulin G (IgG) levels and organochlorine burdens, in order to assess the potential PCB effects on the immune system. No differentiation between sexes were observed, and the median ∑PCB (n=14) was 66.70 ng/g. Gebbink *et al.* (2008) reported a more recent study on PCBs, OH-PCBs, MeSO2-PCBs, PBDEs and OH-PBBs in east Greenlandic polar bears. The tissue examined included blubber, liver, brain, and blood. The contaminants showed significant differences in their values mainly in the liver and brain relative to blubber and blood. Congener pattern differences among tissues and blood are likely due to a combination of factors, e.g. biotransformation and retention in the liver, retention in the blood and the blood-brain barrier transport. These findings suggest that different congener pattern exposures to these classes of contaminants should be considered with respect to potential target tissue-specific effects in East Greenland polar bears. Mean ∑PCB (n=43) values for 20 male and female polar bears for the different tissue samples were given below:

TISSUE	FAT	BLOOD	BRAIN	LIVER
∑PCB (ng/g)	5387	40	148	3125

5.2.2 Baltic Sea

5.2.2.1 Cetaceans

Not much is known about the contaminant levels in the only indigenous cetacean of the Baltic Sea, the harbour porpoise. Porpoises from the Baltic Sea have been shown to have accumulated PCB levels 0.4 to 2.5 times higher than those from the Kattegat and Skagerrak (Berggren *et al.*, 1999). Beineke *et al.* (2005) found indications for contaminant-induced immunosuppression in stranded harbour porpoises on the German Baltic coast. The authors detected very high median PCB concentrations of 2890 ng/g in females and 5033 ng/g in males, which may increase susceptibility of disease in within this region.

5.2.2.2 Pinnipeds

Bergman (2007) assessed contaminant levels in three seal species inhabiting Swedish waters: grey (*Halichoerus grypus*), ringed (*Phoca hispida botnica*), and harbour seals. A severe decline in the Baltic grey and ringed seal populations took place during the second half of the 1960s. It was suggested that this decline was caused by the contamination of industrial chemicals, above all organochlorines such as PCB and DDT. High concentrations of these substances were found in the Baltic biota; first observed in the 1960s and reported by Jensen *et al*. (1969). Since the 1970s, concentrations of all OCs have decreased; DDT since the early 1970s and PCBs somewhat later, but with a more rapid decline (Olsson and Reutergård, 1986; Bignert *et al*., 1998). Some data suggests that the Bothnian Sea may be more contaminated by dioxin than the Baltic proper (Bignert *et al*., 2007).

Studies of juvenile harbour seals from the late 1980s indicate that concentrations of PCBs in specimens from the Baltic Sea were about twice as high as samples from individuals inhabiting waters off the Swedish west coast (Blomkvist *et al*., 1992). Grey seals had the highest concentrations among the three seal species that inhabit the Baltic Sea (Blomkvist *et al*., 1992). Experimental studies indicated that intoxication of organochlorines is an important, but not necessarily the only, factor in the aetiology of the Baltic Seal Disease Complex (BSDC). The decreased concentration of DDTs and PCBs in prey (fish) and predator (seal), which, at least partly, is paralleled by an improved health and increased seal population sizes during the last decade, further supports the hypothesis that BSDC is primarily caused by organochlorine contaminant exposure (Bergman, 2007).

Although the contaminant levels in Baltic seals have decreased since the end of the 1970s, the levels in Baltic seals are still relatively high compared with seals living in unpolluted waters, especially in ringed seals - differences in levels of toxins between grey and ringed seals could be explained by differences in their diets (ICES WGMME 2005).

5.2.3 North-east Atlantic

5.2.3.1 Cetaceans

Alongside the UK Cetacean Strandings Investigation Programme (CSIP), work has been conducted at Cefas to determine contaminant levels in selected animals. Over 20 species have been investigated in total, but most work has focused on harbour porpoises as these have the widest distribution around UK coasts and are the most frequently stranded or bycaught cetacean. In 2008–2009, attention focused on establishing long-term temporal trends in blubber concentrations of chlorobiphenyls, brominated diphenyl ethers and hexabromocyclododecane in harbour porpoises. Chlorobiphenyls are present in the environment as a result of the widespread (and primarily historical) use of polychlorinated biphenyls, particularly in electrical transformers. The use of PCBs was banned progressively from open and closed uses in the UK, beginning in 1981. The other two compounds are flame retardants. Polybrominated diphenyl ethers (PBDEs) comprise three technical products, known as the penta-mix, octa-mix and deca-mix formulations, of different degrees of bromination. The structure of BDEs is similar to that of CBs, and 209 congeners are possible in both cases. The penta- and octa-mix products were withdrawn from the European market prior to August 2004, and the deca-mix product was banned from use in electrical and electronic goods within the EU from July 2008. HBCD has been subject to an EU risk assessment of continued production and use, and currently no restrictions have been placed upon that compound. The results of these temporal trend assessments have been published in the scientific literature (Law *et al*., 2008a; Law *et al*., 2010; Law *et al*., accepted for publication).

The time periods for the three assessments are listed in Table 5.

Table 5. Data available for temporal trend assessment.

Compounds	Temporal range	Number of animals
CBs	1991–2005	440
BDEs	1992–2008	415
HBCD	1994–2006	223

The methods used for determining concentrations of contaminants in blubber were: CBs – gas chromatography with electron-capture detection; BDEs – gas chromatography/electron capture negative ion mass spectrometry; HBCD – high performance liquid chromatography/electrospray negative ion mass spectrometry. Full analytical quality control procedures were applied in all cases, including the analysis within each batch of a certified or laboratory reference material used to track the day-to-day performance of the method, and participation in both a laboratory proficiency scheme and inter-laboratory studies as available. A non-parametric statistical method was used, because it avoids making assumptions about the distribution of the *S*-values, and, more importantly, we do not have to assume any particular functional form for the trend (e.g. linear, exponential). Also, potential confounding factors (area, season, bycaught or stranded, age class, sex, blubber thickness and lipid content) were investigated and found not to confound any of the trends identified.

In studying possible time-trends for CBs, data were available for harbour porpoises (n=440) (Figure 4), bottlenose dolphins (n=15) (Figure 5) and killer whales (n=5) during 1991–2005. In this case, the same suite of 25 CB congeners was determined throughout the study period, comprising CB18, CB28, CB31, CB44, CB47, CB49, CB52, CB66, CB101, CB105, CB110, CB118, CB128, CB138, CB141, CB149, CB151, CB153, CB156, CB158, CB170, CB180, CB183, CB187 and CB194. From the data, it was clear that Σ25CBs concentrations in UK harbour porpoises are declining only slowly in the 1990s and levelled off in the 2000s as a result of a ban on the use of PCBs which began more than two decades ago (Law *et al.*, 2010). This decline is much slower than that observed for organochlorine pesticides (such as DDTs and dieldrin). There are also regional differences in PCBs and OC pesticide levels within UK waters (lower levels in Scotland), possibly reflecting differences in diffuse inputs and transfer between regions, e.g. via the atmosphere. A similar decline in PCB levels was found in a group of common dolphins that mass stranded in the UK in 2008 as compared with levels of stranded common dolphins in the same geographic region from the early 1990s (Jepson and Deaville, 2009). The reason for the slow decline is likely due to both continuing diffuse inputs from e.g. PCB-containing materials in storage and in landfills where these were disposed of prior to the more stringent requirements for such sites being enacted, and to the substantial reservoir of PCBs already in the marine environment. Further efforts to limit or eliminate PCB discharges to the marine environment are still needed.

PCB exposure data also exist for UK-stranded bottlenose dolphins (n=15) and killer whales (n=5) for the same period (1991–2005). The mean level for PCBs in UK-stranded bottlenose dolphins was almost 100 000 ng/g lipid weight (Jepson *et al.*, 2008) and 225 000 ng/g lipid weight for the killer whales (Law, 2006c; Cefas, unpublished data). Although these data are from stranded animals, they show that PCB exposures are similar or greater than levels in biopsied bottlenose dolphins in the SW Atlantic such as Indian River Lagoon (Florida, US), Sarasota Bay (Florida, US) and Charleston (North Carolina, US) (Schwacke *et al.*, 2002; Wells *et al.*, 2005; Hall *et al.*, 2006b; Fair *et al.*, 2010). PCB blubber levels in UK-stranded killer whales are also similar to the very highest PCB levels recorded in adult transient male killer whales blubber in British Columbia, Canada (Ross *et al.*, 2000; McHugh *et al.*, 2007).

One particular flame retardant, hexabromocyclododecane (HBCD) was causing some concern and was found at relative high levels in the blubber of harbour porpoise stranded along the Irish sea coast, where levels were an order of magnitude higher (~3 µg g^{-1} lipid) than elsewhere except the northwest coast of Scotland where levels were ~5 µg/g lipid (Zegers *et al.*, 2005). In the period 1994–2003, a sharp increase in concentrations of HBCD in porpoise blubber from about 2001 onwards was also reported in UK-stranded harbour porpoises (Law *et al.*, 2006). The maximum HBCD concentrations observed was 21.4 $mgkg^{-1}$ lipid weight in a porpoise which died in 2003. A further study of UK-stranded harbour porpoises (n=223) was conducted which took the time-trend forward to 2006 and showed a statistically significant decrease in HBCD levels between 2003 and 2004 that continued to 2006 (Law *et al.*, 2008a). Possible contributory factors to the observed decrease include the closure in 2003 of an HBCD manufacturing plant in NE England which had considerable emissions up to 2003, and the closure in 2002 of a plant in NW England using HBCD in the manufacture of expanded polystyrene. Two voluntary schemes intended to reduce emissions of HBCD to the environment from industry may also have had some impact, though they did not, however, formally begin until 2006.

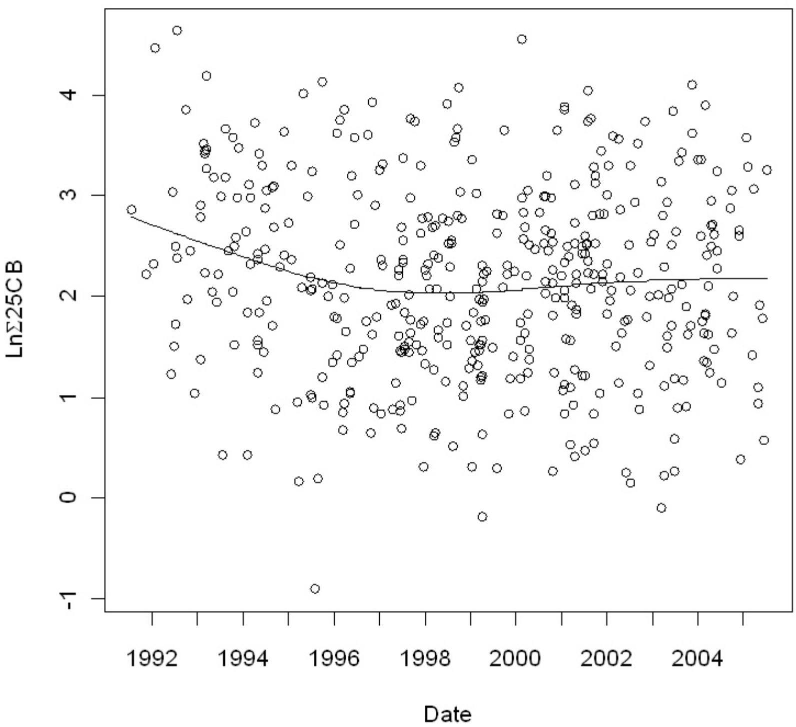


Figure 4. Ln Σ25CB (the natural logarithm of the sum of 25 CB congeners determined) concentrations on a lipid basis by year for 440 harbour porpoises (*Phocoena phocoena*) stranded in the UK from 1991–2005 (based on Law *et al.*, 2010; Marine Pollution Bulletin 60: 470–473).

For the investigation of time-trends for BDEs, data were available for 415 porpoises that were autopsied between 1992 and 2008. Over the full period of the investigation, the suite of BDEs determined changed periodically, but nine congeners were determined throughout and the sum of the concentrations of these was used for the time-trend assessment. The congeners were: BDE28, BDE47, BDE66, BDE85, BDE99, BDE100, BDE138, BDE153 and BDE154. The maximum summed BDE concentration observed was 15.7 $mgkg^{-1}$ lipid weight in an animal which died in 1993. The analysis indicates that the median concentrations peaked around 1998, and have reduced by between 55% and 76% in 2008. The best point estimate is 66% ($p < 0.001$). This finding was not confounded by a range of other factors which were also considered (area, season, nutritional status, bycaught/stranded and age class). The BDE congeners found in UK marine mammals arise primarily from the penta-mix PBDE product, which was banned in the EU in 2004, but this ban was widely foreseen and it is likely that removal of the product from the market and a switch to alternatives began before that date.

The Marine Animals Research and Intervention Network (MARIN) investigated the cause of death of marine mammals stranded on the coastline of the southern North Sea (Belgium and northern France) or incidentally captured in fishing gear. Between 1990 and 2008, 520 porpoises (*Phocoena phocoena*) were necropsied and sampled. Initially, levels of organohalogenated contaminants in porpoises were assessed in 21 individual that stranded between 1997 and 2000 (Covaci *et al.*, 2002). The mean concentration in the liver of 59 PCB congeners was 36 400 ± 26:4 ng/g lipid. Higher concentrations of organochlorine compounds were found in porpoises stranded on the Belgian/Dutch coast of the southern North Sea in comparison with the English coast. A second study on harbour porpoises (n=35) revealed that concentrations of 21 PCBs congeners and 10 PBDE congeners in the blubber were 12 400 ng/g lipid weight and 760 ng/g lipid weight, respectively (Weijs *et al.*, 2009). The highest PCB concentrations were observed in adult males indicating bioaccumulation and the highest PBDE concentrations were measured in juveniles. Porpoises had higher biomagnification factors for lower chlorinated PCBs and for all PBDEs compared with harbour seals; consistent with the theory that porpoises have a less efficient cytochrome P450 system than seals for metabolizing several PCB and PBDE congeners (Weijs *et al.*, 2009).

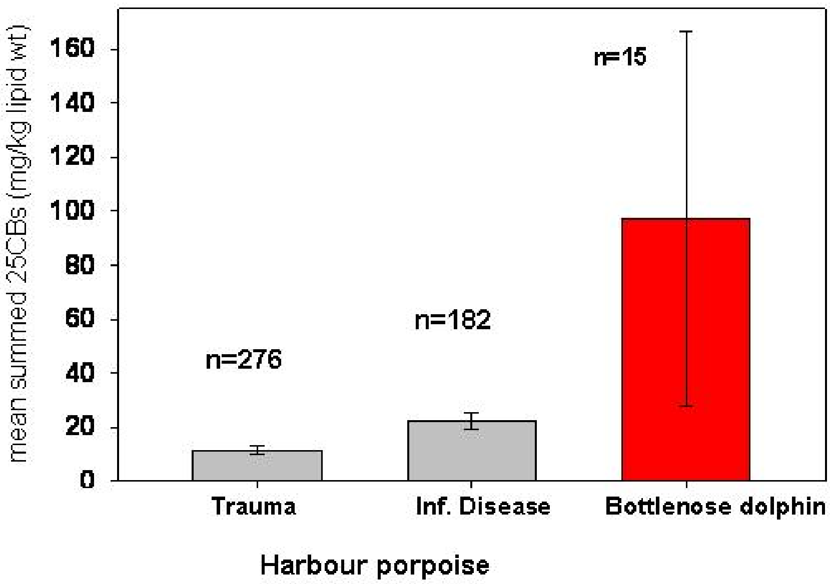


Figure 5. Comparison of mean summed 25CBs concentrations in UK-stranded harbour porpoises (trauma and infectious disease cases) and bottlenose dolphins (1991–2005). Bars=2SE.

For the German North (coast of Schleswig-Holstein) and Baltic Seas coasts, porpoises, including stranded and bycaught animals, are the main species that were necropsied at Forschungs-und Technologiezentrum Westkuoste, Christian-Albrechts-UniversitÌtzu Kiel, Busum – animals were in various stages of decomposition and the majority were frozen (-20°C) prior to necropsy. To evaluate the suspected impact of PCBs and PBDEs on the porpoise immune system (thymic atrophy and splenic depletion), bycaught and stranded individuals from North and Baltic Seas, Norwegian and Icelandic waters were necropsied and the health status was evaluated based upon the severity of the main pathological findings. Results identified that thymic atrophy and splenic depletion were significantly correlated with increased PCB and PBDE levels (Beineke *et al*., 2005).

The thyroids of 57 harbour porpoises from the German and Danish (North and Baltic Seas), Norwegian, and Icelandic coasts were examined histologically, and thyroid morphology differed strongly between areas (Das *et al*., 2006). Porpoises from the German (North and Baltic Seas) and Norwegian coasts displayed a high percentage of connective tissues, between 30 and 38%, revealing severe interfollicular fibrosis and a large number of large follicles (diameter >200 microm). The results are consistent with the hypothesis of a PCB-induced thyroid fibrosis in harbour porpoises (Das *et al*., 2006). A second study conducted further histological and immunohistologic investigations on thyroids of 36 harbour porpoises from Belgian and UK waters. Although interfollicular fibrosis was observed, the largely negative relationships between organochlorine and trace metal (Cd, Fe, Zn, Cu, Se, and Hg) exposure and interfollicular fibrosis did not support the hypothesis that these contaminants have an adverse effect on thyroid morphometry (Schnitzler *et al*., 2008).

Research undertaken in the UK between 1989 and 2001 investigated trends in trace metals levels, including Hg, Cd, Pb, Se, Ag, As, Cr, Cu, Fe, Ni, Zn, predominantly, in harbour porpoises (Law *et al*., 2006c; Jepson 2005; Bennett *et al*., 2001). During this period, most metal were stable or declined over time (Jepson, 2005). Elevated Hg, Se and Zn were associated with animals dying due to infectious disease (compared with porpoises dying due to physical trauma) (Bennett *et al*., 2001; Jepson, 2005) but this may reflect redistribution of methylmercury from muscle via disease-associated loss of nutritional condition (Jepson, 2005). Elevated Zn levels in diseased animals may represent physiological redistribution of Zn in response to stress/infection rather than evidence of Zn-induced toxicity (Jepson, 2005). High Zn and Hg concentrations were also found in harbour porpoises stranded along the northern France, Belgian and German coasts (as compared with bycatch individuals from Iceland, Norway and the Baltic Sea) (Das *et al*., 2004). Similarly to the UK-stranded porpoises, increasing Zn levels were observed with deteriorating health condition (emaciation and bronchopneumonia), while Hg increases were not significant (Das *et al*., 2004).

5.2.3.2 Pinnipeds

Levels of PCBs in the Wadden Sea harbour seals have decreased by 50–65% between 1988 and 2002, those in the blubber of harbour seals in southeastern England (the Wash) have decreased by less than 10% over the same 14 year period, and the concentrations in the seals that died of PDV in 2002 were significantly higher than the survivors (Hall *et al*., 2006b). As in cetaceans, the brominated flame retardants (such as PBDEs) are being reported as potential endocrine disrupting compounds but levels are still considerably lower than PCBs in most oceanic regions. Although the production and use of the lower brominated compounds has been controlled in Europe, the oil industry continues to use BDE209 and the penta-mixtures (commercial formulations with lower brominated compounds) are still used in North America. Hall *et al*. (2003) found a correlation between PBDEs and thyroid hormone levels in grey seals during their first year of life and in adult harbour seals (Hall and Thomas, 2007) but it is not clear whether this relationship is causal.

Between 1990 and 2008, 128 harbour seals stranded on coastlines of Belgium and Northern France were necropsied and sampled by the Marine Animals Research and Intervention Network (MARIN). The most frequent causes of death were trauma (mostly bycatch) and infectious diseases, including morbillivirus infection. 28 seals stranded between 1999 and 2004 (also including some seals stranded on the Dutch coastline) were selected for toxicological investigations and 21 PCB congeners and 10 PBDE congeners were analysed. The median values for PCBs and PBDEs in the blubber were 23.1 mg/g lipid weight and 0.33 mg/g lipid weight, respectively (Weijs *et al.*, 2009). The highest PCB concentrations were observed in adult males indicating bioaccumulation. In contrast, the highest PBDE concentrations were measured in juveniles, which may indicate a development with age of metabolic capacities for these contaminants. PCBs and PBDEs concentrations were also determined in the serum of seals and the conclusions drawn were similar to that of the porpoise study (Weijs *et al.*, 2009). As mentioned earlier, both species differed in their contaminant levels, which may be due to a higher ability in harbour seals in metabolizing PCBs and PBDEs (Weijs *et al.*, 2009).

5.2.4 North-west Atlantic

5.2.4.1 Cetaceans

Cetaceans inhabiting coastal waters, estuaries and bays may be at a greater risk from pollution, due to their proximity to the outflow of industrial waste. A small isolated population of beluga whales that are highly contaminated by pollutants, mostly of industrial origin, resides in the St Lawrence estuary, Quebec, Canada. During the first half of the twentieth century, over-hunting was suggested to be the probable cause for the population decline, from several thousand animals to approximately 500 in the 1990s (De Guise *et al.*, 1995b). The lack of recovery for this population has been attributed to high contaminant burdens. Beluga whales in this region have had historically high levels of a range of contaminants including PCBs, OC pesticides, metals and PAHs (De Guise *et al.*, 1994a; De Guise, 1995a). In recent years though, concentrations of most of the bioaccumulative and toxic (PBT) chemicals examined had exponentially decreased by at least a factor of two between 1987 and 2002 while no increasing trends were observed for any of the PBTs measured (Lebeu *et al.*, 2007). Lebeu *et al.* (2007) reported that, a decreasing trends of PBT concentrations may be due to a decline in contamination of its prey, following North American and international regulations on the use and production of these compounds, or by a change in its diet or a combination of both.

Individual- and population-based models have been conducted that provide quantitative assessments of the accumulation of persistent organic pollutants over the lifetime of the beluga whale population in St Lawrence estuary and consider all aspects of its life history (Hickie *et al.*, 2000). The models are used to examine the history of polychlorinated biphenyl (PCB) accumulation by the endangered St Lawrence beluga population from 1950 to the present and to predict future trends based on likely contaminant loading scenarios. The history of PCB exposure via the diet is reconstructed from existing data and from PCB profiles in dated sediment cores. The models adequately describe the effects of age, growth, sex, and reproductive activity on PCB concentrations in the beluga, and results show good agreement with observed concentrations when migrating American eels (*Anguilla rostrata*) are included as 3% of the annual diet. PCB levels in the population appear to have peaked between 1967 and 1972. The model agrees with recent studies that have shown that PCB concentrations in the population are declining slowly.

Polychlorinated biphenyls (PCBs), chlorinated pesticides (i.e. dichlorodiphenyl-trichloroethane (DDT) and its metabolites, chlordanes (CHLs), dieldrin, hexachlorobenzene (HCB), mirex, polybrominated diphenyl ethers (PBDEs), perfluorinated chemicals (PFCs), and polyaromatic hydrocarbons (PAHs) were measured in blubber biopsy samples collected from 139 wild bottlenose dolphins (*Tursiops truncatus*) during 2003–2005 in Charleston (CHS), South Carolina and the Indian River Lagoon (IRL), Florida (Fair *et al*., 2010). Dolphins accumulated a similar suite of contaminants in both areas with ΣPCB dominating (CHS 64%, IRL 72%), followed by ΣDDT (CHS 20%, IRL 17%), ΣPCBs in adult male dolphins exceed the established PCB threshold of 17 mg/kg (Kannan *et al*., 2000) by a fivefold order of magnitude with a 15-fold increase for many animals; 88% of the dolphins exceed this threshold. For male dolphins, CHS had a higher mean ΣPCBs (93,980 ng/g lipid) compared to the IRL (79 752 ng/g lipid) although this was not statistically different. Whereas the PBDE geometric mean concentration was significantly higher in adult males in CHS (5920 ng/g lipid) compared with IRL (1487 ng/g). Further, blubber ΣPFCs concentrations were significantly higher in CHS dolphins. Collectively, the current ΣPCB, ΣDDT, and ΣPBDEs blubber concentrations found in CHS dolphins are among the highest reported values in marine mammals. PCB concentrations in adult male IRL dolphins were similar to biopsy samples collected from wild bottlenose dolphins along Georgia's estuarine area (Pulster *et al*., 2009) and male and juvenile bottlenose dolphins in Biscayne Bay, FL (Litz *et al*., 2007). Even higher PCB levels (93 300 ng/g lipid) were recorded in stranded bottlenose dolphins from the IRL during 2001–2004 (Johnson-Restrepo *et al*., 2005). Both CHS and IRL dolphin populations (especially CHS), carry a suite of organic chemicals at or above the level where adverse effects have been reported in wildlife, humans, and laboratory animals.

In another study of the Sarasota Bay resident bottlenose dolphin population, 47 blubber samples collected during June 2000 and 2001 were analysed for PCB concentrations of 22 congeners relative to life-history factors and reproductive success (Wells *et al*., 2005). Prior to sexual maturity, males and females exhibited similar concentrations of about 15–50 ppm. Classical patterns of accumulation with age were identified in males, but not in females. Subsequently, males accumulated higher concentrations of PCBs through their lives (>100 ppm), whereas females begin to depurate with their first calf, reaching a balance between contaminant intake and lactational loss (<15 ppm). In primiparous females, PCB concentrations in blubber and plasma and the rates of first-born calf mortality were both high. First-born calves had higher concentrations than subsequent calves of similar age (>25 vs. <25 ppm). Maternal burdens were lower early in lactation and increased as calves approached nutritional independence. Models have predicted that PCBs are probably suppressing Sarasota Bay population growth rates at current levels of PCB exposure (Hall *et al*., 2006c).

It has been reported that Mysticetes, i.e. baleen whales, are less contaminated-often by an order of magnitude - than some odontocetes, as they feed lower down the food chain (O'Shea and Brownell, 1994, Reeves *et al*., 2003). However, enzyme markers in tissues of endangered North Atlantic right whales, for example, indicate significant exposure to a nonbioaccumulative, but potentially toxic, dioxin-like compound, such as one of the polycyclic aromatic hydrocarbons (PAHs) (M. Moore, cited in Reeves *et al*., 2001; Reeves *et al*., 2003).

5.2.4.2 Pinnipeds

Polybrominated diphenyl ethers (PBDEs) were analysed in blubber of harbour seals collected between 1991 and 2005 along the North-west Atlantic (Shaw *et al.*, 2008). PPBDE concentrations (mono- to hexa-BDEs) detected in blubber samples (n = 42) ranged from 80 to 25720 ng g^{1} lw, (overall mean 2403 ± 5406 ng g^{1} lw). By age, mean PPBDE concentrations were: 3645 ± 7388, 2945 ± 5995, 1385 ± 1265, and 326 ± 193 ng g^{1} lw in pups, yearlings, adult males, and adult females, respectively. Unlike the trend for PCBs, no decreasing gradient from urban to rural/remote areas was observed for PBDEs in these samples, likely reflecting inputs from local sources. No significant temporal trend was observed for PBDEs in harbour seals between 1991 and 2005, although congener profiles shifted over time. Tetra-BDE-47 was the dominant congener, followed by BDEs-99, -100, -153, -154, and -155 in varying order, suggesting exposure to the penta-BDE product. In adult males, the hexa-BDEs contributed more to the total (22%) than BDEs-99 and -100 (14%), and concentrations of BDE-155 were elevated compared with -154. Higher BDEs were detected in a subset of seals (n = 12) including hepta-BDE-183, the marker for the octa-BDE mixture, and octa-BDE-197, along with several unidentified hepta- and octa- congeners. BDE-209 was detected in seal blubber at concentrations ranging from 1.1 to 8 ng g^{1} lw, indicating that deca-BDE is bioavailable in this marine foodweb. This is the first study to document the accumulation of BDE-209 at measurable levels in wild harbour seals. While the PBDE patterns in blubber indicate exposure to all three BDE commercial mixtures, the data also suggest that BDE-209 debromination by seal prey fish may contribute to the loading of lower brominated congeners (hexa- to octa-BDEs) in these seals.

5.2.4.3 Other species

Manatees (*Trichechus manatus*) are aquatic herbivores and so their low trophic level has not historically resulted in high levels of organochlorine pesticides, polychlorinated biphenyls or other bioaccumulative contaminants (Ames and Van Vleet, 1996) although copper concentrations in livers were significantly elevated in areas of high herbicidal copper in Florida from 1977 to 1981 (O'Shea, 1984). Environmental threats to manatees include changing weather patterns, habitat destruction and increasingly frequent blooms of toxic dinoflagellates (red tides) along the Florida coastline and low genetic diversity within the population may also put the species at risk of diseases such as papilloma virus (Reep and Bonde, 2006).

5.3 Effects of contaminant exposure

Knowledge of the effects of contaminants on marine mammals remains limited, largely due to the difficulties involved in investigating the responses in wild animals, although it has increased considerably in recent years.

It has always been relatively straightforward to determine the tissue concentrations of a range of chemical compounds in dead and live-captured animals, but the significance of these concentrations for the health and ultimate survival of the individuals is often more difficult to assess robustly. Some studies have investigated the responses to exposure on animals in captivity, compared responses between exposed and control groups and associations between dysfunction and contaminant exposure have been reported in free-living individuals and populations. These studies are increasing whereas those merely reporting levels in tissues are declining. Thus the body of information on correlations among toxic endpoints and contaminant exposure measures continues to increase and is now being supplemented with data from *in vitro* studies using cellular and molecular methods (De Guise *et al.*, 1998; Hammond *et al.*, 2005; Levin *et al.*, 2005). In addition more recent work has also focused on assessing the risk of contaminant exposure at the population level (Hall *et al*., 2006a; Hall *et al*., 2006d).

There are inherent limitations when investigating any potentially toxic effects using biomarkers of either contaminant exposure or effect. Many POPs co-vary so it is not possible to state equivocally that the biomarker response has been caused by a particular contaminant. It is often not possible to determine causality, only that a statistical association has been found between a biomarker and the contaminant in question. Biological variables such as age, sex, nutritive condition, disease, or other stressors may also act as confounders that can cause similar biological effects as those seen from POPs. There may be other contaminants that are not analysed and a range of synergistic, additive or antagonistic effects of contaminant mixtures may ultimately exist. For many reported biological effects in wildlife, the evidence of a causal link with a specific chemical contaminant is often rather weak.

5.3.1 Persistent organic pollutants

A number of persistent marine pollutants may pose a significant and global threat to the health status and viability of marine mammal populations, with persistent organochlorine compounds such as polychlorinated biphenyls (PCBs) generally rating the greatest concern (Tanabe *et al*., 1994). Some of the highest recorded levels of PCBs and other organochlorines have been recorded in marine mammals over the past several decades (Aguilar *et al*., 1999; Ross *et al*., 2000). Marine mammals appear particularly vulnerable to high level bioaccumulation of these contaminants due to their high trophic level, lipid-rich blubber which acts as a reservoir for lipophilic chemicals (Tanabe *et al*., 1994; Aguilar *et al*., 1999). The elimination of these compounds is further limited by the lack of water-blood exchange (via gills) which is the dominant mechanism in other aquatic organisms such as fish (Marsili *et al*., 1995).

The lipophilic nature of organochlorine compounds combined with the high fat content of cetacean milk also results in the phenomenon of maternal offloading whereby adult females redistribute a large part of their body organochlorine burden to their progeny via gestation and lactation (Aguilar and Borrell, 1994b; Tanabe *et al*., 1994). Such perinatal organochlorine exposure may represent a greater (immuno)toxic threat than exposure acquired as a juvenile or adult (reviewed in Vos *et al*., 1997; 1998). The first calf delivered is likely to receive the greatest maternal transfer of organochlorines and associated toxicity (Aguilar and Borrell, 1994b; Schwacke *et al*., 2002). In contrast males are unable to transfer their contaminant load and accumulate high contaminant levels throughout life.

The endocrine and reproductive effects of these chemicals are believed to be due to their ability to: (a) mimic the effect of endogenous hormones; (b) antagonize the effect of endogenous hormones; (c) disrupt the synthesis and metabolizm of endogenous hormones; and (d) disrupt the synthesis of hormone receptors (Amaral Mendes, 2002). Murphy *et al*. (in press) reported that the reproductive effects linked with exposure to endocrine disruptors such as PCBs and associated DDT-like compounds include decreased fecundity, implantation failure and sterility (caused by stenosis, occlusions and leiomyomas) in seals (Helle, 1976; Helle *et al*., 1976; Reijnders, 1986; Olsson *et al*., 1994; Reijnders, 1999; Bredhult *et al*., 2008); premature pupping in sea lions (DeLong *et al*., 1973); and also severe reproductive dysfunction through the development of cancer and possibly hermaphroditism in beluga whales (Martineau *et al*., 1987; De Guise *et al*., 1994; Reijnders, 1999). Elevated PCB levels may have consequences on uterine and placental health and, subsequently, foetal health and survival (Hohn *et al*., 2007; Murphy *et al*. in press). The findings of these studies however, although strongly suggestive, have not been conclusive as the etiology of the observed disorder has usually been uncertain (Reijnders, 2003).

Two observations on wild populations in the 1980s suggested that the uptake of POPs by marine mammals could have toxic effects similar to those reported in laboratory species. The first was the report that a serious decline in the population of harbour seals in the Wadden Sea might be due to the reproductive effects of contaminant exposure (Reijnders, 1980; Reijnders, 1984). Average pup production per female harbour seal in the Dutch Wadden Sea population declined by approximately 30%, and toxicology studies revealed that, of all the OCs analysed, PCB levels were significantly higher (by 5 to 7 times) in the Dutch Wadden Sea population compared with other contiguous populations (Reijnders, 1980). Experimental studies revealed that seals fed on fish from the Wadden Sea showed a decreased reproductive rate at an average total-PCB level of 25–27 $\mu g\ g^{-1}$ lipid; whereas a control group showed normal reproductive rates at mean PCB levels of 5–11 $\mu g\ g^{-1}$ lipid (Reijnders, 1986). Hormone profiles of non-pregnant animals fed fish from the Wadden Sea indicated that the effects occurred at the stage of implantation, whereas the follicular, luteal and post-implantation phases were not affected. On the whole, oestradiol-17β levels in seals fed with fish of a higher contaminant burden were lower than those of the control group. Lower levels of oestradiol could have impaired endometrial receptivity and prevented successful implantation of the blastocyst (Reijnders, 2003). The second effect was investigated following the outbreak of phocine distemper among harbour seals in European waters, in which differential mortality rates were reported among harbour seal populations around the UK coast (Hall *et al.*, 1992a). This observation led to a study of the OC contaminant burdens among animals that were victims and survivors of the epidemic. The results suggested that animals that died of the disease had higher blubber levels of OCs than survivors, although it was not possible to control for all potential confounders (Hall *et al.*, 1992b).

This finding was also repeated in a study of contaminant burdens in striped dolphins following a similar outbreak of dolphin morbillivirus in the Mediterranean Sea in 1990 (Aguilar and Borrell, 1994a) and in the 1987–1988 bottlenose dolphin morbillivirus outbreak in the US (Kuehl *et al.*, 1991). Results from the former study found that PCBs and other organochlorine pollutants with potential for immunosuppressive effects may have triggered the mass die off event, or enhanced its spread and lethality (Aguilar and Borrell, 1994a). In addition to a large number of abortions during the epizootic, unusual luteinized cysts, with the potential to impede ovulation, were reported on the ovaries; these cysts were associated with high levels of PCB exposure (Munson *et al.*, 1998). Furthermore, similar results were obtained in live and dead harbour seals following the 2002 European PDV epidemic (Hall *et al.*, 2006b). Studies by Ross *et al.* (1995) and DeSwart *et al.* (1994) found evidence of the mechanism of the effect. They reported immunosuppression in a group of captive harbour seals fed contaminated fish compared with animals fed clean fish. Natural killer cell activity (white blood cells that are particularly required in the defence against viral infection) in particular was depressed and lymphocyte function measured *in vitro* was lower in the exposed group. The high binding affinity of POPs such as PCBs to the aromatic hydrocarbon receptor (Ah-R) in some cetacean species such as beluga whales is thought to partly explain species-specific toxicity (Jensen and Hahn, 2001).

Bergman and Olsson (1985) also reported the occurrence of adrenocortical hyperplasia, hyperkeratosis and other lesions in grey and ringed seals from the highly polluted Baltic Sea. The pathologies seen were indicative of a Baltic seal disease complex (BSDC) involving OCs and hormone disruption, a finding also demonstrated in laboratory animals (Fuller and Hobson, 1986). Other abnormalities associated with the highest exposures to PCBs include skull and bone lesions in grey seals (Bergman *et al.*, 1992; Zakharov and Yablokov, 1990) and harbour seals from the Baltic (Mortensen *et al.*, 1992). More recently Hammond *et al.* (2005) found that harbour seal immune function assays carried out *in vitro* were impaired when exposed to a commercial mixture of PCBs whereas grey seal immunity was not affected.

Bergman (2007) assessed contaminant levels and health status in grey, ringed, and harbour seals inhabiting Swedish waters over a 25 year period. The decreasing concentrations of DDT but more constant concentrations of PCBs in Baltic grey seals from the 1960s to 1990s, favour the concept that PCBs, in particular, were associated with the generally low reproductive output during that period. This interpretation agrees with experimental results obtained from earlier studies on mink (*Mustela vison*) which showed that PCBs caused reproductive failure (Aulerich and Ringer, 1977). Grey seals had the highest concentrations among the three seal species that inhabit the Baltic Sea, and Baltic female grey seals with pathological changes had a higher PCB concentration than females without such changes (Blomkvist *et al.*, 1992). In Dutch experiments, when groups of captive harbour seals were fed Baltic or Atlantic herring, suppression of various cellular and antibody responses of the immune system was observed in the seals fed the Baltic herring, but not in those fed the Atlantic herring (de Swart *et al.*, 1995; Ross *et al.*, 1995). The decreased concentration of DDT and PCBs in prey (fish) and predator (seal) and experimental studies, which, at least partly, was paralleled by an improved health and increased seal population sizes during the last decade, supports this assumption that BSDC was primarily caused by high contaminant (PCB) exposure (Bergman, 2007).

The high prevalence of occlusions in ringed seals is mostly associated with organochlorine (PCB) pollution, though increased environmental contamination by other chemical factors such as polybrominated compounds might also be considered (Sellström *et al.*, 1993). Another explanation would be of species-specific nature implying that the reproduction in ringed seals is more sensitive to PCB and DDT compounds than grey seals. The PCB levels in Baltic grey seals may still be high enough to negatively impair their immune system (including poor wound healing and increase in prevalence of severe intestinal ulcers seen recently), but low enough to permit close to normal reproduction.

Case-control epidemiological studies by Jepson *et al.* (1999; 2005) and Hall *et al.* (2006a) using large sample sizes found that the risk of mortality from infectious disease in UK-harbour porpoises increased in a dose-dependent manner with increasing blubber PCB concentration (50% increase in relative risk of infectious disease mortality at concentrations of total PCBs >25 μg/g lipid in the blubber). Stranded harbour porpoises from the German, North and Baltic seas were more severely diseased than bycaught animals and thymic atrophy and splenic depletion were significantly correlated with increased PCB and PBDE levels (Beineke *et al.*, 2005). Similar correlations between thymic atrophy and elevated PCB levels (but only in porpoises with total PCB levels above the proposed 17 mg/kg lipid weight toxicity threshold; Kannan *et al.*, 2000) were found in 118 UK-stranded harbour porpoises stranded between 1989 and 2001 (Jepson, 2003). Various immune function endpoints measured *in vitro* in cetaceans (bottlenose dolphins (Lahvis *et al.*, 1995), beluga whales (De Guise *et al.*, 1998) and in wild polar bears (Lie *et al.*, 2005) following PCB exposure further suggest that these compounds are also immunosuppressive to small cetaceans and bears.

Along with marine mammals living in highly polluted waters such as the Baltic Sea (Bergman, 2007) and St Lawrence Estuary, Canada (Hickie *et al.*, 2000), the greatest toxicity concern in marine mammal science must be the very high PCB levels in killer whales, bottlenose dolphins and polar bears within the ICES area. High PCB levels have been reported in biopsied bottlenose dolphins from Sarasota Bay (Florida, US) and associated with potential health effects such as increased first calf mortality (Hall *et al*., 2006c; Wells *et al*., 2005). As mentioned earlier, another study of 139 wild bottlenose dolphins in Charleston (CHS), South Carolina and the Indian River Lagoon (IRL), Florida (USA) found ΣPCBs in adult male dolphins exceed the proposed PCB toxicity threshold of 17 mg/kg (Kannan *et al*., 2000) by a fivefold order of magnitude with a 15-fold increase for many animals; 88% of the dolphins exceed this threshold (Fair *et al*., 2010). Collectively, the levels of ΣPCB, ΣDDT, and ΣPBDEs blubber concentrations found in CHS dolphins in 2003–2005 were among the highest reported values in marine mammals and both CHS and IRL dolphin populations, particularly those in CHS, carry a suite of organic chemicals at or above the level where adverse effects have been reported in wildlife, humans, and laboratory animals.

Similar or even higher PCB levels have been recorded in stranded bottlenose dolphins in UK waters (Jepson *et al*., 2008), which greatly exceed PCB levels associated with infectious disease mortality in case-control studies on UK-stranded harbour porpoises (Jepson *et al*., 2005; Hall *et al*., 2006a). It is difficult to obtain sufficient samples sizes to conduct case-control studies in bottlenose dolphins or killer whales, partly because stranding rates of both these species are low. Although there is a scarcity of data on PCB levels from stranded or biopsied killer whales, the few studies that have been conducted show extremely high levels in killer whales in North-east Atlantic, Arctic waters (Law 2006c; McHugh *et al*., 2007; Wolkers *et al*., 2007) and British Columbia, Canada (Ross *et al*., 2000), which typically exceed proposed thresholds for PCB toxicity (Kannan *et al*., 2000; Jepson *et al*., 2005). Most killer whale populations that have been assessed for abundance and population trends (mainly in the Pacific) are stable or declining (COSEWIC, 2008) and concerns must exist about the very high exposure in North-east Atlantic waters where no reliable population estimates are available.

Another marine top predator with high POP exposure is the polar bear. Bernhoft *et al*. (2000) investigated blood samples from 56 polar bears at Svalbard, and results showed a negative association of PCBs and IgG in blood plasma, which may indicate an immunotoxic effect, i.e. a contaminant-associated suppression of the antibody-mediated immunity. Gebbink *et al*. (2008) reported a more recent study of PCBs, PBDEs in east Greenlandic polar bears (blubber, liver, brain, and blood). Their findings showed that tissue composition of congener pattern varies as a function of the multiple congener class in question and suggest that exposures with respect to congener patterns may elicit target tissue-specific effects in east Greenland polar bears. A large comprehensive study carried out between Norwegian and Canadian researchers comprising epizoological (ecological) studies (reproductive rate, offspring survival), experimental studies on the immune system function, monitoring studies (e.g. physiological assays of thyroid hormones, retinol, IgG and testosterone) and registration of biological data (e.g. sex, age, reproductive status, nutritional status) were coupled with PCB exposure data (Skaare *et al*., 2002). The results indicated that population status and health of polar bears with very high PCB levels may be at some degree of risk from PCBs. Oskam *et al*. (2001) found a significant negative correlation between PCBs and testosterone in plasma, indicating that PCBs may decrease circulating testosterone levels in male polar bears. The significance of these findings of negative associations between PCBs and retinol, thyroid hormones, testosterone and IgG, for the health of the individual and/or population is, however, ultimately difficult to interpret.

An EU funded study known as BIOCET (BIOaccumulation of persistent organic pollutants in small CETaceans in European waters: transport pathways and impact on reproduction; http://www.abdn.ac.uk/biocet/) investigated the potential impact of POPs on reproduction in female small cetaceans, pooling data from harbour porpoises and common dolphins found stranded in many countries around western Europe. Factors such as geographic variation in POP burdens, and relationships between POP burdens and age, fatty acid profiles, health status and reproduction, were taken into account within the analysis. The most important variable explaining POP profiles in common dolphin blubber was individual feeding history, while those in porpoises were more strongly related to individual condition (Pierce *et al*., 2008). A substantial proportion of individuals in the BIOCET sample had contaminant levels above the threshold 17 mg/kg PCB lipid weight (Kannan *et al*., 2000) that has been reported to have adverse health effects - based on experimental studies of both immunological and reproductive effects in seals, otters, and mink. This threshold was frequently exceeded in both porpoises (47% of individuals) and common dolphins (40%), especially porpoises from the southern North Sea (74%) and common dolphins inhabiting waters off the French coast (50%).

A follow up study was undertaken by Murphy *et al*. (2009; in press a) which analysed data from the BIOCET study and also from UK common dolphins (control group study) and harbour porpoises. Results suggested that high contaminant burdens, above the threshold level, were not inhibiting ovulation, conception or implantation in UK female common dolphins or harbour porpoises, though the impact on the foetal survival rate (in both species) requires further investigation. Based on these results and as pinnipeds experience delayed implantation/embryonic diapause, Murphy *et al*. (in press a) suggested that pinnipeds may be more vulnerable than cetaceans at the implantation stage of the reproductive cycle. Further studies on mink reported that although PCBs impair reproduction, ovulation, conception and implantation occurs - and foetuses died during gestation or shortly after birth (Murphy *et al*., in press a; and ref. therein). Recently, an ovotestis (i.e. a true hermaphrodite) was reported in a UK female common dolphin, however no contaminant analyses, cytogenetic investigations or hormone evaluations have been undertaken on this individual to date (Murphy *et al*. in press b). Within the BIOCET harbour porpoise sample, it was suggested that the casual immunotoxic relationship reported in Jepson *et al*. (2005) may have masked any direct affects of POPs, through lowering immunity, on reproductive activity (Murphy *et al*., in press a). The effects of contaminants on the North-east Atlantic continuous system harbour porpoise population is a particular concern, considering the low pregnancy rates reported for this population. If contaminants have an adverse effect on individual reproductive capabilities, the population would be more vulnerable to exploitation than is normally assumed, especially from other anthropogenic activities such as incidental capture, and would not necessarily recover from exploitation in a predictable way (Murphy *et al*., in press a).

Harbour seals in the Wadden Sea experienced some of the highest exposure levels of organochlorine contaminants in Europe (Reijnders, 1980) and it was hypothesized that the 1988 PDV epidemic selected against those seals with the highest contaminant loads. The significantly higher population growth rate following the first outbreak (Reijnders *et al*., 1997) coupled with the reduced levels of contaminants in the surviving population might support this view (Aguilar *et al*., 2002) although other explanations are equally plausible such as density-dependent effects (Hall *et al*., 2006b). Higher levels of PCBs in the UK Wash population were found in harbour seals that died of PDV, compared with those that survived the PDV epizootic (Hall *et al*., 2006b). Most European harbour seal populations recovered rapidly after the 1988 and 2002 epizootics (Reijnders *et al*., 1997; Hall *et al*., 2006b), apart from UK waters (Lonergan *et al*., 2007), suggesting that contaminants probably had a limited impact on the recovery of most harbour seal populations impacted by both the 1998 and 2002 phocine distemper epizootics.

The brominated flame retardants (such as PBDEs) are being reported as potential endocrine disrupting compounds but levels are still considerably lower than PCBs in most oceanic regions. Hall *et al*. (2003) found a correlation between PBDEs and thyroid hormone levels in grey seals during their first year of life and in adult harbour seals (Hall and Thomas, 2007) but it is still unclear whether this relationship is causal. One particular flame retardant, hexabromocyclododecane is causing some concern, as it has found at relative high levels in the blubber of harbour porpoise stranded along the Irish sea coast, where levels were an order of magnitude higher (~3 $\mu g\ g^{-1}$ lipid) than elsewhere except the North-west coast of Scotland where levels were ~5 μg/g lipid (Zegers *et al*., 2005). Levels of HBCD were increasing in harbour porpoises stranded and bycaught throughout the UK (Law *et al*., 2006a) but a more recent study reported a significant downturn in concentrations of HBCD in the blubber of harbour porpoises in the UK from 2003 onwards (Law *et al*., 2008a).

5.3.2 Heavy metals

Of the toxic elements studied those of most concern are mercury, cadmium, lead and zinc. Marine mammals have evolved detoxification mechanisms to mitigate naturally high dietary exposure to trace metals (e.g. Augier *et al*., 1993; Cuvin-Aralar and Furness, 1991) and there is currently little compelling evidence of significant health or population effects from exposure to metals. Further, there is no evidence of increasing concentrations of metals in the marine environment.

Mercury can bioaccumulate through the food chain and is a well-recognized neurotoxin. Its interaction with selenium appears to be protective and various laboratory studies have shown that toxic effects of mercury were prevented or reduced by simultaneous exposure to selenium (Cuvin-Aralar and Furness, 1991). Some of the concentrations of mercury in the liver of marine mammals have exceeded those known to be toxic to other mammals but lethal effects have not been observed (Britt and Howard, 1983). Marine mammals seem able to metabolize mercury from its toxic methyl form found in fish. Although marine mammals can tolerate high concentrations of mercury immobilized as the selenide, methylmercury poisoning has been reported in a ringed seal from an area of heavy industrialization (Helminen *et al*., 1968). Elevated Hg (and Hg:Se) levels were observed in the livers of UK-stranded harbour porpoises that died of infectious disease (as compared with a control group that died of physical trauma) (Bennett *et al*., 2001; Jepson 2005), but this may represent redistribution of methymercury from muscle to the liver during disease-related loss of nutritional status (Das *et al*., 2004; Jepson, 2005). The recent study by Pierce *et al*. (2008) found highest levels of mercury in the liver samples from common dolphins stranded along the French coast, but these were not at concentrations high enough to cause concern.

Cadmium can sometimes be found at high concentrations in the livers of marine mammals (Law *et al*., 1991), but there does not appear to be any published information on cadmium-induced pathology in marine mammals. These high levels are probably due to naturally high cadmium concentrations in prey species such as squid (Bustamante *et al*., 1998). Metallothionen sequestration appears to protect marine mammals from cadmium toxicity.

Lead is also found in many marine mammal tissues, particularly liver and kidney, but not at concentrations that are cause for concern (Law *et al*., 1991). Bone is a long-term storage target organ for lead, although again no associated histopathological lesions have been reported. Smith *et al*. (1990) used isotopic ratios to show that the source of lead in some marine mammal species has shifted from naturally derived lead to anthropogenic aerosol-dominated forms.

Copper is an essential dietary element for mammals and a wide range of concentrations has been reported in marine mammals. In the UK levels of between 3 and 30 mg/kg have been measured in the liver of stranded animals and it has been suggested that this may represent the normal range of homeostatic control in marine mammals (Law, 1996).

Pillet *et al*. (2000) found that zinc exposure affected the phagocytic response of seal white cells *in vitro* and that this response differed between the sexes. Kakuschke *et al*. (2005) reported that a small number of harbour seals appeared to be hypersensitised to a number of heavy metals. Whereas there are few studies that show major impacts of heavy metals, it is possible that they may have combined effects as they often co-occur with the persistent organic contaminants. In the BIOCET study, zinc and other heavy metals were the best (significant) explanatory variables for concentrations of POPs in the blubber tissue of harbour porpoises (Pierce *et al*. 2008). High concentrations of Zn have previously been associated with poor health in harbour porpoises in European waters and may represent a physiological response to stress and/or disease rather than providing evidence of Zn-induced toxicosis (Das *et al*., 2004).

5.3.3 Polyaromatic hydrocarbons

Polyaromatic hydrocarbons have rarely been studied in the tissues of marine mammals but where measurements have been obtained from muscle tissue, liver and blubber, all generally were below 1 μg/g. Law and Whinnett (1992) investigated PAHs in the muscle tissue of harbour porpoises stranded around the UK coast and found total PAH concentrations ranging from 0.11–0.56 μg/g wet weight and 0.47–2.4 μg/g wet weight Ekofisk crude oil equivalents. Specific PAHs were 2–4 ring compounds (naphthalenes, phenanthrenes, anthracene, fluoranthene and pyrene). Bond (1993) found similar compounds in the blubber of seals from the Moray Firth. The PAH levels displayed large variations, with grey seals having higher levels than harbour seals; mean 15.78 (SD 25.54) μg/g dry weight in grey seals, and 2.67 (SD 5.77) μg/g dry weight in harbour seals.

The effects of PAHs on marine mammals are reviewed in Geraci and St Aubin (1990) and various responses from effects on the central nervous system, eyes and mucous membranes, thermal regulatory effects from fouling of fur, to induction of metabolic enzyme systems and effects on hormone levels were reported. These effects are largely observed following short-term acute exposure. Less is known about the effects of long-term chronic exposure. Although studies have shown that fish readily convert aromatic hydrocarbons to metabolites such as dihydrodiols and phenols (Krahn *et al*., 1984) and therefore fish-eating mammals may receive lower doses of parent PAHs, and cetaceans which feed lower down the food chain are likely to be most at risk. In addition Neale *et al*. (2002) assessed the effects of the prototypic polycyclic aromatic hydrocarbon (PAH), benzo[a]pyrene (B[a]P), and two polychlorinated biphenyls (PCBs), CB-156 and CB-80, on the T-cell proliferative response to mitogen in harbour seal peripheral lymphocytes. They found a suppressive effect of B[a]P (10 μm) exposure on T cell mitogenesis. Exposures to 10 μm CB-156 and CB-80, and 1.0 and 0.1 μm B[a]P, did not produce significant depression in lymphocyte proliferation. Exposure to the model PAH at 10 μm resulted in a 61% (range 34–97%) average reduction in lymphocyte proliferation and they hypothesize that extensive exposure of PAHs by some marine mammals affects their cell-mediated immunity against viral pathogens.

The carcinogenic nature of certain PAHs, such as benzo(a)pyrene has been a concern. For example, Beland *et al*. (1993) reported the detection of benzo(a)pyrene adducts in DNA from beluga whales in the Gulf of St Lawrence and high prevalence of tumours have been detected in this species (De Guise *et al*., 1995a; De Guise *et al*., 1994b). Outside the St Lawrence Estuary there is little evidence of the substantial effects of exposure to PAHs in marine mammals. For example, in UK waters one of 27 UK harbour porpoises examined by Law and Whinnett (1992) between 1988 and 1991 was considered to have died as a result of a tumour and only 10/1710 fatal tumours have been detected in harbour porpoises in 20 years of detailed pathological investigation on UK-stranded cetaceans between 1990–2009 (Jepson, 2005; UK CSIP pathology database).

5.3.4 Butyl tins (Tributyl tin (TBT), Dibutyl tin (DBT) and Monobutyl tin (MBT))

Butyl tin compounds, largely tri- and di-butyl tin, have now been reported in the liver and blubber of in grey seals and harbour porpoises (Law *et al*., 1998) pelagic cetaceans in UK waters (Law *et al*., 1999). Levels in UK harbour porpoises were relatively low and largely unassociated with health effects such as infectious disease mortality (Jepson, 2005). Few other reports on their effects have been published in marine mammals.

5.4 Impacts of contaminants at the population level (all species/regions)

5.4.1 ICES regions with high environmental pollutant exposure

As mentioned earlier, cetaceans inhabiting coastal waters, estuaries and bays may be at a greater risk from pollution, due to their proximity to the outflow of industrial waste. Belugas and harbour seals in St Lawrence Estuary, Canada have been exposed to high levels of a range of contaminants (De Guise *et al*., 1995a; De Guise *et al*., 1994b; Hickie *et al*., 2000). The St Lawrence beluga population has not recovered after years of hunting finally ceased in 1985 (declined from several thousand animals). Instead it has remained relatively stable at around 1100 individuals in recent years (Hamill *et al*., 2007). In this population, a range of diseases have been linked to high contaminant exposures including PCBs and metals and a high prevalence of tumours linked to polycylic aromatic hydrocarbons (PAHs) such as benzo[a]pyrene (De Guise *et al*., 1995a; De Guise *et al*., 1994b; Martineau *et al*., 1994). Levels of these contaminants are now declining in St Lawrence belugas (Lebeu *et al*., 2007).

Pinnipeds (especially grey and ringed seals) are still markedly depleted in the Baltic Sea compared with estimated population sizes for the early 20th Century (HELCOM, online[2]). Previous studies suggested that chemical pollutants such as PCBs, and hunting, were drivers of these declines (Bergman, 1999; Bergman and Olsson, 1985). Although the contaminant levels in Baltic seals have decreased since the end of the 1970s, the levels in Baltic seals are still relatively high compared with seals living in unpolluted waters, especially in ringed seals - differences in levels of toxins between grey and ringed seals could be explained by differences in their diets (ICES WGMME 2005). In recent years, it has been reported that the grey seal population in the Baltic Sea is recovering and growing up to 10%/year (HELCOM, online[3]). Whereas, based

2 www.helcom.fi/environment2/biodiv/endangered/Mammals/en_GB/Halichoerus_grypus/

3www.helcom.fi/environment2/biodiv/endangered/Mammals/en_GB/Phoca_hispida_botnica/

on data from 1988 to 2006, the annual rate of increase for the ringed seal management unit/population in Bothnian Bay was only 4.3%, which is less than half the intrinsic capacity (Karlsson *et al*., 2007). In Gulf of Riga and the Gulf of Finland the rate of increase for ringed seals was zero between 1996 and 2003 (Karlsson *et al*., 2007). Ringed seals and, to a lesser degree grey seals, still have lesions consistent with what is tentatively called Baltic Seal Disease Complex (BSDC) although the prevalence of lesions are decreasing as concentrations of chemical pollutants in Baltic fish are now declining (Bergman, 2007) Helle *et al*. (2005) reported that uterine occlusions still affect roughly 20% of the adult female ringed seals. In contrast, harbour seals in the Baltic Sea appear more resilient to BSDC and, although Baltic harbour seals suffered two phocine distemper epizootics, the population fully recovered from the 1988 event and is recovering from the 2002 event (Bergman, 2007).

Harbour porpoises in the Baltic Sea have also declined markedly since the 1960s (Berggren *et al*., 1999) but a current abundance estimate for the harbour porpoise population in the Baltic is lacking. In the southern Baltic Proper, a mean abundance of 599 porpoise groups was estimated in June 1995 (Hiby and Lovell, 1996, cited in Berggren *et al*., 2004). This survey was repeated in 2002 resulting in a mean estimate of 93 porpoises (Berggren *et al*., 2004). These survey results confirm the extremely low and probably decreasing population abundance in the Baltic Proper. For the Skagerrak, Belt and the Arkona Seas, the mean abundance of harbour porpoises was estimated to be about 36 000 animals in July 1994 (SCANS-I; Hammond *et al*., 2002) and about 23 000 individuals in July 2005 (SCANS-II 2008) - though it should be noted that the design blocks surveyed in 1994 and 2005 were not the same, and therefore these estimates are not directly comparable.

5.4.2 ICES species with high pollutant exposure

Killer whales have the highest persistent bioaccumulative pollutant levels of all the marine mammals (Ross *et al*., 2000). Most killer whale populations that have been assessed (mainly in NE Pacific) are small and stable or declining (COSEWIC, 2008). It has been suggested that there may be fewer than 1000 mature individuals in the North-west Atlantic/eastern Arctic population, and possibly even less than 250 mature individuals (COSEWIC 2008). This population's small size and the species' life history and social attributes justify its conservation status designation as “Special Concern” (COSEWIC, 2008). Further, killer whales in North-east Atlantic could be negatively impacted by PCBs at the population level throughout their entire range. Due to the lack of abundance estimates for killer whales in the North-east Atlantic, any population level declines due to PCBs, and other factors, would be largely undetected.

A similar population level effect could be occurring in some inshore bottlenose dolphin populations with high PCB exposure in the North-east Atlantic (e.g. North-west France; Sado Estuary, Portugal). Current abundance estimates for bottlenose dolphins are good in many North-east Atlantic regions (e.g. UK) but may be insufficiently robust in some regions to detect small population level changes that might be attributed to contaminants or other drivers.

In one study of the Sarasota Bay resident bottlenose dolphin population, 47 blubber samples collected during June 2000 and 2001 were analysed for PCB concentrations of 22 congeners relative to life-history factors and reproductive success (Wells *et al*., 2005). First-born calves had higher concentrations than subsequent calves of similar age (>25vs.<25 ppm). Although female bottlenose dolphins have been reported to offload 80% of their contaminant load during the first seven weeks of lactation (Cockcroft *et al*., 1989), maternal burdens were lower in the Sarasota population early in lactation and increased as calves approached nutritional independence. Empirical data were generally consistent with a published theoretical risk assessment and supported the need for incorporation of threats from indirect anthropogenic impacts such as environmental pollutants into species management plans. Toxicological effects such as high first calf mortality have also been predicted in Sarasota and other US bottlenose dolphin populations (Hall *et al*., 2006c; Schwacke *et al*., 2002).

Potential effects of PCBs in individual harbour porpoises in European waters have been identified including immunosuppression (Beineke *et al*., 2005) and mortality due to infectious disease (Jepson *et al*., 2005; Hall *et al*., 2006a). At the population level, however, the SCANS-II estimate (July 2005) for harbour porpoises in the North Sea was not significantly different from the first SCANS survey conducted in 1994 (Hammond *et al*., 2002). A marked decline in the highly contaminated Baltic Sea harbour porpoise population has been recorded since the 1960s although levels of most pollutants in the Baltic Sea are now declining.

5.5 Conclusions

1) Despite being banned for two-three decades, polychlorinated biphenyls still occur at concentrations that exceed proposed thresholds for mammalian toxicity (e.g. Kannan *et al*., 2000; Jepson *et al*., 2005) in some marine mammal top predator species including bottlenose dolphins, killer whales and polar bears.

2) Compared with many other legacy pollutants, PCBs are declining only very slowly in many geographic regions (e.g. harbour porpoises in UK waters).

3) Given their high exposure levels in marine mammals (compared with proposed toxicity thresholds for marine mammals), resistance to environmental degradation and relative toxicity, PCBs undoubtedly continue to pose the greatest toxicological threat to some marine mammal species within the ICES range.

5.6 Recommendations

1) In order to better detect future contaminant-related population level effects, there is a need for more robust population estimates for some marine mammal populations with low abundance and high pollutant (esp. PCB) exposure (e.g. killer whales and bottlenose dolphins).

2) Research should be continued and expanded to assess trends in contaminant exposure (PCBs and newer contaminants), population structure and to conduct risk assessments for health and reproductive effects from contaminant exposure in species of highest risk (e.g. killer whales, St Lawrence belugas, polar bears, bottlenose dolphins, and Baltic marine mammals). The use of biopsy techniques would allow for simultaneous sampling for genetics and contaminant exposure.

3) Contaminant levels (including PCBs) should continue to be monitored in marine mammals (or marine fish) in regions of highest environmental exposure (Baltic Sea and St Lawrence Estuary).

4) Closer standardization of stranding network protocols for conducting necropsies, storing samples and conducting contaminant analyses across the ICES range would be beneficial.

5) Better integration of data on health status and contaminant exposure within the ICES range would help assess potential long-term impacts of chemical contaminants in regions and species with highest exposures (e.g. establishment of European strandings/live biopsy database and tissue bank).

5.7 References

Aguilar, A., and Borrell, A. 1994a. Abnormally high polychlorinated biphenyl levels in striped dolphins (*Stenella coeruleoalba*) affected by the 1990–1992 Mediterranean epizootic. Science of the Total Environment 154(2-3): 237–247.

Aguilar, A., and Borrell, A. 1994b. Reproductive transfer and variation of body load of organochlorine pollutants with age in fin whales (*Balaenoptera physalus*). Archives of Environmental Contamination and Toxicology 27(4): 546–554.

Aguilar, A., and Borrell, A. 1997. Marine Mammals and Pollutants - An annotated bibliography. Barcelona, Fundacio pal Desenvolupament Sostenible.

Aguilar, A., Borrell, A., and Pastor, T. 1999. Biological factors affecting variability of persistent pollutant levels in cetaceans. Journal of Cetacean Research and Management Special Issue 1: 83–116.

Aguilar, A., Borrell, A., and Reijnders, P.J.H. 2002. Geographical and temporal variation in levels of organochlorine contaminants in marine mammals. Marine Environmental Research 53(5): 425–452.

Allchin, C.R., Law, R.J., and Morris, S. 1999. Polybrominated diphenylethers in sediments and biota downstream of potential sources in the UK. Environmental Pollution 105: 197–207.

AMAP. 2004. AMAP Assessment 2002: Persistent Organic Pollutants in the Arctic. Arctic Monitoring and Assessment Programme (AMAP) Oslo, Norway. xvi+310 pp.

Amaral Mendes, J.J. 2002. The endocrine disrupters: a major medical challenge. Food and Chemical Toxicology 40(6): 781–788.

Ames, A.L. and Van Vleet, E.S. 1996. Organochlorine residues in the Florida manatee, *Trichechus manatus latirostris* Marine Pollution Bulletin 32, 374–377.

AMAP. 2002. AMAP Assessment 2002: Persistent Organic Pollutants in the Arctic. Arctic Monitoring and Assessment Programme (AMAP) Oslo, Norway. xvi+310 pp.

Aulerich, R.J. and Ringer, R.K. 1977. Current status of PCB toxicity to mink and effect of their reproduction. Arch. Environ. Contam. Toxicol. *5:* 279–292.

Augier, H., Benkoel, L., Brisse, J., Chamlian, A., and Park, W.K. 1993. Microscopic localization of mercury-selenium interaction products in liver, kidney, lung and brain of Mediterranean striped dolphins (*Stenella Coeruleoalba*) by silver enhancement kit. Cellular and Molecular Biology (Noisy-le-Grand, France) 39(7): 765–772.

Bang, K., Jenssen, B.M., Lydersen, C., and Skaare, J.U. 2001. Organochlorine burdens in blood of ringed and bearded seals from north-western Svalbard. Chemosphere 44(2): 193–203.

Beineke, A., Siebert, U., McLachlan, M., Bruhn, R., Thron, K., Failing, K., Müller, G., and Baumgärtner, W. 2005. Investigations of the potential influence of environmental contaminants on the thymus and spleen of harbour porpoises (*Phocoena phocoena*). Environ Sci Technol. 39: 3933–3938.

Beland, P., De Guise, S., Girard, C., Legace, A., Martineau, D., Michaud, R., Muir, D.C.G., Norstrom, R.J., Pelletier, E., Ray, S., and Shugart, L.R. 1993. Toxic compounds and health and reproductive effects in St Lawrence beluga whales. Jnl. Gt. Lakes Res. 19(4): 766–775.

Bennett, P.M., Jepson, P.D., Law, R.J., Jones, B.R., Kuiken, T., Baker, J.R., Rogan, E., and Kirkwood, J.K. 2001. Exposure to heavy metals and infectious disease mortality in harbour porpoises from England and Wales. Environmental Pollution 112(1): 33–40.

Berge, J., Brevik, E., Bjorge, A., Folsvik, N., Gabrielsen, G., and Wolkers, H. 2004. Organotins in marine mammals and seabirds from Norwegian territory. Journal of Environmental Monitoring 6: 108–112.

Berggren, P., Ishaq, R., Zebuhr, Y., Naf, C., Bandh, C., and Broman, D. 1999. Patterns and levels of organochlorines (DDTs, PCBs, non-ortho PCBs and PCDD/Fs) in male harbour porpoises (*Phocoena phocoena*) from the Baltic Sea, the Kattegat-Skagerrak Seas and the west coast of Norway. Marine Pollution Bulletin 38(12): 1070–1084.

Berggren, P., Hiby, L., Lovell, P., and Scheidat, M. 2004. Abundance of harbour porpoises in the Baltic Sea from aerial surveys conducted in summer 2002. 16pp. Paper SC/56/SM7 submitted to the Scientific Committee of the International Whaling Commission. Available from www.iwcoffice.org.

Bergman, A. 1999. Health condition of the Baltic grey seal (*Halichoerus grypus*) during two decades. Gynaecological health improvement but increased prevalence of colonic ulcers. Acta Pathology Microbiology and Immunology of Scandinavia 107: 886.

Bergman, A. 2007. Pathological Changes in Seals in Swedish Waters: The Relation to Environmental Pollution. PhD thesis. Department of Biomedical Sciences and Veterinary Public Health Uppsala.

Bergman, A., and Olsson, M. 1985. Pathology of Baltic gray seal and ringed seal females with special reference to adrenocortical hyperplasia: is environmental pollution the cause of a widely distributed disease syndrome? Finn. Game. Res. 44: 47–62.

Bergman, A., Olsson, M., and Reiland, S. 1992. Skull-bone lesions in the Baltic Grey Seal (*Halichoerus grypus*). Ambio 21(8): 517–519.

Bernhoft, A., Skaare, J.U., Wiig, O., Derocher, A.E., and Larsen, H.J.S. 2000. Possible immunotoxic effects of organochlorines in polar bears (*Ursus maritimus*) at Svalbard. Journal of Toxicology and Environmental Health, Part A: Current Issues 59(7): 561–574.

Bernt, K.E., Hammill, M.O., Lebeuf, M., and Kovacs, K. 1999. Levels and patterns of PCBs and OC pesticides in harbour and grey seal from the St Lawrence Estuary, Canada. The Science of the Total Environment 243–244: 243–262.

Berrow, S.D., Long, S.C., McGarry, A.T., Pollard, D., Rogan, E., and Lockyer, C. 1998. Radionuclides (Cs-137 and K-40) in harbour porpoises *Phocoena phocoena* from British and Irish coastal waters. Marine Pollution Bulletin 36(8): 569–576.

Bignert, A., Olsson, M., Persson, W., Jensen, S., Zakrisson, S., Litzén, K., Eriksson, U., Häggberg, L. and Alsberg, T. 1998. Temporal trends of organochlorines in northern Europe, 1967–1995. Relation to global fractionation, leakage from sediments and international measures. *Environmental Pollution 99,* 177–198.

Bignert A., Nyberg E., Sundqvist K. L., and Wiberg K. 2007. Spatial and seasonal variation in concentrations and patterns of the PCDD/F and dioxin-like-PCB content in herring from the northern Baltic Sea. J. Environ. Monit. DOI:10.1039/b700667e

Blomkvist, G., Roos, A., Jensen, S., Bignert, A., and Olsson, M. 1992. Concentrations of DDT and PCB in seals from Swedish and Scottish waters. Ambio 21(8): 539–545.

Bond, P. 1993. Determination of petroleum contamination in common and grey seals from eastern Scotland, School of Natural and Environmental Sciences, Coventry, UK.

Borrell, A., Bloch, D., and Desportes, G. 1995. Age trends and reproductive transfer of organochlorine compounds in long-finned pilot whales from the Faroe Islands. Environmental Pollution 88(3): 283–292.

Bowles, D. 1999. An overview of the concentrations and effects of metals in cetacean species. Journal of Cetacean Research and Management (Special Issue 1): 125–148.

Breivik K., Sweetman, A., Pacyna, J.M., and Jones, K.C. 2007. Towards a global historical emission inventory for selected PCB congeners — a mass balance approach. 3. An update. Sci Total Environ 377:296–307.

Britt, J.O., and Howard, E.B. 1983. Tissue residues of selected environmental contaminants in marine mammals. *In* Pathobiology of marine mammal diseases Vol II. *Edited by* E.B. Howard. CRC Press, Boca Raton, FL. pp. 79–94.

Bustamante, P., Caurant, F., Fowler, S.W., and Miramand, P. 1998. Cephalopods as vectors for the transfer of Cadmium to top marine predators in the north-east Atlantic Ocean. The Science of the Total Environment 220: 71–80.

Cockcroft, V.G., DeKock, A., Lord, D., and Ross, G. 1989. Organochlorines in bottlenose dolphins Tursiops truncatus from the east coast of South Africa. S Afr J mar Sci 8: 207–217.

COSEWIC. 2008. COSEWIC assessment and update status report on the Killer Whale Orcinus orca, Southern Resident population, Northern Resident population, West Coast Transient population, Offshore population and Northwest Atlantic / Eastern Arctic population, in Canada. Committee on the Status of Endangered Wildlife in Canada (COSEWIC). Ottawa. viii + 65 pp. (www.sararegistry.gc.ca/status/status_e.cfm).

Covaci, A., Van de Vijver, K., DeCoen, W., Das, K., Bouquegneau, J.M., Blust, R., and Schepens, P. 2002. Determination of organohalogenated contaminants in liver of harbour porpoises (*Phocoena phocoena*) stranded on the Belgian North Sea coast. Marine Pollution Bulletin 44(10): 1157–1165.

Cuvin-Aralar, L.A., and Furness, R.W. 1991. Mercury and selenium interaction: A review. Ecotoxicol. and Environ. Safety 21: 348–364.

Das, K., Siebert, U., Fontaine, M., Jauniaux, T., Holsbeek, L., and M., B.J. 2004. Ecological and pathological factors related to trace metal concentrations in harbour porpoises *Phocoena phocoena* from the North Sea and adjacent areas. Marine Ecology Progress Series 281: 283–295.

Dam, M., and Bloch, D. 2000. Screening of mercury and persistent organochlorine pollutants in long-finned pilot whale (*Globicephala melas*) in the Faroe Islands. Marine Pollution Bulletin 40(12): 1090–1099.

Das, K., Vossen, A., Tolley, K., VÃkingsson, G., Thron, K., MÃ¼ller, G., BaumgÃrtner, W., and Siebert, U. 2006. Interfollicular Fibrosis in the Thyroid of the Harbour Porpoise: An Endocrine Disruption? Archives of Environmental Contamination and Toxicology V51(4): 720–729.

De Guise, S., Lagace, A., and Beland, P. 1994a. Tumors in St Lawrence beluga whales (*Delphinapterus leucas*). Vet. Pathol. 31(4): 444–449.

De Guise, S., Lagacé, A., and Béland, P. 1994b. Tumors in St Lawrence beluga whales (*Delphinapterus leucas*). Vet. Pathol. 31: 444–449.

De Guise, S., Flipo, D., Boehm, J.R., Martineau, D., Beland, P., and Fournier, M. 1995a. Immune functions in beluga whales (*Delphinapterus leucas*): Evaluation of phagocytosis and respiratory burst with peripheral blood leukocytes using flow cytometry. Veterinary Immunology and Immunopathology 47(3–4): 351–362.

De Guise, S., Martineau, D., Béland, P. and Fournier, M. 1995b. Possible mechanisms of action of environmental contaminants on St Lawrence beluga whales (*Delphinapterus leucas*). Environmental Health Perspectives 103: 73–77

De Guise, S., Martineau, D., Beland, P., and Fournier, M. 1998 Effects of in vitro exposure of beluga whale leukocytes to selected organochlorines. Journal of Toxicology and Environmental Health-Part A 55: 479–493.

de Swart, R.L., Ross, P.S., Vedder, L.J., Boink, P.B.T.J., Reijnders, P.J.H., Mulder, P.G.H., and Osterhaus, A.D.M.E. 1995. Haematology and clinical chemistry values for harbour seals (*Phoca vitulina*) fed environmentally contaminated herring remain within normal ranges. Canadian Journal of Zoology - Revue Canadienne de Zoologie 73(11): 2035–2043.

de Wit, C.A. 2002. An overview of brominated flame retardants in the environment. Chemosphere 46: 583–624.

Debier, C., Pomeroy, P.P., Dupont, C., Joiris, C., Comblin, V., Le Boulenge, E., Larondelle, Y., and Thome, J.P. 2003. Quantitative dynamics of PCB transfer from mother to pup during lactation in UK grey seals *Halichoerus grypus*. Marine Ecology Progress Series 247: 237–248.

DeLong, R.L., Gilmartin, W.G., and Simpson, J.G. 1973. Premature births in California Sea Lions: Association with high organochlorine pollutant residue levels. Science 181: 1168–1170.

De Swart, R.L., Ross, P.S., Vedder, L.J., Timmerman, H.H., Heisterkamp, S., Van Loveren, H., Vos, J.G., Reijnders, P.J.H., and Osterhaus, A.D.M.E. 1994. Impairment of immune function in harbour seals (*Phoca vitulina*) feeding on fish from polluted waters. Ambio 23: 155–159.

Dietz, R., Riget, F., and Johansen, P. 1996. Lead, cadmium, mercury and selenium in Greenland marine animals. Science Of The Total Environment 186: 67–93.

Erickson, M.D. 1986. Physical, chemical, commercial, environmental, and biological properties. Analytical Chemistry of PCBs. Boston, MA: Butterworth Publishers; p. 1–53.

Fair, P.A., Adams, J., Mitchum, G., Hulsey, T.C., Reifd, J.S. Houde, M., Muir, D., Wirth, E., Wetzel, D., Zolman, E., McFee, W., and Bossart G.D. 2010. Contaminant blubber burdens in Atlantic bottlenose dolphins (*Tursiops truncatus*) from two southeastern US estuarine areas: Concentrations and patterns of PCBs, pesticides, PBDEs, PFCs, and PAHs, Sci Total Environ, doi:10.1016/j.scitotenv.2009.12.021.

Fuller, G.B., and Hobson, W.C. 1986. Effect of PCBs on reproduction in mammals. *In* PCBs and the Environment Vol II. *Edited by* J.S. Waid. CRC Press, Boca Raton, Florida. pp. 101–125.

Gabrielsen, G.W., and Henriksen, E.O. 2001. Persistent organic pollutants in Arctic animals in the Barents Sea area and Svalbard: Levels and Effects. Memoirs of National Institute of Pol Res, Special Issue, 54:349–364.

Gabrielsen G.W. 2007. Levels and effects of persistent organic pollutants in arctic animals. In "Arctic-Alpine Ecosystems and People in a Changing Environment" (Orbaek, J.B., Kallenborn, R., Tombre, I., Hegseth, E.N., Falk-Petersen, S. and A. H. Hoel A.H. (eds)). Springer Verlag, Berlin, Chapter 20, page 377–412.

Gaydos, J.K., Balcomb, K.C., Osborne, R.W., and Dierauf, L. 2004. Evaluating potential infectious disease threats for southern resident killer whales, *Orcinus orca*: a model for endangered species. Biological Conservation 117(3): 253–262.

Gebbink, W.A., Sonne, C., Dietz, R., Kirkegaard, M., Riget, F.F., Born, E.W., Muir, D.C.G., and Letcher, R.J. 2008. Tissue-specific congener composition of organohalogen and metabolite contaminants in East Greenland polar bears (*Ursus maritimus*). Environmental Pollution 152(3): 621–629.

Geraci, J.R., and St Aubin, D.J. 1990. Sea mammals and oil: confronting the risks. Academic Press, San Diego.

Hall, A.J., Law, R.J., Wells, D.E., Harwood, J., Ross, H., Kennedy, S., Allchin, C.R., Campbell, L.A., and Pomeroy, P.P. 1992a. Organochlorine levels in common seals (*Phoca vitulina*) which were victims and survivors of the 1988 phocine distemper epizootic. Science of the Total Environment 115: 145–162.

Hall, A.J., Pomeroy, P.P., and Harwood, J. 1992b. The descriptive epizootiology of phocine distemper in the UK during 1988/89. Science of the Total Environment 115: 31–44.

Hall, A.J. 2001. Organohalogenated contaminants in marine mammals. *In* Marine Mammals: Biology and Conservation. *Edited by* P. Evans and A. Raga. Kluwer Academic/Plenum Publishers, London. pp. 523–563.

Hall, A., Kalantzi, O., and Thomas, G. 2003. Polybrominated diphenyl ethers (PBDEs) in grey seal pups during their first year of life - are they thyroid hormone endocrine disrupters? Environmental Pollution 126: 29–37.

Hall, A.J., Hugunin, K., Deaville, R., Law, R.J., Allchin, C.R., and Jepson, P.D. 2006a. The risk of infection from polychlorinated biphenyl Bxposure in the harbour porpoise (*Phocoena phocoena*): A Case–Control Approach Environmental Health Perspectives 114(5): 704–711.

Hall, A.J., Jepson, P.D., Goodman, S.J., and Harkonen, T. 2006b. Phocine Distemper Virus in the North and European seas - Data and models, nature and nurture. Biological Conservation 131: 221–229.

Hall, A.J., McConnell, B.J., Rowles, T.K., Aguilar, A., Borrell, A., Schwacke, L., Reijnders, P.J.H., and Wells, R.S. 2006c. Individual-Based Model Framework to Assess Population Consequences of Polychlorinated Biphenyl Exposure in Bottlenose Dolphins. Environmental Health Perspectives 114(Suppl 1): 60–64.

Hall, A.J., McConnell, B.J., Rowles, T.K., Aguilar, A., Borrell, A., Schwacke, L., Reijnders, P.J.H., and Wells, R.S. 2006d. Population consequences of polychlorinated biphenyl exposure in bottlenose dolphins - an individual based model approach. Environmental Health Perspectives 114(Suppl. 1): 60–64.

Hall, A.J., and Thomas, G.O. 2007. Polychlorinated biphenyls, DDT, polybrominated diphenyl ethers and organic pesticides in UK harbour seals - mixed exposures and thyroid homeostasis. Environmental Toxicology and Chemistry 26(26): 851–861.

Hamanaka, T., Itoo, T., and Mishima, S. 1982. Age-related change and distribution of cadmium and zinc concentrations in the Steller sea lion (*Eumetopias jubata*) from the coast of Hokkaido, Japan. Marine Pollution Bulletin 13: 57–61.

Hammill, M.O., L.N. Measures, J.-F. Gosselin, V. Lesage. 2007. Lack of Recovery in St Lawrence Estuary beluga. Research Document 2007/026. Canadian Science Advisory Secretariat (CSAS).

Hammond, J.A., Hall, A.J. and Dyrynda, E.A. 2005. Comparison of polychlorinated biphenyl (PCB) induced effects on innate immune functions in harbour and grey seals. Aquatic Conservation 74: 126–138.

Hammond, P.S., Berggren, P., Benke, H., Borchers, D.L., Collet, A., Heide-Jorgensen, M.P., Heimlich, S., Hiby, A.R., Leopold, M.F., and Oien, N. 2002. Abundance of harbour porpoise and other cetaceans in the North Sea and adjacent waters. Journal of Applied Ecology 39(2): 361–376.

Helle, E. 1976. PCB Levels Correlated with Pathological Changes in Seal Uteri. Ambio 5: 261–263.

Helle, E., Olsson, M., and Jensen, S. 1976. DDT and PCB levels and reproduction in ringed seal from Bothnian Bay. Ambio 5: 188–189.

Helle, E., Nyman, M and Stenman, O. 2005. Reproductive capacity of grey and ringed seal females in Finland. International conference on Baltic seals, 15–18 February Helsinki, Finland.

Helminen, M., Karppanen, E., and I., K.J. 1968. Mercury content of the ringed seal of Lake Saama. Suom. Laaklehti 74: 87–89.

Hickie, B.E., Kingsley, M.C.S., Hodson, P.V., Muir, D.C.G., Béland, P., and Mackay, D. 2000. A modelling-based perspective on the past, present, and future polychlorinated biphenyl contamination of the St Lawrence beluga whale (*Delphinapterus leucas*) population. Can. J. Fish. Aquat. Sci. 57: 101–112.

Hickie, B.E., R.W. Macdonald, J.K.B. Ford and P.S. Ross. 2007. Killer whales *Orcinus orca* face protracted health risks associated with lifetime exposure to PCBs. Environmental Science and Toxicology 41: 6613–6619.

Higdon, J. 2007. Status of knowledge on killer whales *Orcinus orca* in the Canadian Arctic. Canadian Science Advisory Secretariat Research Document 2007/ 048.

Hobbs, K.E., Muir, D.C.G., Born, E.W., Dietz, R., Haug, T., Metcalfe, T., Metcalfe, C., and Øien, N. 2003. Levels and patterns of persistent organochlorines in minke whale (*Balaenoptera acutorostrata*) stocks from the North Atlantic and European Arctic. Environmental Pollution 121(2): 239–252.

Hohn, A.A., Ewing, R.Y., and Zaias, J. 2007. Reproduction in relation to conservation and commercial exploitation. *In* Reproductive biology and phylogeny of cetacea. Volume 7 of series: Reproductive biology and phylogeny. *Edited by* D. Miller, L. Science Publishers, Enfield. pp. 371–389.

ICES WGMMH, and WGMMPD. 2000. Report of the Working Group on Marine Mammal Habitats and the Working Group on Marine Mammal Population dynamics and trophic interactions. Helsinki, Finland 28 February–3 March 2000.

ICES WGMMPD. 2001. Report of the Working Group on Marine Mammal Population Dynamics and Habitats. ICES Headquarters 23–27 April 2001.

ICES WGMME. 2005. ICES. Report of the Working Group on Marine Mammal Ecology. 9–12 May 2005, Savolinna, Finland.

Iwata, H., Tanabe, S., Miyazaki, N., and Tatsukawa, R. 1994. Detection of butyltin compound residues in the blubber of marine mammals. Marine Pollution Bulletin 28: 607–612.

Jensen, B. A. and Hahn, M. E. 2001. cDNA cloning and characterization of a high affinity aryl hydrocarbon receptor in a cetacean, the beluga, *Delphinapterus leucas*. *Toxicological Sciences* 64, 41–56.

Jensen, S., Johnels, A. G., Olsson, M. and Otterlind, G. 1969. DDT and PCB in marine animals from Swedish waters. *Nature 224*, 247–250.

Jepson, P.D. 2003. Pathology and toxicology of stranded harbour porpoises (*Phocoena phocoena*) in UK waters. PhD Thesis, Royal Veterinary College, University of London.

Jepson, P.D. (editor). 2005. Cetacean Strandings Investigation and Coordination in the UK 2000–2004. Final report to the *Department for Environment, Food and Rural Affairs.* pp 1–79. http://www.defra.gov.uk/wildlife-countryside/resprog/findings/index.htm.

Jepson, P.D., Bennett, P.M., Allchin, C.R., Law, R.J., Kuiken, T., Baker, J.R., Rogan, E., and Kirkwood, J.K. 1999. Investigating potential associations between chronic exposure to polychlorinated biphenyls and infectious disease mortality in harbour porpoises from England and Wales. The Science of The Total Environment 243–244: 339–348.

Jepson, P., Bennett, P.M., Deaville, R., Allchin, C.R., Baker, J.R., and Law, R.J. 2005. Relationships between polychlorinated biphenyls and health status in harbour porpoises (*Phocoena phocoena*) stranded in the UK Environmental Toxicology and Chemistry 24(1): 238–248.

Jepson, P.D., Tregenza, N., and Simmonds, M.P. 2008. Disappearing bottlenose dolphins (*Tursiops truncatus*) - is there a link to chemical pollution? International Whaling Commission, SC/60/E.

Jepson, P.D. and Deaville, R. (compilers); K. Acevedo-Whitehouse, J. Barnett, R.L. Brownell, A. Colloff, F. C. Clare, N. Davison, R. Law, J. Loveridge, S.K. Macgregor, S. Morris, R. Penrose, M. Perkins, E. Pinn, V. Simpson, M. Tasker, N. Tregenza, A.A. Cunningham and A. Fernández. 2009. Investigation of the common dolphin mass stranding event in Cornwall, 9th June 2008. Report to the *Department for Environment, Food and Rural Affairs* (under a variation to Contract CR0364). http://randd.defra.gov.uk/Document.aspx?Document=WC0601_8031_TRP.pdf.

Johnson-Restrepo B, Kannan K, Addink R, Adams DH. 2005. Polybrominated diphenyl ethers and polychlorinated biphenyls in a marine foodweb of coastal Florida. Environ Sci Technol;39:8243–8250.

Kakuschke, A., Valentine-Thon, E., Griesel, S., Fonfara, S., Siebert, U., and Prange, A. 2005. Immunological Impact of Metals in Harbour Seals (*Phoca vitulina*) of the North Sea. Environmental Science and Technology 39(19): 7568–7575.

Kannan, K., Blankenship, A.L., Jones, P.D., and Giesy, J.P. 2000. Toxicity reference values for the toxic effects of polychlorinated biphenyls to aquatic mammals. Human and Ecological Risk Assessment 6(1): 181–201.

Karlsson, O., Härkönen, T. and Bäcklin, B-M. 2007. Havet 2007. Available from the Swedish Environmental Protection Agency.

Kelly, B.C., Ikonomou, M.G., Blair, J.D., Surridge, B., Hoover, D., Grace, R., and Gobas, F.A.P.C. 2009. Perfluoroalkyl Contaminants in an Arctic Marine Foodweb: Trophic Magnification and Wildlife Exposure. Environmental Science and Technology 43(11): 4037–4043.

Kleivane, L., Espeland, O., Ugland, K.I., Skaare, J.U., Arnoldus Schytte Blix, L.W., and Øyvind, U. 1995. Seasonal variation of organochlorine concentrations in harp seal (*Phoca groenlandica*). *In* Developments in Marine Biology. Elsevier Science. pp. 599–605.

Krahn, M.M., Myers, M.S., Burrows, D.G., and Malins, D.C. 1984. Determination of metabolites of xenobiotics in the bile of fish from polluted waterways. Xenobiotica 14: 633–646.

Kuehl, D.W., Haebler, R., and Potter, C. 1991. Chemical residues in dolphins from the US Atlantic coast including Atlantic bottlenose obtained during the 1987/88 mass mortality. Chemosphere 22: 1071–1084.

Lahvis, G.P., Wells, R.S., Kuehl, D.W., Stewart, J.L., Rhinehart, H.L., and Via, C.S. 1995. Decreased lymphocyte responses in free-ranging bottlenose dolphins (*Tursiops truncatus*) are associated with increased concentrations of PCBs and DDT in peripheral blood. Environmental Health Perspectives 103: 67–72.

Law, R.J., Fileman, C.F., Hopkins, A.D., Baker, J.R., Harwood, J., Jackson, D.B., Kennedy, S., Martin, A.R., and Morris, R.J. 1991. Concentrations of trace metals in the livers of marine mammals (seals, porpoises and dolphins) from waters around the British Isles. Marine Pollution Bulletin 22(4): 183–191.

Law, R.J., and Whinnett, J.A. 1992. Polycyclic aromatic hydrocarbons in muscle tissue of harbour porpoises (*Phocoena phocoena*) from UK waters. Marine Pollution Bulletin 24: 550–553.

Law, R.J., Allchin, C.R., and Morris, R.J. 1995. Uptake of organochlorines (chlorobiphenyls, dieldrin; total PCB and DDT) in bottlenose dolphins (*Tursiops truncatus*) from Cardigan Bay, West Wales. Chemosphere 30(3): 547–560.

Law, R.J. 1996. Metals in marine mammals. In Environmental contaminants in wildlife: interpreting tissue concentrations. Edited by N.W. Beyer, G.H. Heinz and A.W. Redmond-Norwood. Lewis Publishers, London.

Law RJ, Blake SJ, Jones BR, Rogan E. 1998. Organotin compounds in liver tissue of harbour porpoises (*Phocoena phocoena*) and grey seals (*Halichoerus grypus*) from the coastal waters of England and Wales. Marine Pollution Bulletin 36:241–247.

Law RJ, Blake SJ, Spurrier CJH. 1999. Butyltin compounds in liver tissues of pelagic cetaceans stranded on the coasts of England and Wales. Marine Pollution Bulletin 38:1258–1261.

Law RJ, Bennett MF, Blake SJ, Allchin CR, Jones BR, Spurrier CJH. 2001. Metals and organochlorines in pelagic cetaceans stranded on coasts of England and Wales. Marine Pollution Bulletin 42:522–526.

Law RJ, Allchin CR, Bennett ME, Morris S, Rogan E. 2002. Polybrominated diphenyl ethers in two species of marine top predators from England and Wales. Chemosphere 46:673–681.

Law, RJ, Allchin, C.R., and Mead, L.K. 2005. Brominated diphenyl ethers in the blubber of twelve species of marine mammals stranded in the UK. Marine Pollution Bulletin 50(3): 356–359.

Law, RJ, Bersuder, P., Allchin, C.R., and Barry, J. 2006a. Levels of the Flame Retardants Hexabromocyclododecane and Tetrabromobisphenol A in the Blubber of Harbour Porpoises (*Phocoena phocoena*) Stranded or Bycaught in the UK, with Evidence for an Increase in HBCD Concentrations in Recent Years. Environmental Science and Technology 40(7): 2177–2183.

Law, RJ, Allchin, C.R., de Boer, J., Covaci, A., Herzke, D., Lepom, P., Morris, S., Tronczynski, J., and de Wit, C.A. 2006b. Levels and trends of brominated flame retardants in the European environment. Chemosphere 64(2): 187–208.

Law, R.J., Jepson, P.D., Deaville, R., Reid, R.J., Patterson, I.A.P., Allchin, C.R. and Jones, B.R. 2006c. Collaborative UK Marine Mammals Strandings Project: summary of contaminant data for the period 1993–2001. Sci. Ser. Tech Rep., Cefas Lowestoft, 131: 72pp.

Law RJ, Bersuder P, Barry J, Wilford BH, Allchin CR, Jepson PD. 2008a. A significant downturn in levels of hexabromocyclododecane in the blubber of harbour porpoises (*Phocoena phocoena*) stranded or bycaught in the UK: An update to 2006. Environmental Science and Technology 42:9104–9109.

Law RJ, Bersuder P, Mead LK, Jepson PD. 2008b. PFOS and PFOA in the livers of harbour porpoises (*Phocoena phocoena*) stranded or bycaught around the UK. Marine Pollution Bulletin 56:792–797.

Law, RJ, Bersuder, P., Barry, J., Deaville, R., Reid, R.J., and Jepson, P.D. 2010. Chlorobiphenyls in the blubber of harbour porpoises (*Phocoena phocoena*) from the UK: Levels and trends 1991–2005. Marine Pollution Bulletin 60, 470–473.

Law, RJ, Jon Barry, Philippe Bersuder, Jon Barber, Rob Deaville, Robert J. Reid RJ and Jepson PD. accepted. Levels and trends of BDEs in blubber of harbour porpoises (*Phocoena phocoena*) from the UK, 1992–2008. Environmental Science and Technology (accepted for publication).

Lebeu, M., Michelle, N., Trottier, S., and Measures, L. 2007. Temporal trends (1987–2002) of persistent, bioaccumulative and toxic (PBT) chemicals in beluga whales (*Delphinapterus leucas*) from the St Lawrence Estuary, Canada. Science of The Total Environment 383(1–3): 216–231.

Levin, M.J., De Guise, S., and Ross, P.S. 2005 Association between lymphocyte proliferation and organochlorine contaminants in recently weaned free-ranging harbour seal (*Phoca vitulina*) pups from British Columbia, Canada. Environ Toxicol Chem 24: 1247–1252.

Lie, E., Larsen, H.J.S., Larsen, S., Johansen, G.M., Derocher, A.E., Lunn, N.J., Norstrom, R.J., Wiig, O., and Skaare, J.U. 2005. Does high organochlorine (OC) exposure impair the resistance to infection in polar bears (*Ursus maritimus*)? Part II: Possible effect of OCs on mitogen- and antigen-induced lymphocyte proliferation. J Toxicol Environ Health 68A: 457–484.

Litz, J., Garrison, L., Feiber, L., Martinez, A., Contillo, J., and Kucklick, J. 2007. Fine-scale spatial variation of persistent organic pollutants in bottlenose dolphins (*Tursiops truncatus*) in Biscayne Bay, Florida. Environ Sci Technol 41:7222–8.

Lonergan, M., Duck, C.D., Thompson, D., Mackey, B.L., Cunningham, L., and Boyd, I.L. 2007. Using sparse survey data to investigate the declining abundance of British harbour seals. Journal of Zoology 271: 261–269.

Marsili, L., Gaggi, C., Bortolotto, A., Stanzani, L., Franchi, A., Renzoni, A., and Bacci, E. 1995. Recalcitrant organochlorine compounds in captive bottlenose dolphins (*Tursiops truncatus*): Biomagnification or bioaccumulation? *Chemosphere* 31, 3919–3929.

Martineau, D., Béland, P., Desjardins, C., and Lagacé, A. 1987. Levels of organochlorine chemicals in tissues of beluga whales (*Delphinapterus leucas*) from the St Lawrence Estuary, Quebec Canada. Archives of Environmental Contamination and Toxicology 16(2): 137–147.

Martineau, D., De Guise, S., Fournier, M., Shugart, L., Girard, C., Lagace, A., and Beland, P. 1994. Pathology and toxicology of Beluga whales from the St-Lawrence estuary, Quebec, Canada - Past, present and future. Science of the Total Environment 154(2–3): 201–215.

McHugh, B., Law, R.J., Allchin, C.R., Rogan, E., Murphy, S., Foley, M.B., Glynn, D., and McGovern, E. 2007. Bioaccumulation and enantiomeric profiling of organochlorine pesticides and persistent organic pollutants in the killer whale (*Orcinus orca*) from British and Irish waters. Marine Pollution Bulletin 54(11): 1724–1731.

Mortensen, P., Bergman, A., Bignert, A., Hansen, H.-J., Harkonen, T., and Olsson, M. 1992. Prevalence of skull lesions in harbour seals (*Phoca vitulina*) in Swedish and Danish museum collections: 1835–1988. Ambio 21(8): 520–524.

Muir, D.C.G., Wagemann, R., Hargrave, B.T., Thomas, D.J., Peakall, D.B., and Norstrom, R.J. 1992. Arctic marine ecosystem contamination. Science of the Total Environment 122: 75–134.

Muir, D., Braune, B., De March, B., Norstrom, R., Wagemann, R., Lockhart, L., Hargrave, B., Bright, D., Addison, R., Payne, J., and Reimer K. 1999. Spatial and temporal trends and effects of contaminants in the Canadian Arctic marine eco-system: a review. Sci Tot Environ, 230:83–144.

Muir, D., Innes, S., Cleement, M., Riget, F., Dietz, R., Kleivane, L., Skaare, J., Severinsen, T., Nakata, H., and Tanabe, S. 2000. Circumpolar trends of PCBs and organochlorine pesticides in the Arctic marine environment inferred from levels in ringed seals. Workshop on persistent organic pollutants (POPs) in the Arc-tic: Human health and environmental Concerns. Rovaniemi, Finland, Jan 2000, AMAP Report 2000:1, abstract 49.

Munson, L., Calzada, N., Kennedy, S., and Sorensen, T. 1998. Luteinized ovarian cysts in Mediterranean striped dolphins. J Wildl Dis 34(3): 656–660.

Murphy, S. 2009. Effects of contaminants on reproduction in small cetaceans. Final report of phase one to ASCOBANS. Sea Mammal Research Unit, University of St Andrews, St Andrews, 63pp.

Murphy, S., Pierce, G.J., Law, R.J., Bersuder, P., Jepson, P.D., Learmonth, J.A., Addink, M., Dabin, W., Santos, M.B., Deaville, R., Zegers, B.N., Mets, A., Rogan, E., Ridoux, V., Reid, R.J., Smeenk, C., Jauniaux, T., López, A., Farré, J.M.A., González, A.F., Guerra, A., García-Hartmann, M., Lockyer, C., and Boon, J.P. in press a. Assessing the effect of persistent organic pollutants on reproductive activity in common dolphins and harbour porpoises. NAFO/ICES/NAMMCO symposium "The Role of Marine Mammals in the Ecosystem in the 21st Century". Journal of Northwest Atlantic Fishery Science.

Murphy, S., Jepson, P., Deaville, R., and Monies, R.J. in press b. True hermaphroditism: First evidence of an ovotestis in a cetacean species.

Nigro, M., Campana, A., Lanzillotta, E., and Ferrara, R. 2002. Mercury exposure and elimination rates in captive bottlenose dolphins. Marine Pollution Bulletin 44: 1071–1075.

Oskam, I.C., Derocher, A.E., Lie, E., Skaare, J.U., Wiig, Ø., Larsen, S., Dahl, E., and Ropstad, E. 2001. Organochlorines effect the major androgenic hormone, testosterone, in male polar bears (*Ursus maritimus*). Pharmacol. Toxicol. 88, 23.

Olsson, M., Karlsson, B., and Ahnland, E. 1994. Diseases and environmental contaminants in seals from the Baltic and the Swedish west coast. The Science of the Total Environment 154(2–3): 217–227.

Olsson, M. and Reutergårdh, L. 1986. DDT and PCB pollution trends in the Swedish aquatic environment. *Ambio 15,* 103–109.

O'Shea, T.J., Moore, JF and Kochman, H.I. 1984. Contaminant concentrations in manatees in Florida. J. Wildlife Manag. 48, 741–748.

O'Shea, T.J., and Brownell, R.L. 1994. Organochlorine and metal contaminants in Baleen whales - A review and evaluation of conservation implications. Science of the Total Environment 154(2–3): 179–200.

O'Shea, T.J., Reeves, R.R., and Long, A.K. 1999. Marine Mammals and Persistent Ocean Contaminants. 1–150.

Pierce, G.J., Santos, M.B., Murphy, S., Learmonth, J.A., Zuur, A.F., Rogan, E., Bustamante, P., Caurant, F., Lahaye, V., Ridoux, V., Zegers, B.N., Mets, A., Addink, M., Smeenk, C., Jauniaux, T., Law, R.J., Dabin, W., López, A., Alonso Farré, J.M., González, A.F., Guerra, A., García-Hartmann, M., Reid, R.J., Moffat, C.F., Lockyer, C., and Boon, J.P. 2008. Bioaccumulation of persistent organic pollutants in female common dolphins (*Delphinus delphis*) and harbour porpoises (*Phocoena phocoena*) from western European seas: Geographical trends, causal factors and effects on reproduction and mortality. Environmental Pollution 153(2): 401–415.

Pillet, S., Lesage, V., Hammil, M., Cyr, D.G., Bouquegneau, J.-M., and Fournier, M. 2000. Effects of zinc and mercury on phagocytic activity of peripheral blood leucocytes of two species of seals from the St Lawrence estuary. Marine Pollution Bulletin 40: 921–927.

Pulster EL, Smalling KL, Zolman E, Schwacke L, Mruya KA. 2009. Persistent organochlorine pollutants and toxaphene congener profiles in bottlenose dolphins (*Tursiops truncatus*) frequenting the Turtle/Brunswick river estuary, Georgia, USA. Environ Toxicol Chem 28:1390–9.

Reep R.L., and Bonde, R.K. 2006. The Florida Manatee – Biology and Conservation, University Press of Florida ISBN 081302949X 224pp.

Reeves, R.R., Rolland, R., and Clapham, P.J. 2001. Causes of reproductive failure in North Atlantic right whales: new avenues of research, Northeast Fisheries Science centre reference document 01–16, Falmouth, Massachusetts.

Reeves, R.R., Smith, B.D., Crespo, E.A., and Notarbartolo di Sciara, G. 2003. Dolphins, whales and porpoises: 2002–2010 Conservation Action plan for the World's Cetaceans, IUCN/SSC Cetacean specialist group, IUCN, Gland, Switzerland and Cambridge, UK.

Reijnders, P.J.H. 1980. Organochlorine and heavy metal residues in harbour seals from the Wadden sea and their possible effects on reproduction. Netherlands Journal of Sea Research 14: 30–65.

Reijnders, P.J.H. 1984. Man-induced environmental factors in relation to fertility changes in pinnipeds. Envir. Conserv. 11: 61–65.

Reijnders, P.J.H. 1986. Reproductive failure in common seals feeding on fish from polluted coastal waters. Nature 324: 456–457.

Reijnders, P. 2003. Reproductive and developmental effects of environmental organochlorines on marine mammals. *In* Toxicology of marine mammals. *Edited by* J.G. Vos, G.D. Bossart, D.A. Fournier and O.S.T. J. Taylor and Francis, London. pp. 55–66.

Reijnders, P.J.H., Ries, E.H., Tougaard, S., Norgaard, N., Heidemann, G., Schwarz, J., Vareschi, E., and Traut, I.M. 1997. Population development of harbour seals *Phoca vitulina* in the Wadden Sea after the 1988 virus epizootic. J. Sea Res. 38, 161–168.

Reijnders, P.J.H., Aguilar, A., and Donovan, G.P. 1999. Chemical Pollutants and Cetaceans. *In* Journal of Cetacean Research and Management (Special Issue 1) International Whaling Commission, Cambridge.

Ross, P.S., DeSwart, R.L., Reijnders, P.J.H., Van Loveren, H., Vos, J.G., and Osterhaus, A.D.M.E. 1995. Contaminant-related suppression of delayed-type hypersensitivity and antibody responses in harbour seals fed herring from the Baltic Sea. Environmental Health Perspectives 103: 162–167.

Ross, P.S., Ellis, G.M., Ikonomou, M.G., Barrett-Lennard, L.G., and Addison, R.F. 2000. High PCB Concentrations in Free-Ranging Pacific Killer Whales, *Orcinus orca*: Effects of age, sex and dietary preference. Marine Pollution Bulletin 40(6): 504–515.

Ruus A, S.M., Ugland KI, and Skaare JU. 2002. Factors influencing activities of biotransformation enzymes, concentrations, and compositional patterns of organochlorine contaminants in members of a marine foodweb. Aquatic Toxicology 61: 73–87.

SCANS-II. 2008. Small cetaceans in the European Atlantic and North Sea (SCANS II). Final report to the European Commission under contract LIFE04NAT/GB/000245.

Schnitzler, J.G.G., Siebert, U., Jepson, P.D., Beineke, A., Jauniaux, T., Bouquegneau, J-M., and Das, K. 2008. Harbour porpoise thyroids: Histological investigations and potential interactions with environmental factors. Journal of Wildlife Diseases 44, 888–901.

Schwacke, L.H., Voit, E.O., Hansen, L.J., Wells, R.S., Mitchum, G.B., Hohn, A.A., and Fair, P.A. 2002. Probabilistic risk assessment of reproductive effects of polychlorinated biphenyls on bottlenose dolphins (*Tursiops truncatus*) from the Southeast United States coast. Environmental Toxicological Chemistry 21: 2752–2764.

Sellström, U., Jansson, B., Kierkegard, A., de Wit, C., Odsjö, T. and Olsson, M. 1993. Polybrominated diphenyl ethers (PBDE) in biological samples from the Swedish environment. *Chemosphere 26:* 1703–1718.

Shaw, S.D., Brenner, D., Berger, M.L., Fang, F., Hong, C.-S., Addink, R., and Hilker, D. 2008. Bioaccumulation of polybrominated diphenyl ethers in harbour seals from the northwest Atlantic. Chemosphere 73(11): 1773–1780.

Sipes, I.G., and Gandolfi, A.J. 1991. Biotransformation of toxicants. *In* Casarett and Doull's Toxicology. *Edited by* M.O. Amdur, J. Doull and C.D. Klaassen. Pergamon Press, New York. pp. 88–126.

Skaare, J.U., Larsen, H.J., Lie, E., Bernhoft, A., Derocher, A.E., Norstrom, R., Ropstad, E., Lunn, N.F., and Wiig, Ø. 2002. Ecological risk assessment of persistent organic pollutants in the arctic. Toxicology 181–182: 193–197.

Smith, D.R., Niemeyer, S., Estes, J.A., and Flegal, A.R. 1990. Stable lead isotopes evidence of anthropogenic contaminant in Alaskan sea otters. Environmental Science and Technology 24: 1517–1521.

Tanabe, S., Iwata, H., and Tatsukawa, R. 1994. Global contaminants by persistent organochlorines and their ecological impacts on marine mammals. The science of the Total Environment 154: 163–177.

Tanabe, S., Watanabe, S., Kan, H., and Tatsukawa, R. 1988. Capacity and mode of PCB metabolism in small cetaceans. Mar Mammal Sci 4:103–24.

Toft, G., Hagmar, L., Giwercman, A., and Bonde, J.P. 2004. Epidemiological evidence on reproductive effects of persistent organochlorines in humans. Reproductive Toxicology 19(1): 5–26.

van Bavel, B., Dam, M., Tysklind, M., Lindstrom, G. 2001. Levels of polybrominated diphenyl ethers in marine mammals. Organoh Comp, 52:99–103.

Van de Vijver, K.I., Hoff, P.T., Das, K., Van Dongen, W., Esmans, E.L., Jauniaux, T., Bouquegneau, J.M., Blust, R., and de Coen, W. 2003. Perfluorinated chemicals infiltrate ocean waters: link between exposure levels and stable isotope ratios in marine mammals. Environ Sci Technol. 37: 5545–5550.

Van de Vijver, K.I., Hoff, P., Das, K., Brasseur, S., Van Dongen, W., Esmans, E., Reijnders, P., Blust, R., and De Coen, W. 2005. Tissue distribution of perfluorinated chemicals in harbour seals (*Phoca vitulina*) from the Dutch Wadden Sea. Environmental Science and Technology 39: 6978–6984.

Vos, J. G., De Heer, C., and Van Loveren, H. 1997/98. Immunotoxicity effects of TCDD and Toxic Equivalency Factors. Teratogenesis, Carcinogenesis, and Mutagenesis 17, 275–284.

Watson, W.S., Sumner, D.J., Baker, J.R., Kennedy, S., Reid, R., and Robinson, I. 1999. Radionuclides in seals and porpoises in the coastal waters around the UK. Science of the Total Environment 234(1–3): 1–13.

Weijs, L., Dirtu, A.C., Das, K., Gheorghe, A., Reijnders, P.J.H., Neels, H., Blust, R., and Covaci, A. 2009. Inter-species differences for polychlorinated biphenyls and polybrominated diphenyl ethers in marine top predators from the Southern North Sea: Part 1. Accumulation patterns in harbour seals and harbour porpoises. Environmental Pollution 157(2): 437–444.

Wells, R.S., Tornero, V., Borrell, A., Aguilar, A., Rowles, T.K., Rhinehart, H.L., Hofmann, S., Jarman, W.M., Hohn, A.A., and Sweeney, J.C. 2005. Integrating life-history and reproductive success data to examine potential relationships with organochlorine compounds for bottlenose dolphins (*Tursiops truncatus*) in Sarasota Bay, Florida. Science of The Total Environment 349(1–3): 106–119.

Wolkers, J., Burkow, I.C., Lydersen, C., Dahle, S., Monshouwer, M., Witkamp R.F. 1998. Congener specific PCB and polychlorinated camphene (toxaphene) levels in Svalbard ringed seals (*Phoca hispida*) in relation to sex, age, condition and cytochrome P450 enzyme activity. Sci Tot Environ, 216:1–11.

Wolkers, H., Burkow, I.C., Lydersen, C., Witkamp, R.F. 2000. Chlorinated pesticide concentrations, with an emphasis on the polychlorinated camphenes (toxaphenes), in relation to cytochrome P450 enzyme activities in harp seals (*Phoca groenlandica*) from the Barents Sea. Environ Toxicol Chem, 19:1632–1637.

Wolkers, H., van Bavel, B., Derocher, A., Wiig, Ø., Kovacs, K., Lydersen, C., and Lindsrom, G. 2004a. Congener-Specific accumulation and food chain transfer of polybrominated diphenyl ethers in two Arctic food chains. Environ Sci Technol, 38: 1667–1674.

Wolkers, H., Lydersen, C., and Kovacs, K.M. 2004b. Accumulation and lactational transfer of PCBs and pesticides in harbour seals (*Phoca vitulina*) from Svalbard, Norway. Science of The Total Environment 319(1–3): 137–146.

Wolkers, H., Corkeron, P.J., Van Parijs, S.M., Similä, T., and Van Bavel, B. 2007. Accumulation and transfer of contaminants in killer whales (*Orcinus orca*) from Norway: Indications for contaminant metabolism. Environmental Toxicology and Chemistry 26(8): 1582–1590.

Zakharov, V.M., and Yablokov, A.V. 1990. Skull asymmetry in the Baltic grey seal: effects of environmental pollution. Ambio 19: 266–269.

Zegers, B.N., Mets, A., van Bommel, R., Minkenberg, C., Hamers, T., Kamstra, J.H., Pierce, G.J., and Boon, J.P. 2005. Levels of Hexabromocyclododecane in harbour porpoises and common dolphins from western European Seas, with evidence of stereoisomer-specific biotransformation by cytochrome P450. Environ. Sci. Technol. 39(7): 2095–2100.

6 ToR c. Further development of the framework for surveillance and monitoring of marine mammals applicable to the ICES area

6.1 Introduction

In the 2009 Report we briefly reviewed the relevant legislation, issues with collecting the required data and the range of data sources available. In this complementary document, we describe the legislative framework in more detail and review current surveillance and monitoring. We then attempt to identify best practice and conclude with a series of general recommendations.

Long-term monitoring is fundamental to the management of ecosystems and their components. Surveillance and monitoring of marine mammal populations typically involves:

1) Surveys for abundance, distribution and movements on local, regional and international scales. These include visual surveys, telemetry, passive acoustic detectors and photo-identification. All these occur at various scales, and can be associated with EIAs for development activities;

2) Strandings monitoring and associated collection of life history, diet, disease and contaminant data;

3) Fishery monitoring (normally for bycatch, but also for damage and depredation caused by marine mammals, and, in some areas, for directed catches). This may involve on-board observation (by observers or video surveillance), compulsory or voluntary reporting, carcase recovery schemes, or interview surveys;

4) Monitoring for mitigation, e.g. during seismic surveys, the use of explosives, pile driving, wind farm construction, and naval activities;

5) Health assessment of live-caught animals. These may either be caught specifically for this purpose or as part of other studies (e.g. telemetry studies);

6) Scat collection. This is primarily used for dietary analysis (especially in pinnipeds), but can also be used to monitor animal health.

Ideally, conservation and management of a population requires information on its current status and trends in that status. Successful monitoring programmes, therefore, require adequate planning (i.e. good design, including power analysis), clearly defined objectives and links to management and policy (Figure 6). Lindenmayer and Likens (2009) proposed "adaptive monitoring", in which the entire monitoring and surveillance framework (from defining questions and experimental design through to data collection, analysis and interpretation) is an iterative process. In this way the framework can also evolve to meet new needs and constraints as they arise. However, consistently collected (i.e. internally comparable) long-term datasets are extremely valuable and this should be borne in mind when taking any decisions about the way data are collected and/or the type of data collected.

In general, national Governments implement monitoring programmes based on legislative drivers (e.g. the EU Habitats Directive in Europe or the Marine Mammal Protection Act in the US). The legislation may define the units of study (e.g. specific sites, species, or regional components of populations) as well as the types of information required. Governments may also respond to bottom–up pressure (e.g. public opinion, lobbying by NGOs, scientific advice) to undertake localized or short-term monitoring for a specific issue. Additionally, monitoring (usually short-term) is undertaken by a range of institutions, including, universities, research institutes and NGOs. For example, the NGO CEMMA in Galicia, NW Spain, has monitored strandings and undertaken sightings surveys since 1990.

Interpretation of results from monitoring data, to guide management action is a critical part of the process (Figure 6). There is clear scope for accidental or deliberate misuse of results. Consequently, scientists responsible for collection and analysis of monitoring data should be mindful of the need to also take some responsibility for its use. Inappropriate inferences may be drawn due to simplistic and unscientific assumptions, leading to poorly justified recommendations for management action. Equally, decisions to not take any action to mitigate a threat may be justified by appealing to the need for more information.

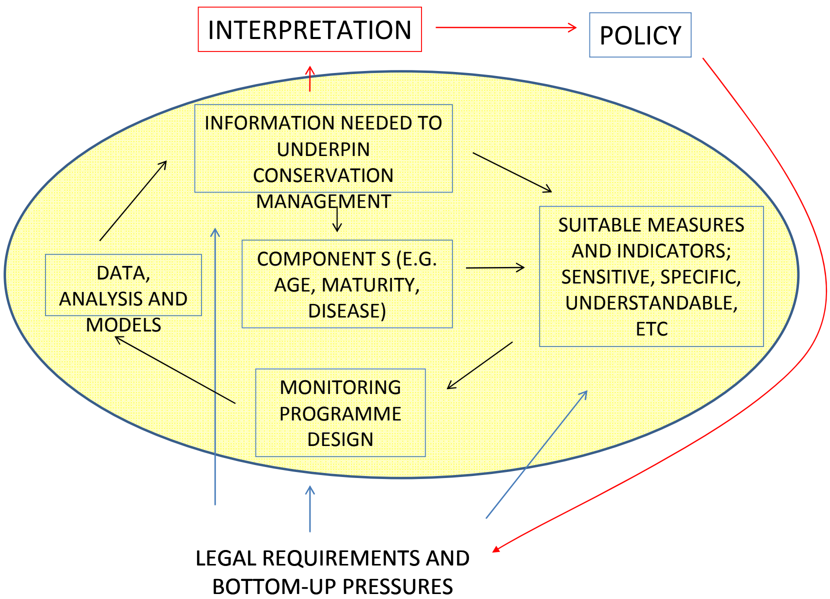


Figure 6. Schematic diagram of surveillance and monitoring to meet conservation management needs.

Legislative requirements for monitoring, the information necessary to define status and trends, and the information that it is feasible (logistically, financially, etc) to collect, are not entirely the same (Figure 7). However, while the value of some current monitoring could be questioned, and some current criteria for evaluating conservation status are undoubtedly difficult to apply, probably the most beneficial changes to current surveillance and monitoring have to do with improving coordination and standardization (including setting up of common databases and sample banks), working at more appropriate spatial scales, improving geographic coverage (e.g. through integration of non-governmental monitoring programmes) and maximizing use of available data and samples, and ensuring adaptability.

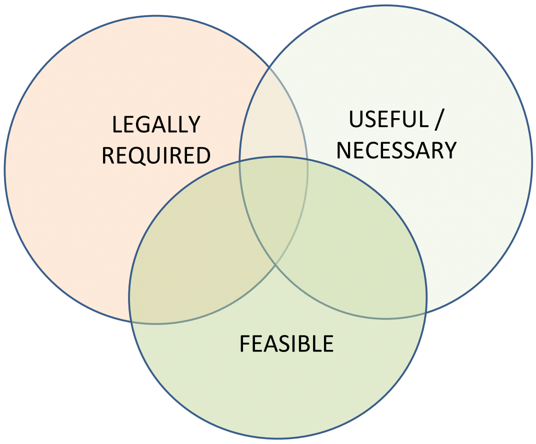


Figure 7. *M*arine mammal monitoring – schematic representation of overlap between legal requirements, information needed for management, and feasibility.

6.2 Legislative drivers and obligations for monitoring in the ICES area

Full details of relevant legislation are given in Annex 1. The most relevant European legislation is as follows:

- Bern Convention and the Habitats Directive (Table 6);
- CMS and ASCOBANS;
- Marine Strategy Framework Directive and Good Environmental Status;
- The Convention for the Protection of the Marine Environment of the North-East Atlantic (OSPAR) (Table 7);
- Council Regulation 812/2004;
- North Atlantic Marine Mammal Commission (NAMMCO).

The main relevant North American legislation is

- Marine Mammal Protection Act (USA);
- The Species at Risk Act (SARA) (Canada)

6.3 Current monitoring

Table 8 provides a summary of the current monitoring requirements within the ICES area. Annex 2 and 3 provide overviews of current monitoring of cetaceans and seals respectively. It is clear that NGOs are, in several regions, filling gaps in monitoring schemes. For example, very few Member States are making sufficient efforts to fulfil their obligations under EC Regulation 812/2004 (EC, 2009). In several countries (notably Spain and Portugal), voluntary organizations and NGOs provide the only monitoring of marine mammal strandings, as well as carrying out sightings surveys and monitoring interactions with fisheries (indeed with a substantially greater, albeit more localized, coverage than official fishery monitoring programmes).

Table 6. Habitats Directive criteria to assess the conservation status of listed species.

Parameter	Conservation status			
	Favourable	Unfavourable - inadequate	Unfavourable - bad	Unknown
Range	Stable (loss and expansion in balance) or increasing AND Not smaller than the 'favourable reference range'	Any other combination	Large decline: Equivalent to a loss of more than 1% per year since 1994 OR More than 10% below favourable reference range	No or insufficient reliable information available
Population	Population above 'favourable reference population' AND Reproduction, mortality and age structure not deviating from normal (if data available)	Any other combination	Large decline: Equivalent to a loss of more than 1% per year (indicative value MS may deviate from if duly justified) within period specified by MS AND below 'favourable reference population' OR More than 25% below favourable reference population OR Reproduction, mortality and age structure strongly deviating from normal (if data available)	No or insufficient reliable information available
Habitat for the species	Area of habitat is sufficiently large (and stable or increasing) AND Habitat quality is suitable for the long-term survival of the species	Any other combination	Area of habitat is clearly not sufficiently large to ensure the long-term survival of the species OR Habitat quality is bad, clearly not allowing long-term survival of the species	No or insufficient reliable information available
Future Prospects	Main pressures and threats to the species not significant; species will remain viable on the long-term	Any other combination	Severe influence of pressures and threats to the species; very bad prospects for its future, long-term viability at risk.	No or insufficient reliable information available

Parameter	Conservation status			
Overall assessment of CS	All Favourable 'green' OR three Favourable 'green' and one Unknown	One or more Unfavourable - Inadequate 'amber' but no Unfavourable – Bad 'red'	One or more Unfavourable - Bad 'red'	Two or more 'Unknown' combined with Favourable or all "Unknown"

Table 7. OSPAR criteria to assess the status of harbour porpoise populations.

	Favourable	Unfavourable-inadequate	Unfavourable – bad
Occurrence/ Distribution	In >90% of known historical area (or similar baseline)	In 70–90% of known historical area (or similar baseline)	In <70% of known historical area (or similar baseline)
Population estimate and trend (national average)	Stable or increasing with respect to historical or other baseline reference value	Decreasing with respect to historical or other baseline reference value	Large decline with respect to historical or other baseline reference value
Population density (national average)	high (>1.0 animal per km^2)*	medium (0.3–1.0 animal per km^2)*	low or decreasing (<0.3 animal per km^2)*
Population structure (national average)	Reproduction, mortality and age structure not deviating from normal (if data available)	Reproduction, mortality and age structure deviating from normal (if data available)	Reproduction, mortality and age structure strongly deviating from normal (if data available)
Habitat quality (national average)	Sufficiently large area of good quality habitat suitable for the long-term survival of the species	Habitat quality deteriorating and/or being reduced in area	Habitat quality is poor and/or insufficiently large enough, and clearly not allowing the long-term survival of the species
Health Status - toxin loading (POPs and metals)	< 17 mg/kg lipid total PCBs**	> 17 mg/kg lipid total PCBs**	Not yet determined
Anthropogenic mortality (including bycatch)	< 1.0% of estimated population size	1.0–1.7% of estimated population size	> 1.7% of estimated population size
Fisheries monitoring and reporting of bycatch (to support mortality values as above)	appropriate monitoring and reporting of harbour porpoise bycatch for all affected fisheries	monitoring and reporting conforming to the minimum requirements of EU Reg 812/2004***(or equivalent for Non-EU Member States)	incomplete monitoring and reporting of harbour porpoise bycatch
Anthropogenic disturbances / displacement (relative ranking****)	little or no: ship traffic, motorised tourism, military sonar, seismic testing, other noise, or extraction activities	some: ship traffic, motorised tourism, military sonar, seismic testing, other noise, or extraction activities	extensive: ship traffic, motorised tourism, military sonar, seismic testing, other noise, or extraction activities

*** Tentative working values in OSPAR Region II, subject to change when more data are available; a thorough understanding of long-term and seasonal variability is prerequisite.**

**** Tentative working values, subject to change when more data are available.**

***** Or superseding legislation.**

****** Relative ranking approach needs to be elaborated.**

Table 8. Comparison of legal requirements and obligations for monitoring in North Atlantic. CP = contracting parties.

	Habitats Directive (FCS) (all marine mammals)	ASCOBANS (small cetaceans only)	OSPAR (harbour porpoise monitoring and seal EcoQOs)	EU Regulation 812/2004 and OSPAR porpoise bycatch EcoQO	NAMMCO	MMPA (all marine mammals)	SARA (only species designated as being at risk)
Population structure	Report by species within national waters rather than by population	Requires CP to assess population structure	Case assessment undertaken by OSPAR area but CP work in national waters	Required for accurate assessments of bycatch	Based on IWC stock assessments	Primary requirement of status assessment	Required for status assessment
Abundance	Report trends by national waters rather than biological population	Assessment of abundance, including trends	abundance of porpoise and harbour seals[4], including trends. Pup production for grey seals	Requires a robust estimate of population size to contextualise bycatch estimates. Currently the SCANS II (2005) estimates are used.	Assessment of abundance and trends based on NASS surveys in 87, 1989 and 1995.	Assessment of minimum population size and any trends by stock	Provides abundance estimates and trends
Range and/or distribution	Requirement to assess trends in range	Requires CP to assess distribution, including seasonal movements			Description of distribution within each stock area	Assessment of range by stock	Outlines distribution
Habitat	Required to assess available habitat, usage of that habitat and trends by national waters.	Requires identification of areas of importance	Enhanced monitoring required in area of particular importance			Broad assessment of habitat preferences	Availability of habitat and trends, which tend to be broad assessments of habitat preferences

4 as measured by numbers hauled-out

	Habitats Directive (FCS) (all marine mammals)	ASCOBANS (small cetaceans only)	OSPAR (harbour porpoise monitoring and seal EcoQOs)	EU Regulation 812/2004 and OSPAR porpoise bycatch EcoQO	NAMMCO	MMPA (all marine mammals)	SARA (only species designated as being at risk)
Threats	Information obtained from bycatch and strandings monitoring	Data collected on bycatch, vessel strikes and strandings.	Data collected from bycatch and strandings	Requires bycatch observers on some fleet segments and areas	Mainly covers harvesting and pollutants.	Information obtained from bycatch observers and strandings monitoring	Information obtained from bycatch observers and strandings monitoring
Other measures	Future prospects					Net reproductive estimate, PBR estimate, factors impeding stock recovery	
Time frame	Every 6 years	Annual national reports updating available information	Every 6 years minimum, preferably every 3. Enhanced monitoring every year.	Annual reporting of bycatch estimates	Assessment undertaken following surveys. T-NASS undertaken in 2007, but results not yet available.	Annual for strategic stocks, every 3 years for non-strategic stocks	Every 10 years, unless otherwise warranted

6.4 Surveillance and monitoring needs and best practice

6.4.1 Criteria for favourable status and adaptive monitoring

In designing a framework for surveillance and monitoring, ICES should take into account similar initiatives undertaken by other organizations, e.g. ASCOBANS, ACCOBAMS and OSPAR and the transboundary nature of most marine mammal populations. It is, therefore, suggested that ICES convene a joint forum to agree details of a monitoring framework that could be used to fulfil a variety of requirements. Nevertheless, here we describe some of the essential features of such a monitoring programme.

Clearly, monitoring targets should mesh with criteria used to evaluate the status of a species (e.g. criteria for Favourable Conservation Status under the Habitats Directive). Equally obviously, criteria which cannot be met by any feasible monitoring programme, or which do not provide a basis for meaningful management action, should be amended.

Lindenmayer and Likens (2009) define "adaptive monitoring", a process which evolves iteratively as questions and constraints evolve. These authors highlight three main types of flaws with many monitoring programmes:

a) Imprecise definition of objectives. For example monitoring programmes may be initiated in response to short-term funding availability or a current "hot topic" and detailed identification of rationale may come after the fact;

b) Poor design. For example, statistical power is often not evaluated, and the need for contrasts between treatments (e.g. areas with and without human intervention) is ignored. The authors also highlight the value of rotating sampling to increase the number of sites monitored;

c) Disagreements and uncertainty about what to measure. This can lead to a wasteful "laundry list" approach or to the use of poorly justified "indicator" species (or variables).

Following from this, they identify the key features of a monitoring programme (see Figure 8) as being:

1) It addresses well-defined questions specified in advance of commencement of monitoring;

2) It is underpinned by rigorous statistical design, including power analysis;

3) It is based on a conceptual model of ecosystem, or the monitored components function;

4) It is driven by a "need to know", i.e. it is specifically relevant to management.

A key point is the transboundary nature marine of mammal populations, so that monitoring of each population needs to be coordinated internationally, and evaluation of status should be carried out by a supranational forum. The intermediate stage of carrying out national evaluations will be both ineffective and an unnecessary duplication of effort. Monitoring and assessments must be focused at the natural biological population and not on artificial political boundaries, such as the national level, proposed boundaries for MSFD or the ICES areas if inappropriate to the species.

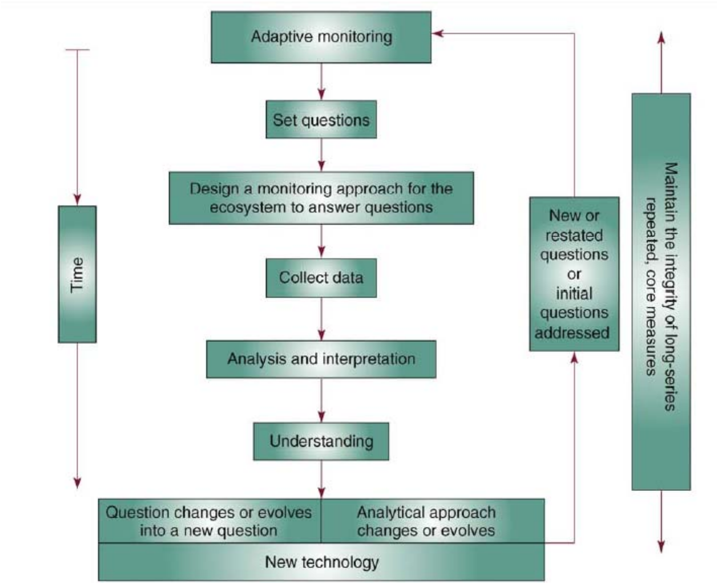


Figure 8. Adaptive monitoring (from Lindenmayer and Likens, 2009).

6.4.2 Power analysis

Power analysis is required for all monitoring programmes to ensure that they are robustly designed with sufficient ability to detect particular trends. It is one of the most difficult aspects to get right, and one of the most commonly neglected. However, Seavy and Reynolds (2007) caution against giving undue emphasis to the statistical power to detect annual trends, on the grounds that such calculations require many assumptions (see Section 4.5).

6.4.2.1 Trends in abundance

WGMME 2008 (Section 8.4.5) discussed the issue of power to detect trends in abundance, survey design and over- vs. underprotection decisions that derive from the choice of β and α levels:

'*The statistical power of a monitoring program is the probability that the monitoring will detect a trend in the data despite the 'noise' associated with seasonal cycles and other fluctuations (Nichols and Williams, 2006)….if the risk of over and underprotection are to be similar, a trade-off is required between power (i.e. β) and level of significance (i.e. α) with consideration given to using a value of 0.2.*'

WGMME 2009 (see Section 9.3.2) considered these issues further and looked at the power of the large-scale SCANS and CODA surveys to detect trends in abundance over time (Figure 9). '*Results indicated a high power to detect trends only for harbour porpoise (based on SCANS II data) and bottlenose dolphins in offshore waters (based on CODA data). With an effort of 10 000 km every year for ten annual surveys, there is a power of 0.92 to detect a 5% decline of harbour porpoises per year (i.e. a 37% decline over 9 years) during that period. However, the power to detect a 37% decline between two abundance estimates (i.e. with the current periodicity of large-scale surveys undertaken every 10 years) with the same CV is only 0.29.*'

However, WGMME 2008 noted that the 'precision of known estimates has implications for future monitoring requirements. If the required power of a monitoring programme is 80%, where precision is known to be high (e.g. CV of approximately 0.15), surveys could be undertaken less frequently if the decision criteria are altered (e.g. α = 0.1 or 0.2). In contrast where the precision is low (e.g. CV of approximately 0.3), both frequent surveys and lower decision criteria thresholds will be required. Moving from $\alpha = 0.05$ to 0.1 or 0.2 means that we will be prepared to make an over protection error 1 in 20, 10 or 5 times respectively (i.e. conclude that a particular trend is occurring when in fact it is not).'

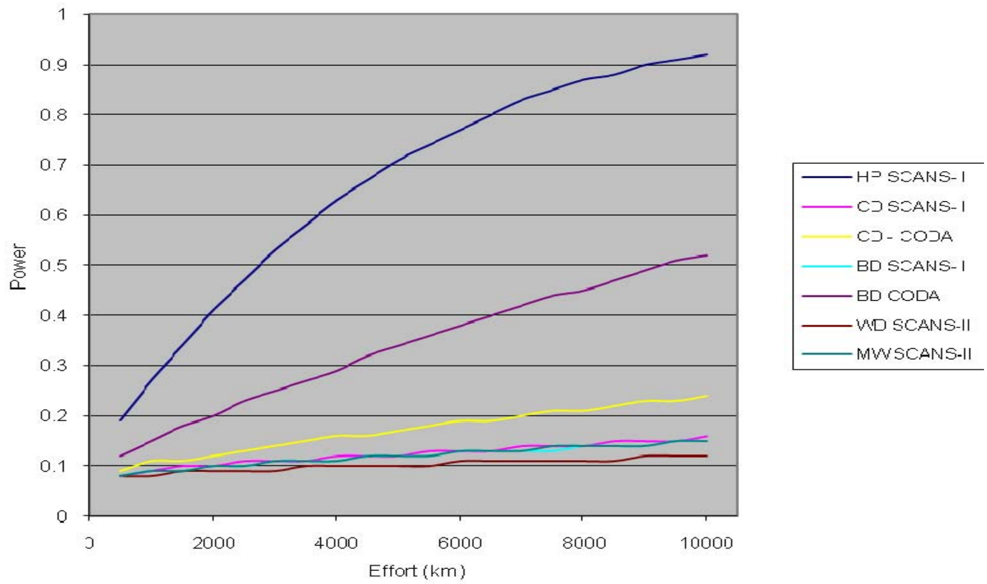


Figure 9. Power analyses from SCANS II and CODA using β=0.2 and α=0.05.

The CVs for the abundance estimates obtained during SCANS II and CODA were 0.15 for harbour porpoise and 0.26 for bottlenose dolphin, respectively. For the remaining species CV ranged from 0.49 to 1.0. Consequently, changing the decision criteria (i.e. making β and α levels equivalent to 0.2) will increase the power to detect trends in harbour porpoise and bottlenose dolphin, but will not be sufficient to improve the output from these surveys for other species (see Figure 10). It should also be noted that these analyses were for a SCANS/CODA type survey to be repeated annually, something that is not feasible either logistically or financially.

An alternative approach is required that takes a longer-term view. Because of the issues of power surrounding long-term monitoring studies, Seavy and Reynolds (2007) proposed that the evaluation of monitoring programmes should include prospective power analysis, as well as standards for sampling design and precision. These complementary approaches should design monitoring programmes that are both effective at detecting trends and generating precise estimates of population parameters.

An example of such an approach is provided by Davey and Aebischer (2009). They examined trends in the data using percentage change (with confidence intervals) for the most recent 5, 10, 15, 25 and 30 year intervals, where there was sufficient data, despite the data being collected annually. It has been suggested that such an approach could be adopted for reporting FCS under the Habitats Directive using six yearly intervals (J. Batterby, pers. comm.). If adopted, this would lead to trend reporting under the Habitats Directive occurring only when sufficient robust data exists. For marine mammals, particularly cetaceans, this could mean trends are not reported with any degree of confidence for another 10 to 15 years, maybe longer depending on the type of surveillance data collected and species under consideration.

Seavy and Reynolds (2007) also indicate that conservation biologists and/or managers need to come up with creative or non-traditional mechanisms of assessing population trends. One such mechanism that could potentially be applied to cetaceans and seals is the use of life-history data.

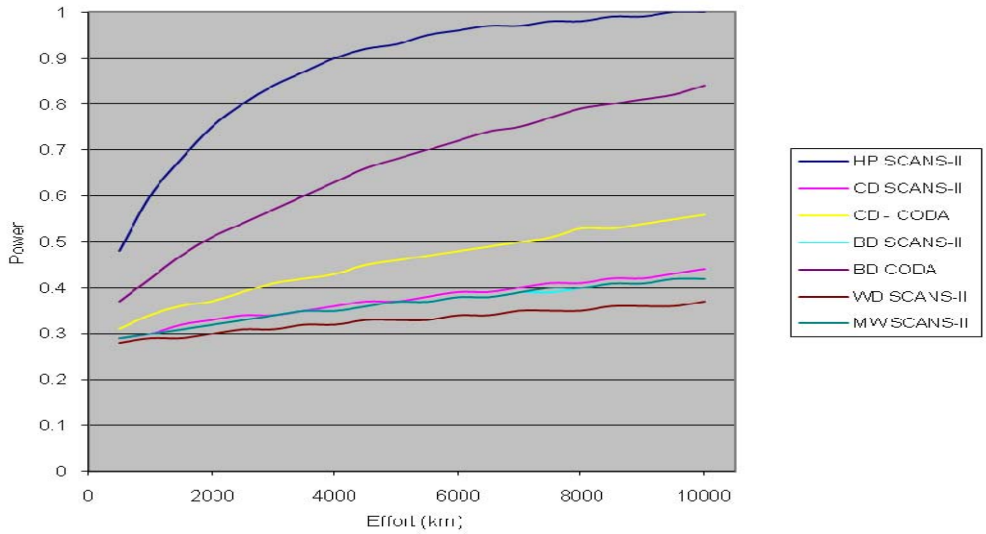


Figure 10. Power analyses from SCANS II and CODA using β=α=0.2

6.4.2.2 Pregnancy rate in common dolphins

Estimates of reproductive parameters in marine mammals can be used to assess changes in dynamics of populations as a result, for example, of incidental bycatch. Further, they allow assessment of the long-term effects from anthropogenic toxins, such as PCBs and DDT, and infectious disease outbreaks on reproductive out at the individual and population level. Although evidence is extremely limited at this time, it is thought that anthropogenic noise may also affect reproductive rates (Wright *et al*., 2007).

To date within European waters, population/stock reproductive parameters have been determined for common dolphins and harbour porpoises using post-mortem data (Learmonth, 2006; Murphy, 2008; Murphy *et al*., 2009). These data have been used to determine the effects of incidental capture in pelagic trawl fisheries on the common dolphin population in the North-east Atlantic as part of the EC NECESSITY project; the effects of anthropogenic toxins on reproductive output in both species as part of the EC BIOCET and ASCOBANS funded projects; and incorporated into the production of bycatch mortality limits for harbour porpoises and common dolphins as part of the EC-LIFE SCANS II and CODA projects, respectively. A full assessment of available data for other species has not been undertaken.

As part of various European stranding programmes, including the UK Cetacean Stranding Investigation Programme (CSIP), cause of death and nutritional condition of individuals are investigated. Teeth, ovaries and testes are collected for subsequent analysis assessing reproductive parameters such as maturity status and age. These data allow an assessment of temporal variations in: population pregnancy rates, proportion of mature individuals, proportion of females simultaneously pregnant and lactating, average age attained at sexual maturity, nutritional condition, length and timing of the oestrus period, and variations in reproductive parameters with age. Temporal variations in the above parameters can occur due to alterations in the availability of prey resources and population density. Cetacean populations are regulated through density-dependent changes in reproduction and survival, and it has been proposed that food resources are the main causative agent in the expression of density-dependence, resulting in an increase in population growth rates (and reproductive output) at low densities (e.g. following large-scale incidental mortality in fishing gear) and a decrease in growth rates (and reproductive output) at high densities (Murphy *et al*., 2009).

Knowledge of extrinsic factors such as bycatch rates and contaminant loads are required to give context to cross sectional life-history information. Anthropogenic toxins and disease can alter reproductive rates by decreasing fertility, and causing abortions, premature parturition and neonatal mortality.

Although largely ignored, an important part of the monitoring requirements under the Habitats Directive is the monitoring of changes/trends in life-history parameters as this can also be used as a measure of conservation status. It is also a requirement of ASCOBANS monitoring. The regular monitoring of reproductive rates in common dolphins and harbour porpoises would therefore contribute significantly to assessments of conservation status.

Pregnancy rate is relatively easily measured, but power analyses suggest that only extremely large variations in the common dolphin pregnancy rate of the North-east Atlantic population would be detectable under current data sampling regimes (Murphy *et al*., 2009). For common dolphins, with an initial pregnancy rate of 25% (data collected between 1990 and 2006, n = 248 mature females), more than an additional 150 mature females would have to be necropsied to have a ≥80% chance (power) of detecting a reduction to a 12%, or an increase to a 40%, pregnancy rate at a significance level of 5%. A reduction to a pregnancy rate of 10% should be detectable from 100 samples. Whereas, 50 mature females would have the same power in detecting a reduction to a pregnancy rate of 5% (Murphy *et al*., 2009). Sample sizes of 50 to 100 mature females are approximately equivalent to the total number of suitable specimens obtained in western European waters over a 5 to 10 year period.

Changes in pregnancy rates are therefore likely to become biologically significant long before they can be detected statistically. However, where the power of a monitoring scheme (e.g. >80%, $\beta = 0.2$) to detect change is different from the level of significance (e.g. 5% or α=0.05) there is an imbalance in the risks of under and over protection. Using a lower significance level of 0.2, a power of ≥80%, and an initial pregnancy rate of 25%, a sample size of only 50 mature females would be required to detect an absolute decline of >16% in the pregnancy rate, and an absolute increase of >20% in the pregnancy rate. It has been reported in other studies that adequate age and reproductive data from males and females (at least 50 individuals of each sex) are also vital for estimating the average age attained at sexual maturity (e.g. Hohn, 1989; Chivers and Myrick, 1993).

Obtaining a large sample size of sexually immature and mature individuals will be difficult (as a large number of samples in the Murphy *et al*. (2009) study were obtained from tuna driftnet fishery observer bycatch programmes in the 1990s), and requires that European stranding and observer bycatch programmes continue sampling all available and suitable carcasses. One compromise would be to alter the criteria used for significance, as has been suggested for abundance, or to report likelihoods of declines having occurred rather than whether a particular threshold has been achieved.

6.4.3 Integration of monitoring from other sectors – common protocols and databases

In relation to sightings surveys, mechanisms are currently being developed at an international level that will enable as much of the cetacean surveillance undertaken in European waters by various agencies, research bodies and the voluntary sector to be included and used in the conservation status assessments. The amalgamation of the 1994 SCANS survey data with 1973–1997 Seawatch Foundation effort related sightings data and the 1979–1999 European Seabirds at Sea (ESAS) data formed the Joint Cetacean Database (JCD) which enabled the production of an atlas of cetacean distribution (Reid *et al.*, 2003). This represented the most up-to-date statement on the distribution and relative abundance of all 28 species recorded on the European continental shelf in the latter part of the 20th Century, but it is now out of date. Development of the Joint Cetacean Protocol, a web-based portal for effort-related sightings data, will enable our knowledge of distribution and relative abundance to remain current; thereby ensuring that up-to-date information is available.

Similarly, for strandings monitoring, it would be possible to build on protocols published by the European Cetacean Society and (US) Society for Marine Mammalogy to create an international database. Since its inception, ASCOBANS has held a long standing ambition to develop an international strandings database. A previous initiative from the voluntary sector to create a common database (ATLANCETUS) in the 1990s is no longer active. The current Belgian initiative to create a "European Marine Mammal Tissue Bank" could perhaps accommodate a strandings database (see Section 9 (ToR f)), because it already *de facto* contains a partial database, namely for all animals that have contributed samples.

In both these cases, a European (e.g. through ASCOBANS) and/or ICES level network is needed, possibly with European funding (e.g. through COST or similar initiatives). Alternative to this may be hosting it under the auspices of the European Cetacean Society (which, despite its name, also encompasses other marine mammals).

6.4.4 General aims of monitoring

Marine mammal monitoring is primarily focused on assessment of population status, in particular whether the current population status or trends therein, or specific threats, give cause for concern. It is difficult or impossible to objectively define the "preferred" or baseline abundance of a marine mammal population. Usually, carrying capacity is unknown and variable and, in the case of large whales, abundances have been greatly reduced by whaling.

Nevertheless, there is a need to reach a consensus on appropriate reference points (e.g. precautionary reference points, below which there is a significant risk of local extinction). Specifically:

a) Is current population status "favourable"? (e.g. compared with some notional baseline status or reference level; if necessary, in relation to a precautionary level);

b) Is there any trend in population status? (e.g. is the population declining);

c) Is there evidence of adverse effects of specific threats (e.g. fishery bycatch)?

6.4.5 Critique of different monitoring categories

Abundance: it is generally recognized that even expensive and labour-intensive large-scale surveys have a limited ability to detect trends in marine mammal population size (e.g. Taylor *et al.*, 2007). However, Seavy and Reynolds (2007) cautioned against giving undue emphasis to the statistical power to detect annual trends, on the grounds that such calculations require many assumptions. They argue that monitoring standards should emphasize attributes of sampling design that increase precision (e.g. randomization, bias, and detection probability). Clearly, regardless of trends, estimates of abundance are a potentially important indicator of status (e.g. to determine favourable conservation status, provided that a baseline value can be identified) and as a basis for evaluating threats (e.g. for comparison with bycatch mortality). However, the difficulty of objectively defining a baseline level (and consequent lack of standardization in how it is done) is a major issue.

In relation to bycatch and hunting, some essentially arbitrary limits have been proposed, e.g. potential biological removal (PBR) and the IWC´s Revised Management Procedure. Nevertheless, in general, it should be possible to identify situations where a population is at risk of extinction and some management action should thus be triggered (cf. precautionary stock reference points in fisheries). However, unlike the case in fisheries, an optimum stock level is not necessarily definable, because there is no implication of optimizing harvesting, which raises the issue of the appropriate advice to give.

Range and distribution: Changes in range can undoubtedly indicate changes in population status but they are extremely difficult to detect, as well as having obvious resource implications. Thus it can be useful to use additional data sources, e.g. strandings data, sightings records from the voluntary sector, while bearing in mind that occurrences of rare species should not be over-interpreted. Changes in distribution within the species range can have a variety of causes and do not necessarily imply a change in population status. Such information will however readily emerge from surveys designed to estimate abundance and as such no additional costs are implied.

Habitat: while there has been a large amount of research on marine mammal habitat use, it is intrinsically difficult to define all the dimensions of a species´ niche and most habitat models explain relatively small proportion of variation in local density. Mobile animals invariably occupy a range of habitat types to differing degrees over different temporal and spatial scales. However, while "essential habitat" of marine mammals may not always be well-defined in terms of environmental characteristics, in a limited number of species it is well-defined in terms of regular use of specific sites, for example nursery, feeding or resting areas. Thus, while it is difficult if not impossible to monitor the availability or quality of suitable habitat in general, for a very limited number of species, monitoring of preferred areas can be valuable.

Health status: information on the prevalence of debilitating and potentially fatal diseases and other causes of ill health (e.g. high contaminant burdens) is fundamental to evaluating population status. Often such information is derived from necropsy and subsequent analysis of samples from strandings and the issue of bias therefore arises. First, not all dead animals are stranded and, second, dead animals will not be representative of the living population (although it may be possible to map one onto the other using a life table. Another issue is the relatively high cost of carrying out full necropsies and, especially, of carrying out certain analyses, e.g. for persistent organic pollutants (POPs). However, because there are known links between POP burdens and both immunosuppression and reproductive success such information is essential (Jepson *et al*., 2005).

Population structure: This is a term that can have several different meanings, including (a) genetic structure (information clearly fundamental to interpretation of status because it helps define "natural populations" and "management units") and (b) age, maturity and sex structure. Aside from allowing construction of a picture of the population structure (which could, presumably, be compared with a "healthy" age structure, sex ratio, etc), such data can provide indicators of population dynamics parameters, in particular mortality and birth rates, as well as individual age at maturity and growth rates. Where this information derives from strandings some caveats exist about biases in available data. However, even if absolute values are suspected to be inaccurate, the information provided on trends is valuable.

Specific threats – bycatch: The favoured method of bycatch monitoring is on-board observation. This has the disadvantage of being expensive, making it logistically difficult to provide adequate coverage of fleets. In addition, the main driver for on-board monitoring, EC Regulation 812/2004, covers only some of the fisheries and boat sizes likely to contribute to marine mammal bycatch. Use of on-board camera systems would allow relatively inexpensive wider coverage, provided that relevant ethical and logistic issues can be overcome. Diagnosis of bycatch during necropsy of stranded animals is an additional source of information and can provide an indication of where more observer coverage is needed. With the caveat that potential biases should be considered and an adequate sample size obtained, necropsy data can also be used to provide a direct estimate of the overall fishery bycatch rate, because an estimate of population mortality rate (proportion of the population dying per year from all causes) can be derived from the age distribution of stranded animals. The value of results from strandings can be illustrated by reference to Spanish data. Monitoring under 812/2004 suggests that very few cetaceans are caught per year in monitored fleets, whereas an average of more than 100 bycaught cetaceans per year are recorded among strandings in Galicia alone (CEMMA, unpublished data). Useful data can also be gathered from interview surveys of fishers and from voluntary reporting/carcase recover schemes, with the obvious caveat that success is higher where fishers feel that their cooperation is viewed in a positive way rather than likely to lead to new restrictions on fishing.

Other information: Among other data that can be collected from stranded animals (or, mainly in the case of seals, from faecal samples) is diet. Dietary information can be used to identify potential competition with fisheries and changes in diet may indicate underlying problems in the availability of prey species.

6.4.6 Framework for monitoring within the ICES Area

WGMME believes that monitoring needs within the ICES Area are well-reflected by current commitments. However, there is a clear need for an improved framework including:

a) improved international coordination of monitoring at the population level rather than a focus on national jurisdictions;

b) a flexible and adaptive approach;

c) critical review of the status indicators used in relation to their statistical power, and of the assessment criteria used and the spatial scale at which they are applied;

d) ensuring that gaps are filled and that results of monitoring that takes place outside Government-sponsored schemes are taken into account, where they meet appropriate levels of quality assurance.

The framework envisaged implies a far greater degree of coordination between numerous organizations than is currently the case. Kick-starting the process is likely to require:

a) Creation of a Steering Committee (including those with responsibility for implementing legislative requirements), to oversee the process;

b) Workshops to discuss common protocols, including baselines and databases, and to form the basis of ongoing networks with regular communication.

This should facilitate an adaptive monitoring process, by which the monitoring objectives, monitoring programmes and the use of monitoring data are all regularly reviewed. Clearly, to be credible and effective, the monitoring outcomes must be linked to appropriate management measures including, where relevant, mitigation measures. The process should also, where appropriate, lead to recommendations for amendments to agreements, directives and legislative drivers which are, in practice, unworkable.

The monitoring parameters listed are not prescriptive, enabling an adaptive and cooperative approach to monitoring depending on the species and regional area. For example, the methods to be used are not specifically identified because the wide variation in abundance, distribution, size and behaviour between species means that no single survey method is the most appropriate to all species. However, if long-term trends are to be detected, sources of variation, such as seasonal movements and vessels, observers etc, need to be minimized. Monitoring within the ICES Area should therefore comprise:

a) Identification of population structure, e.g. genetic structure, age/maturity structure, sex ratio;

b) Trends in abundance, e.g. absolute abundance, relative abundance, "occupancy", population dynamics parameters (mortality rate, birth rate) and their components (e.g. age/size-at-maturity, pregnancy rate at age, incidence of specific causes of mortality);

c) Trends in range and/or distribution. Note: this is inherently difficult to measure;

d) Health status, e.g. condition indices, incidence of diseases, etc;

e) Threats: specific causes of direct mortality or potentially affecting fitness (e.g. by inhibiting reproduction, causing immune-suppression or negatively affecting energy intake), notably fishery bycatch but also naval sonar, seismic surveys, wind farms, vessel and propeller strikes, disturbance, pollution (e.g. POPs, metals), prey depletion, climate change.

6.5 Recommendations

Adoption of an adaptive monitoring and surveillance framework for marine mammals under which objectives, monitoring and outcomes are regularly reviewed and updated by a Steering Group composed of representatives from all relevant bodies. While adaptive monitoring has the advantage that the monitoring programme can respond to changing requirements and constraints, the value of consistently collected long-term datasets should be taken into account.

To further facilitate international coordination of monitoring, we recommend creation of ICES area/Europe-wide networks (e.g. for strandings, sightings, bycatch monitoring) and common databases (and sample banks), and under which the unit of monitoring will be the natural population or (minimally) broad-scale spatial divisions that take into account the transboundary nature of most marine mammal populations. Examples of which already exist include the ICES database for grey and harbour seals in the North-east Atlantic, including the North and Baltic Seas, and the SGBYC database which collates data on bycatch of protected species.

Adoption of a two tier system under which priorities for baseline monitoring for marine mammal will be specified but additional information can be collected and integrated as available. Core priorities are:

a) Surveys for abundance and distribution, being a combination of large-scale dedicated surveys and a range of smaller-scale surveys and alternative data collection methods (tailored to particular species, regions and causes for concern), the integration of which will enhance coverage spatially and temporally (ensuring all seasons are covered and more years are covered) and improve the power to detect trends;

b) Strandings monitoring, with adequate number of necropsies to underpin monitoring of trends in different causes of death (e.g. diseases, contaminants and bycatch mortality) and collection of life-history and dietary data. Where appropriate additional life history, contaminant, health and dietary data to be derived from other sources (e.g. seal faeces, biopsies);

c) Bycatch monitoring: an integrated programme which extends on-board monitoring to cover all fleets known or suspected of causing significant marine mammal bycatches, including observers and possible use of on-board camera systems, supplementing this with voluntary reporting, carcass recover and interview survey programmes, and above all seeks to do this in cooperation with the fishing industry. Thus monitoring should include evaluation of any direct damage caused by marine mammals and cooperation should be rewarded rather than penalized.

Review and improvement of mechanisms to translate monitoring findings into appropriate management action for marine mammals.

Adoption of a coordinated international approach to developing a single assessment for each marine mammal species at an appropriate biological scale when such assessments are required (e.g. the FCS reporting at six yearly intervals).

6.6 References

Chivers, S.J., and Myrick, A.C. 1993. Comparison of age at sexual maturity and other reproductive parameters for two stocks of spotted dolphins, *Stenella attenuata.* Fish Bull (Wash DC) 91:611–618.

Davey, P.A. and Aebischer, N.J. 2009. Participation of the National Gamebag Census in the Mammal Surveillance Network. A report to JNCC for the year 2008/09. The Game and Wildlife Conservation Trust. Research Report Contract No: F90-01-708

European Commission. 2009. Cetacean incidental catches in Fisheries: Report on the implementation of certain provisions of Council Regulation (EC) No 812/2004 and on a scientific assessment of the effects of using in particular gillnets, trammel nets and entangling nets on cetaceans in the Baltic Sea as requested through Council Regulation (EC) No 187/2005. Available on ASCOBANS MOP6 website.

Hohn, A.A. 1989. Variation in life-history traits: the influence of introduced variation. PhD thesis, University of California, Los Angeles.

Jepson, P.D., Bennett, P.M., Deaville, R., Allchin, C.R., Baker, J.R. and Law, R.J. 2005a. Relationships between PCBs and health status in UK-stranded harbour porpoises (*Phocoena phocoena*). Environmental Toxicology and Chemistry, 24, 238–248.

Learmonth, J.A. 2006. Life history and fatty acid analysis of harbour porpoises (*Phocoena phocoena*) from Scottish waters. PhD Thesis.

Lindenmayer, D.B., and Likens, G.E. 2009. Adaptive monitoring: a new paradigm for long-term research and monitoring. Trends in Ecology and Evolution 24(9): 482–486.

Murphy, S., Winship, A., Dabin, W., Jepson, P.D., Deaville, R., Reid, R.J., Spurrier, C., Rogan, E., López, A., González, A.F., Read, F.L., Addink, M., Silva, M., Ridoux, V., Learmonth, J.A., Pierce, G.J., and Northridge, S.P. 2009. Importance of biological parameters in assessing the status of *Delphinus delphis*. Marine Ecology Progress Series 388: 273–291.

Nichols, J.D. and Williams, B.K. 2006. Monitoring for conservation. Trends in Ecology and Evolution, 21 (12): 668–673.

Reid, J.B., Evans, P.G.H. and Northridge, S.P. 2003. Atlas of cetacean distribution in north-west European waters. Joint Nature Conservation Committee, Peterborough. Available at: http://www.jncc.gov.uk/page 2713.

Seavy, N.E. and Reynolds, M.H. 2007. Is statistical power to detect trends a good assessment of population monitoring? Biological Conservation, 140, 187–191.

Taylor, B.L., Martinez, M., Gerrodette, T. and Barlow, J. 2007. Lessons from monitoring trends in abundance of marine mammals. Marine Mammal Science, 23, 157–175.

Wright, AQ.J., Aguilar de Soto, N., Baldwin, A. L., Bateson, M., Beale, C. M., Clark, C., Deak, T., Edwards, E. F., Fernandez, A., Godinho, A., Hatch, L.T. Kakuschke, A., Lusseau, D., Martineau, D., Romero, M.L., Weilgart, L.S., Wintle, B.A., Notarbartolo-di-Sciara, G., and Martin, V. 2007. Do Marine Mammals Experience Stress Related to Anthropogenic Noise? International Journal of Comparative Psychology: Vol. 20: No. 2, Article 15. Available at: http://repositories.cdlib.org/uclapsych/ijcp/vol20/iss2/ art15.

7 ToR d. Review and report on any new information on population sizes, population/stock structure and management frameworks for marine mammals

7.1 European Commission request on EC Regulation 812/2004

ICES received a letter from the European Commission asking for an evaluation regarding possible advice on the several topics, listed below:

1) To provide an assessment of the national reports from 2007 and 2008, and specific scientific reports provided by Member States in the context of Reg. 812/2004;

2) Based on the best available knowledge of the cetacean species concerned by Regulation 812/2004 provide an assessment of the population status and map their yearly distribution and density in European waters since 2004;

3) Identify areas outside the scope of Reg. 812/2004 where measures would be necessary to be applied to reduce the incidental catches of cetaceans;

4) Provide an evaluation of mitigation measures currently in place and an assessment on the most recent developments of mitigation measures used to reduce the incidental catches of cetaceans, including information on cost;

5) Following the assessment made in point 4) identify the most efficient mitigation measure for each species concerned by Reg.812/2004 and according to the fishing gear in use.

The SGBYC fully addressed item 1 earlier this year, and the WGMME was asked to address item 2. A response to items 3–5 will be drafted at a separate workshop, proposed to be held later this year.

With regard to item 2, our interpretation of the request is that the following questions are also implied: (a) have the measures taken by the EC regulation 812/2004 had a noticeable effect, and (b) are there any areas in particular where further consideration of the measures is required. The latter will be dealt with further in the response to item 3 at the Workshop proposed to be held later this year.

For this request we will be reporting on "natural biological populations" in both the Northeast Atlantic and Mediterranean Sea.

7.1.1 North-east Atlantic

Within the North-east Atlantic the two main species of concern with regards to Regulation 812/2004 are the harbour porpoise and common dolphin. Therefore we have included only these species in our assessment. Other cetacean species have been reported as bycatch in fishing gear of 812/2004 fleets in recent years within the North-east Atlantic, such as the striped dolphin, bottlenose dolphin and (one) pilot whale (ICES SGBYC, 2009), although not regularly or in such large numbers. For further information on abundance, distribution and habitat use of other cetacean species, see Section 7.3 (CODA distribution and density maps) and previous reports by the WGMME, i.e. Section 7 (CODA abundance data) in 2009, and Section 4 in 2007 (SCANS-II abundance data and density maps).

The EU requested that WGMME map the yearly distribution and density of bycaught species in European waters since 2004. With the information currently available for harbour porpoises and common dolphins, it is not possible to present annual distribution and density maps. Since 2004, three large-scale surveys have been undertaken within the North-east Atlantic, which did not overlap in their area of coverage. These surveys were:

a) SCANS-II[1] (Small Cetacean Abundance in the North Sea and adjacent waters) which surveyed continental shelf waters ranging from southern Norway (c60ºN) to the straits of Gibraltar in July 2005 Figure 11a);

b) CODA (Cetacean offshore Distribution and Abundance in European waters) which surveyed waters off the continental shelves of Britain, Ireland, France and Spain in July 2007 (Figure 11b);

c) T-NASS (Trans North Atlantic Sightings Survey) which was also undertaken in July 2007 and surveyed waters to the west of the area covered by CODA and more northern European waters.

It should be noted that as all these surveys were undertaken during July, they do not provide any information on distribution and abundance of cetaceans at other times of year. Comprehensive information is not available on seasonal movements or interannual variation in abundance/densities for different regions in the North-east Atlantic.

SCANS-II and CODA produced spatial distribution/density maps for their respective survey areas. However, to date T-NASS has published only sightings data for common dolphins in the entire survey area (ranging from waters west of Norway to North America).

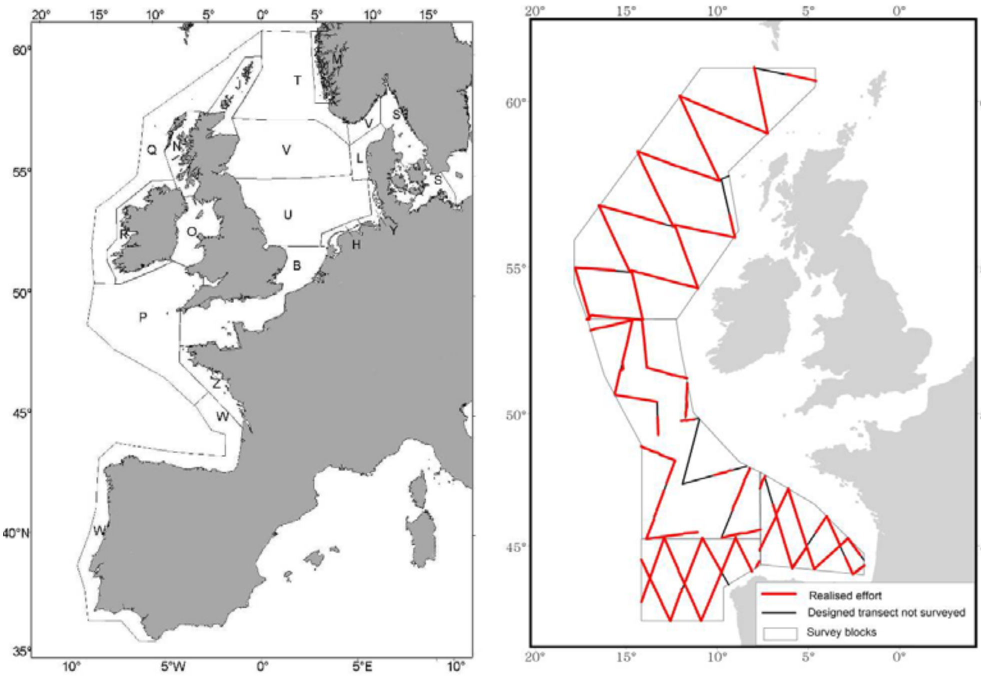


Figure 11. (a) Survey blocks defined for the SCANS II survey. Blocks S, T, V, U, Q, P and W were surveyed by ship. The remaining blocks were surveyed from aircraft (SCANS-II 2008), (b) CODA survey region divided into the survey blocks, and survey route (in red) (CODA 2009).

1 This covered a similar, but slightly larger, area to the 1994 SCANS survey.

7.1.1.1 Harbour porpoise

Population structure

Recent assessments of population and stock structure were undertaken by the WGMME in 2009, and the ASCOBANS-HELCOM Small Cetacean Population Structure Working Group (Evans *et al*., 2009). A summary of the results from the ASCOBANS-HELCOM report is presented in Section 7.2. A single continuous population exists in the North-east Atlantic, ranging from waters off France to northern Norway, although there is significant isolation by distance (i.e. the greater the distance the smaller the genetic correlation). A separate Iberian population exists, the range of which is not yet fully described but which includes the Portuguese and Spanish Atlantic coasts. The ASCOBANS-HELCOM Working Group proposed a number of Management Units within the North-east Atlantic which are outlined in Section 7.2.

Distribution and abundance

In the North-east Atlantic, the harbour porpoise is common and widely distributed from the Barents Sea and Iceland in the north to the waters off the Iberian coast in the south – although there appears to be a break in the distribution in the southern Bay of Biscay (Basque area). This species is mainly confined to shelf waters, though sightings have occurred in deep waters; for example between Faroe Islands and Iceland. The 'Cetacean Atlas' collated all available cetacean sightings data and mapped the distribution of the harbour porpoise in western European waters, which is shown in Figure 12 (Reid *et al*., 2003). The data used to produce this map spanned the time period 1978 to 1998. Within this region, highest population densities were found in the Belt Sea to the east of Denmark and in the northwestern North Sea, in waters shallower than ca. 100 m. Few sightings were reported in the English Channel. High densities were also reported off the Faroes Islands, western Scotland, southwest Ireland, and southwest Wales. A second edition of the 'Cetacean Atlas' is currently in preparation and this will incorporate data obtained from 1998 onwards, with publication anticipated in 2013.

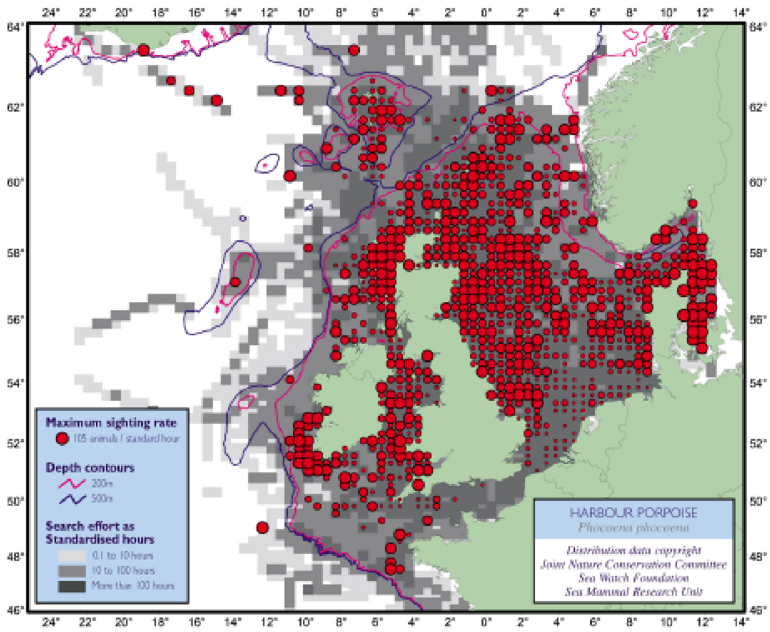


Figure 12. Distribution of harbour porpoises in western European waters (data obtained from 1978–1998). Taken from Reid et al. (2003).

From line transect surveys (SCANS) conducted in July 1994, the population abundance in a portion of continental shelf waters was estimated at 341 366 (CV = 0.14; 95% CI 260 000–449 000) individuals, including *ca.* 250 000 in the North Sea, 33 000 in the Baltic Sea, and 36 000 in the Celtic Sea (Evans *et al*., 2008, Hammond *et al*., 2002). In July 2005 SCANS-II covered a wider geographical area and produced an estimate of 386 000 (CV=0.20; 95% CI: 261 300–569 200) individuals for the European continental shelf, and an estimate of 335 000 harbour porpoises for the region surveyed in 1994 (SCANS-II 2008).

Baltic Sea and adjacent waters

For the Kattegat, Belt Seas and western Baltic Sea (Block S), SCANS-II estimated 23 227 porpoises, whereas SCANS estimated 36 000 animals in July 1994 (Hammond *et al*., 2002) - though it should be noted that the design blocks surveyed in 1994 and 2005 were not the same, and therefore these estimates are not directly comparable.

Within the Polish Baltic Sea, although acoustic and visual survey effort amounted to 1602 km, no sightings were recorded of harbour porpoises, though two probable acoustic detections were made (SCANS-II 2008). 599 (CV=57%; 95% CI = 200–3300) harbour porpoises were estimated for an area corresponding to ICES Subdivisions 24 and 25 but excluding a 22 km wide corridor off the Polish coast (Hiby and Lovell, 1996; Hammond *et al*., 2008). The most recent estimate for the Baltic Sea population was 93 (95% CI 10-460) in 2002, indicating that there is no apparent improvement in the number of porpoises within the Baltic (Berggren *et al*., 2004). Little is known about its current distribution in the inner Baltic, but its status is highly critical. The International Union for the Conservation of Nature (IUCN) has listed harbour porpoises in the Baltic Sea as 'critically endangered' justified by the consideration that the current population size is likely to be fewer than 250 mature individuals.

North Sea

Overall abundance in the North Sea did not change substantially between the two SCANS surveys. However, results from the SCANS-II survey showed that between 1994 and 2005, there was a southerly shift in the distribution, with densities in the southern part of the North Sea increasing while densities in more northern regions, such as off Shetland, Orkney and eastern Scotland, declined by similar amounts (SCANS-II 2008; see Figure 13a and Figure 13b). Prior to SCANS-II there was a notable increase in individuals in the southern North Sea (Camphuysen *et al*., 2004; Kiszka *et al*., 2004a; Haelters and Camphuysen, 2009), especially during winter and early spring. This strongly suggests that the difference reflects a shift in distribution (SCANS-II 2008).

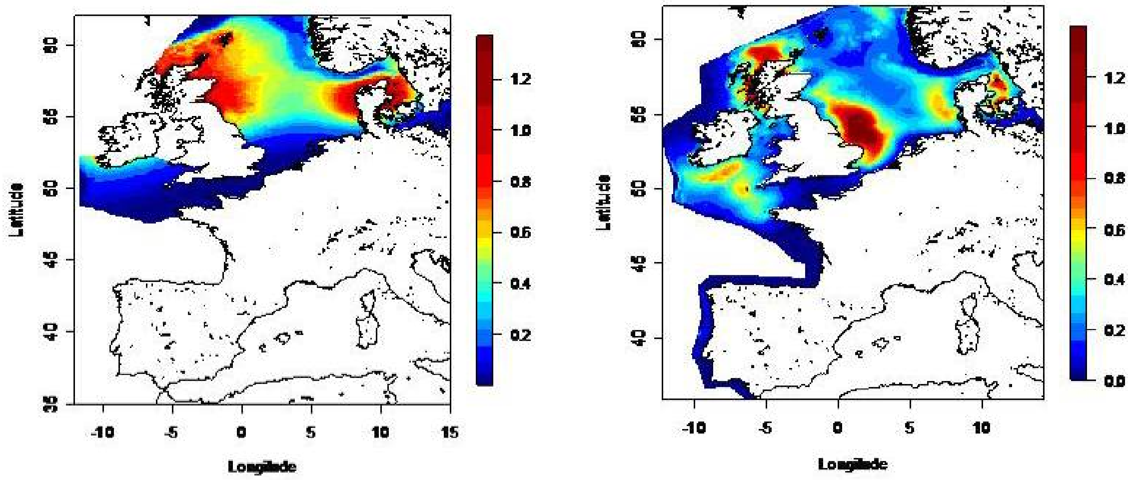


Figure 13. Estimated harbour porpoise density (animals per km²) in July in (a) 1994 and (b) 2005 (SCANS-II, 2008).

During SCANS II, 40 900 (CV = 0.38) individuals were estimated in the Channel and contiguous southern North Sea. It should be noted, however, that the majority of these animals were sighted in the southern North Sea, as the number of animals sighted in the Channel still remained low, apart from off the southwest coast of the UK – the most western part of the English Channel (see Figure 14).

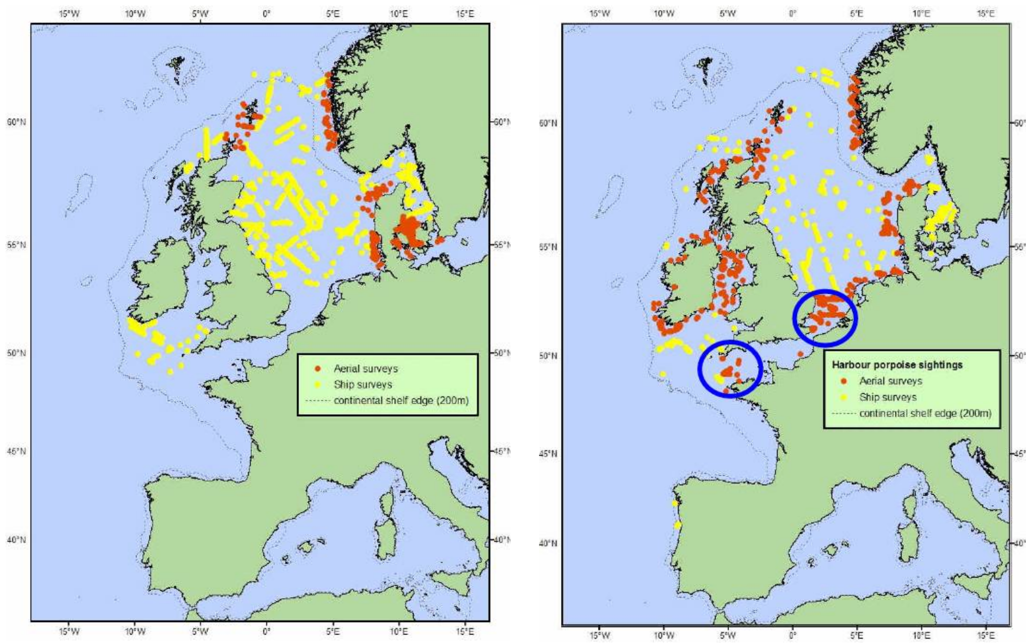


Figure 14. Sightings of harbour porpoises during (a) SCANS-I in July 1994 and (b) SCANS-II in July 2005. Note SCANS-II survey extended into the Bay of Biscay and waters off Iberia. Note that the SCANS-I survey did not include the Irish Sea, or waters to the west of Scotland and Ireland (SCANS-II, 2008).

Bay of Biscay

Off the French coast, no harbour porpoises were reported (visually or acoustically) within the inner Bay of Biscay (SCANS-II Block Z) or in the outer Bay of Biscay, west of France (Figure 4b). However, this result is not representative of the year-round distribution or abundance of porpoises within this region. High bycatch rates have been reported in the inner Bay of Biscay, with an estimated ca. 600 porpoise caught in 2008 (ICES SGBYC 2009), and since 2002 an increase in harbour porpoise strandings has been reported along the Atlantic coast of France (Van Canneyt *et al*., 2009).

Celtic Sea/SW Approaches

During the SCANS survey in July 1994, there were an estimated porpoise abundance of 36 280 (CV = 0.57) animals in the Celtic Sea and adjacent shelf waters (Hammond *et al*., 2002). In July 2005, the number of animals in this region was higher, with an estimated 80 600 (CV = 0.50) individuals (Table 9). Highest densities in 2005 were reported off southern Ireland and the southwest coast of the UK.

At present, it is not known why harbour porpoise abundance has increased in the Celtic Sea and western English Channel. As with the North Sea, a likely explanation is the movement of animals into these waters since 1994. Macleod *et al*. (2009) reported a significant trend in increasing occurrence of harbour porpoises in summer in the English Channel between 1996 and 2006.

Iberia

Only one abundance estimate of 2600 (CV = 0.80) porpoises exists for the Iberian population, which was obtained by the SCANS-II project. Although this only provides a snap-shot of summer (July) abundance of porpoises in this region in 2005, the extremely low abundance estimate is a cause for concern.

Table 9. Results from SCANS-II: estimates of group abundance, mean group size, animal abundance and animal density (individuals.km^{-2}) for *P. phocoena*. CVs are given in parentheses. Figures in square brackets are 95% confidence intervals. There were no sightings of harbour porpoise in block Z.

Block	Group abundance	Mean group size	Animal abundance	Animal density
B	32 052 (0.39)	1.28 (0.04)	40 927 (0.38)	0.331 (0.38)
H	3138 (0.37)	1.24 (0.16)	3891 (0.45)	0.355 (0.45)
J	8294 (0.37)	1.24 (0.08)	10 254 (0.36)	0.274 (0.36)
L	9152 (0.43)	1.26 (0.04)	11 575 (0.43)	0.555 (0.43)
M	3230 (0.37)	1.22 (0.08)	3948 (0.38)	0.305 (0.38)
N	9309 (0.41)	1.30 (0.07)	12 076 (0.43)	0.394 (0.43)
O	11 118 (0.36)	1.37 (0.07)	15 230 (0.35)	0.335 (0.35)
P	25 334 (0.52)	3.18 (0.21)	80 613 (0.50)	0.408 (0.50)
Q	7679 (1.27)	1.30 (0.19)	10 002 (1.24)	0.067 (1.24)
R	7685 (0.35)	1.39 (0.10)	10 716 (0.37)	0.278 (0.37)
S	14 788 (0.34)	1.57 (0.09)	23 227 (0.36)	0.340 (0.36)
T	11 519 (0.35)	2.06 (0.12)	23 766 (0.33)	0.177 (0.33)
U	54 357 (0.28)	1.19 (0.09)	88 143 (0.23)	0.562 (0.23)
V	19 909 (0.32)	2.37 (0.22)	47 131 (0.37)	0.294 (0.37)
W	1022 (0.77)	2.59 (0.15)	2646 (0.80)	0.019 (0.80)
Y	1473 (0.47)	1.00 (0.00)	1473 (0.47)	0.125 (0.47)
Total	220 059 (0.18) [64 984–532 333]		385 617 (0.20) [261 266–569 153]	

Other areas

Information to the north and south of the area covered by SCANS and CODA is more limited. An earlier survey undertaken in the 1990s reported 11 000 porpoises in waters north of 66ºN and the Barents Sea (Bjorge and Øien, 1995).

Data from other sources – strandings and local scale sightings work

Southern North Sea

Because of the large southerly shift in the harbour porpoise distribution in the North Sea between the 1990s and 2000s, it is not known if the results from SCANS-II, obtained in July 2005, are representative of this species' current summertime distribution in this region. Using both sightings and strandings data from the Netherlands and Belgium, Haelters and Camphuysen (2009) reported a peak in harbour porpoise numbers in coastal waters of the southern North Sea between February and April. In late spring, a northward migration towards more offshore waters occurs and, by summer, smaller numbers of porpoises are reported using coastal waters of the southern North Sea (Haelters and Camphuysen, 2009). Observations during 2007 and 2008 have indicated that this apparent seasonal pattern might not be stable. However, these data do indicate that the spring (February–April) estimate of abundance for harbour porpoises off the Dutch and Belgium coast is likely to be higher than that for summer (i.e. SCANS-II 2005).

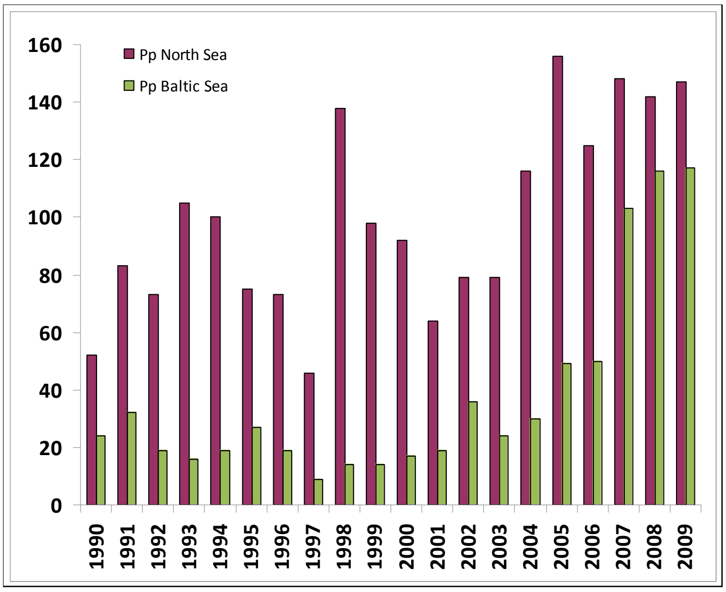
Strandings data from German, Dutch and Belgium waters in the southern North Sea are presented in Figure 15. What is apparent is the continued increase in reported strandings in this region from 2000 onwards. Although strandings data showed a slight decline in 2007 and 2008, high levels were reported again in Dutch waters in

2009 (>400 individuals). These data suggest that high densities of harbour porpoises are still occurring in southern North Sea, at least during spring (period of highest strandings). The high stranding rates within the southern North Sea are attributed to high bycatch rates within region as up to half of the stranded porpoises along the Dutch and Belgium coasts were identified as bycatch (Haelters and Camphuysen, 2009).

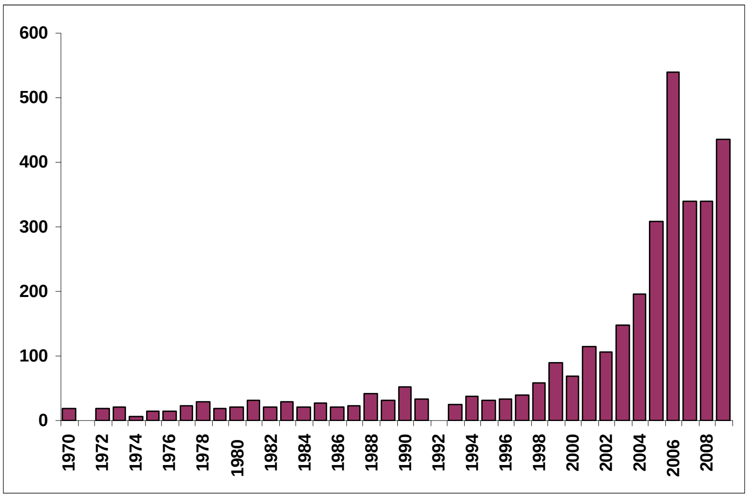
Interestingly, strandings along the German Baltic Sea coast have also increase since 2000 (Figure 15a). However, there is no indication of a population increase in the western Baltic that could explain the increase in stranding occurrence (Jastarnia Group and Bräger, 2009). Minimum bycatch estimates for this region were determined using different approaches, which produced estimates of 51, 82, 150 and 69 porpoises. When these four different bycatch estimates are applied to the local abundance estimates to calculate bycatch rates, all resulting rates are above 1%, with most of the rates above 1.7% or considerably higher (Jastarnia Group and Bräger, 2009).

In contrast along the English North Sea coast a decline in reported porpoise strandings has been observed in recent years. Further, UK post-mortem data suggests that bycatch as a cause of death in stranded animals has reduced (Deaville and Jepson, 2009, see Figure 16). The reasons for this reduction cannot be elucidated at this time. It should however be noted that the fleet sectors known or suspected to have the highest rates of porpoise bycatch are not currently covered by EU Regulation 812/2004 and, therefore, the implementation of this regulation is unlikely to account for this change. It could be that, due to changes in the distribution, porpoises are not coming into as much contact with the industry or the recorded reduction in fishing effort would also have lead to a reduction in bycatch.

(a)


160
140
120
100
80
60
40
20
0
Pp North Sea
Pp Baltic Sea
1990
1991
1992
1993
1994
1995
1996
1997
1998
1999
2000
2001
2002
2003
2004
2005
2006
2007
2008
2009

(b)


600
500
400
300
200
100
0
1970
1972
1974
1976
1978
1980
1982
1984
1986
1988
1990
1992
1994
1996
1998
2000
2002
2004
2006
2008

(c)

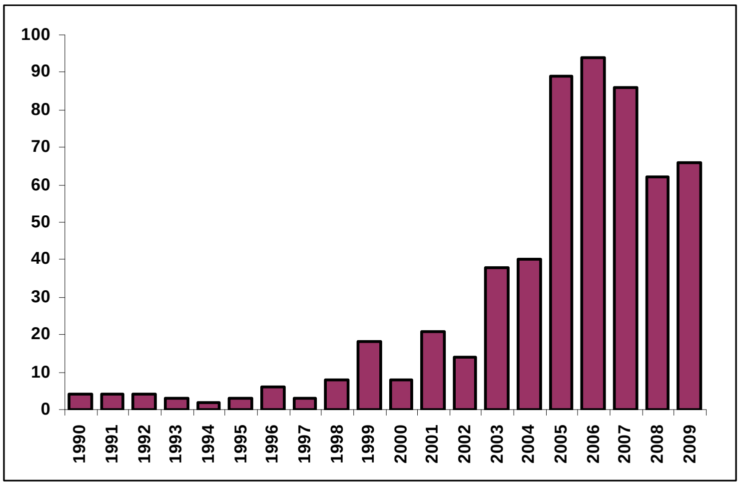
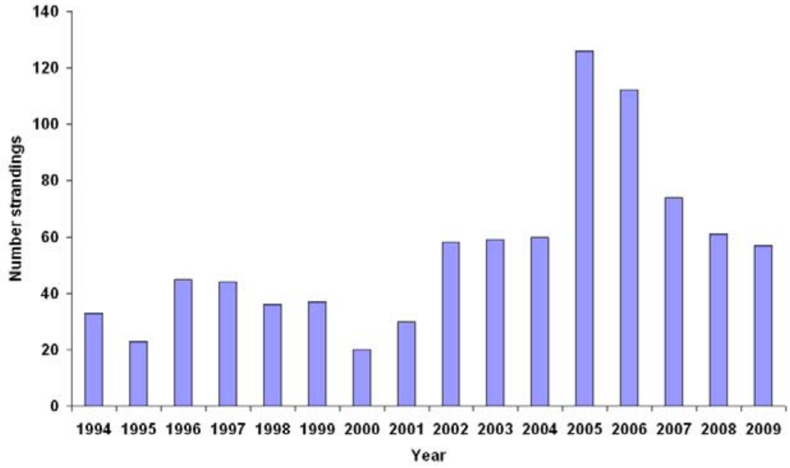


Figure 15. Strandings of harbour porpoises along (a) German - Schleswig-Holstein (Hasselmeier, unpublished data) (b) Dutch (Camphuysen, unpublished data[2]), and (c) Belgium coasts (Haelters, unpublished data).

2 Minimum estimate for 2009.

(a)



(b)

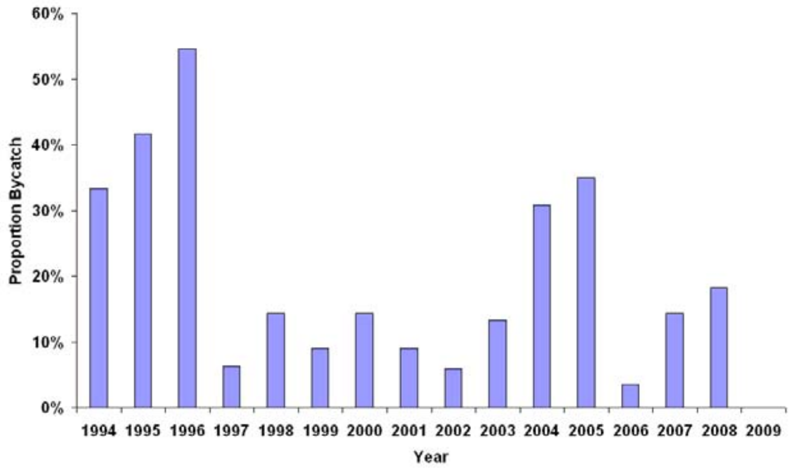


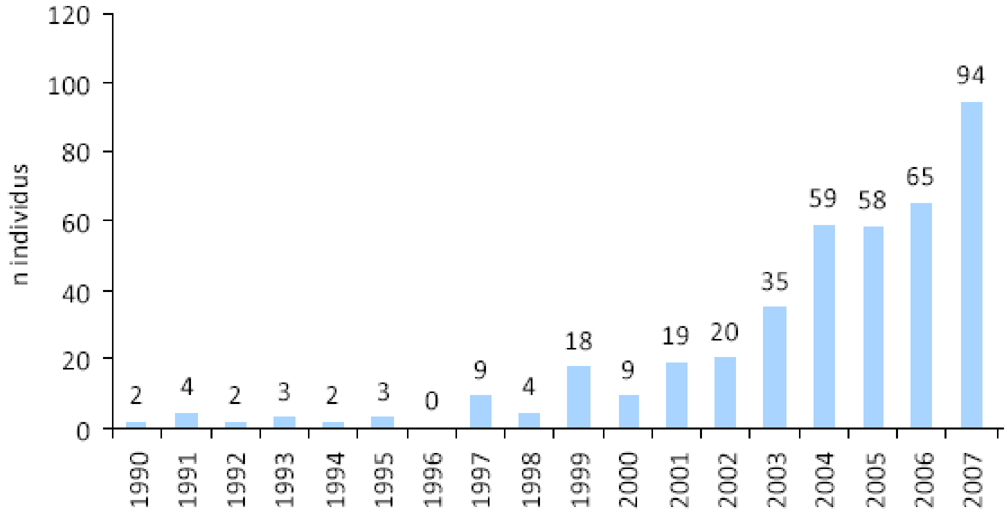
Figure 16. (a) The number of strandings reported per year for the whole English Channel and English North Sea coasts and, (b) of those for which a post-mortem was undertaken, the proportion of individuals that died as a result of bycatch for the English Channel and the English North Sea areas. Note: in 2009 none of the post-mortems (n = 8) identified bycatch as the cause of death for the Channel/English North Sea areas (Deaville, unpublished data,).

France and the Iberian waters

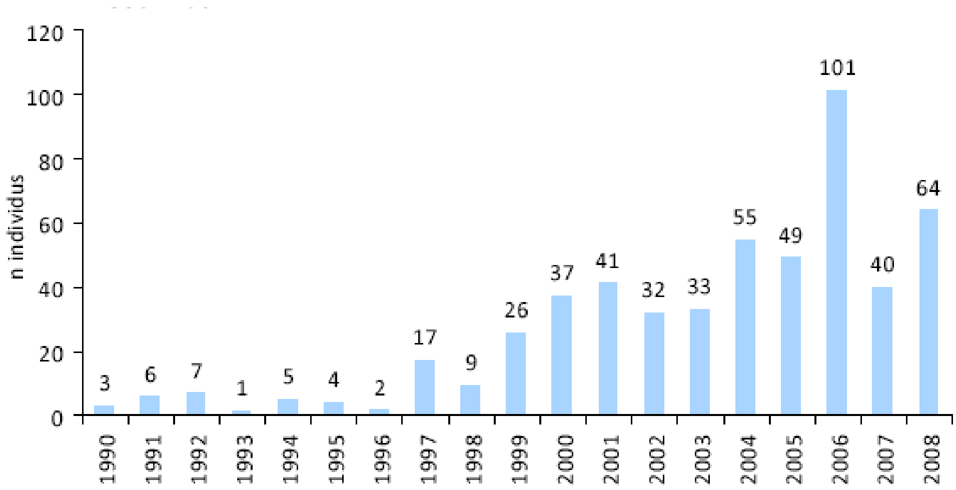
The recent increase in strandings of harbour porpoises along both the channel and Atlantic coasts of France since 2002 (see Figure 17a, b) has been mirrored by increases in sightings particularly in northern parts of France (Kiszka *et al.*, 2004b; 2007).

During the last decade, an average of 9.5 porpoises stranded per year in Galicia of which 29.6% showed signs of bycatch (Figure 17). Earlier data (1990–1999) indicated 10.4 porpoise strandings/year with a larger proportion of bycatches (31.1%) (CEMMA, unpublished data, see Table 10a). Of the 94 porpoises that stranded along the northern and central Portuguese coast between 2000 and 2009, 37.2% had evidence of bycatch. Excluding the decomposed animals (code ≥4), 50.7% of the remaining porpoises had signs of incidental capture (see Table 10b, M. Ferreira, unpublished data).

(a)


120
100
80
60
40
20
0
n individus
2
4
2
3
2
3
0
9
4
18
9
19
20
35
59
58
65
94
1990
1991
1992
1993
1994
1995
1996
1997
1998
1999
2000
2001
2002
2003
2004
2005
2006
2007

(b)


120
100
80
60
40
20
0
n individus
3
6
7
1
5
4
2
17
9
26
37
41
32
33
55
49
101
40
64
1990
1991
1992
1993
1994
1995
1996
1997
1998
1999
2000
2001
2002
2003
2004
2005
2006
2007
2008

(c)

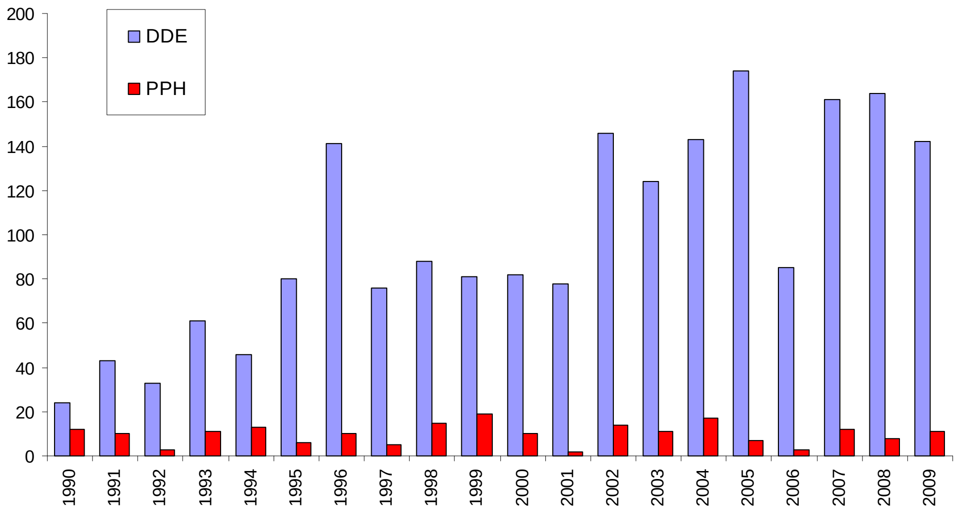


Figure 17. Numbers of harbour porpoises that stranded along the (a) Channel and the (b) Atlantic coasts of France (taken from Van Canneyt *et al.*, 2009); and (c) the number of harbour porpoises (PPH) and common dolphins (DDE) that stranded along the Galician coastline (1990–2009; CEMMA, unpublished data).

Age data derived from stranded porpoises in Galicia and northern Portugal have been used to construct a life table, providing an estimated annual population mortality rate of around 15%. Depending on which figures for the proportion of bycatch mortalities are used, this would translate into bycatch mortality of between 3% and 9% of the population per annum. While this is based on a relatively small sample size and it is difficult to account for all potential biases, these figures suggest that the bycatch rate is above the recommended limit of 1.7% per annum (F. Read *et al.*, unpublished data).

Table 10a. Bycatch diagnosis from harbour porpoise strandings in Galicia, 1990–1999 and 2000–2009.

Time-period	Strandings	External examination	Evidence of bycatch	% bycatch*
1990–1999	104	45	14	31.1
2000–2009	95	27	8	29.3

*** These figures are expressed in relation to the number of animals examined externally and represent a minimum estimate.**

Table 10b. Bycatch diagnosis from harbour porpoise strandings in Centre/North Portugal, 2000–2009.

TIME-PERIOD	STRANDINGS	EXTERNAL EXAMINATION	EVIDENCE OF BYCATCH	% BYCATCH*
2000–2009	94	94	35**	37.2

****19 were addressed to beach purse-seines, 12 had evidence of monofilament net marks (gillnet or trammelnet) and four had other bycatch evidences (e.g. multifilament marks, removed dorsal musculature or amputated fins).**

Summary

It should be noted that the 812/2004 Regulation is not fully implemented by all EU countries and does not cover all boats or fisheries known, or suspected, to have porpoise bycatch. Further, the distribution and abundance data available cannot be used to evaluate the impact of this regulation because:

a) No large-scale survey of coastal waters has taken place since 2005.

b) Available data suggest that the North-east Atlantic harbour porpoise population had a similar abundance and range but different distribution pattern in 2005 compared with 1994. However, as the SCANS-II survey was undertaken in 2005, the year that implementation of the 812 regulation began (including use of pingers), any effects of the 812/2004 Regulation would not have been seen.

c) The nature of trends that would have occurred in the absence of the regulation is clearly unknown.

Other indicators, e.g. strandings, point to a continued high rate of porpoise bycatch in some fisheries in particular areas. The main fishing gears responsible for the porpoise bycatch in the southern North Sea are gill- and tanglenets (Haelters and Camphuysen, 2009), much of which is undertaken by vessels not covered by EU Regulation 812/2004.

7.1.1.2 Common dolphin

In 2005 the WGMME reviewed all available literature and unpublished data on common dolphins for assessing the population status, and also interactions with fisheries, within of the North-east Atlantic.

Population structure

The short-beaked common dolphin is the only *Delphinus* species recognized in the North Atlantic. In 2009, the WGMME agreed with the main conclusions from the ASCOBANS/HELCOM Report (Murphy *et al*., 2009) which suggested that only one *D. delphis* population exists in the Northeast Atlantic ranging from waters off Scotland to Portugal, with separate populations in the Mediterranean Sea and the Northwest Atlantic.

Distribution and abundance

Common dolphins are abundant and widely distributed throughout the North-east Atlantic, with summer sightings as far north as approximately 70ºN latitude, west of Norway (Murphy, 2004, and ref. therein; Cañadas *et al*., 2009, and ref. therein), although the majority of common dolphin sightings have been reported in waters south of 60°N. Distribution patterns in western European waters show long-term changes. During the 1930s to 1970s, an increase in strandings were reported along the Dutch and Danish coasts, which coincided with a decline along the Irish and the southern and western English coasts; this strongly suggests a shift in the general distribution during that period, and it has been proposed that this may have been related to North Atlantic Oscillation (Murphy, 2004, and ref. therein).

Using sightings data obtained between 1963 and 2007, although the majority of sightings were obtained after 1980 and predominately during summertime, it appears that *D. delphis* are distributed, at least during summertime, from coastal waters in the North-east Atlantic to the mid Atlantic ridge, and as far south as the Azores (Figure 18). In fact, *D. delphis* may be distributed across the whole North Atlantic, between 35 and 55°N (partially covering a region heavily influenced by the Gulf Stream/North Atlantic drift). However, as a consequence of a lack of observer effort, beyond the mid Atlantic ridge (approx. 30–40°W), the full distributional range of common dolphins in the North Atlantic Ocean is not known (Murphy *et al*., 2009). Further, the actual distributional boundary of the Northeast Atlantic population is also not known, as to date sampling of individuals for genetic analysis has been confined to continental shelf and slope waters and oceanic waters of the Bay of Biscay.

Using sightings data obtained between 1978 and 1998, Reid *et al*., (2003) mapped the distribution of common dolphins within western European waters as part of the 'Cetacean Atlas'. Highest numbers were reported to the west of Ireland, and in the Celtic Sea, southern Irish Sea/St Georges Channel, the western English Channel and off the west coast of Brittany (see Figure 19). Off western Scotland, sightings peak in June–July, then decline markedly; in the Irish Sea, numbers peak in summer but individuals are present throughout winter, particularly in south; in the western English Channel, and south to the Bay of Biscay, common dolphins occur year-round, but numbers are highest in winter (Murphy *et al*., 2008, and ref. therein).

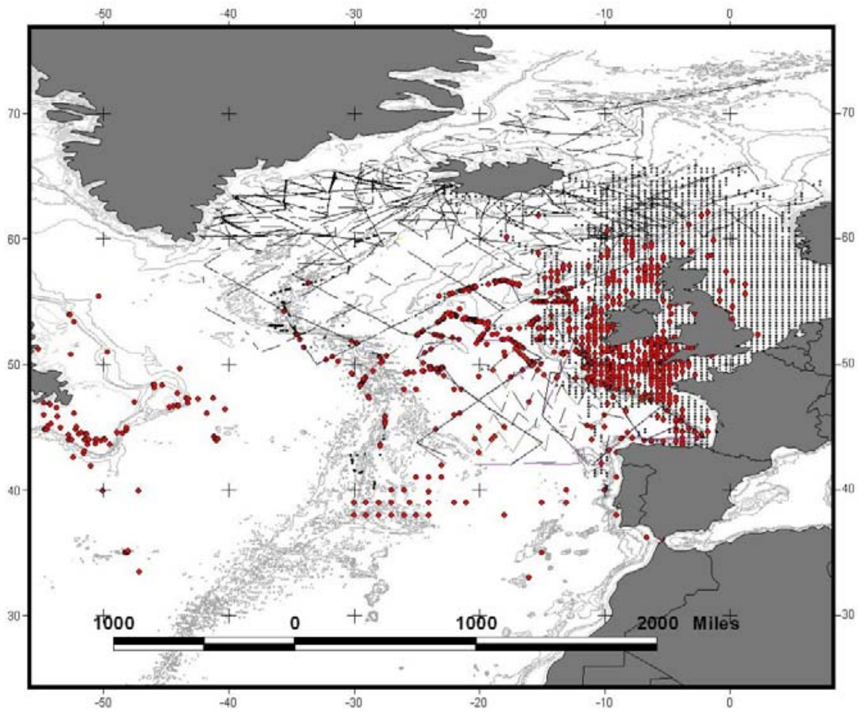


Figure 18. Outline of available information on track lines and areas covered (black dots) by various surveys in the Northeast Atlantic (taken from Murphy *et al*., 2009).

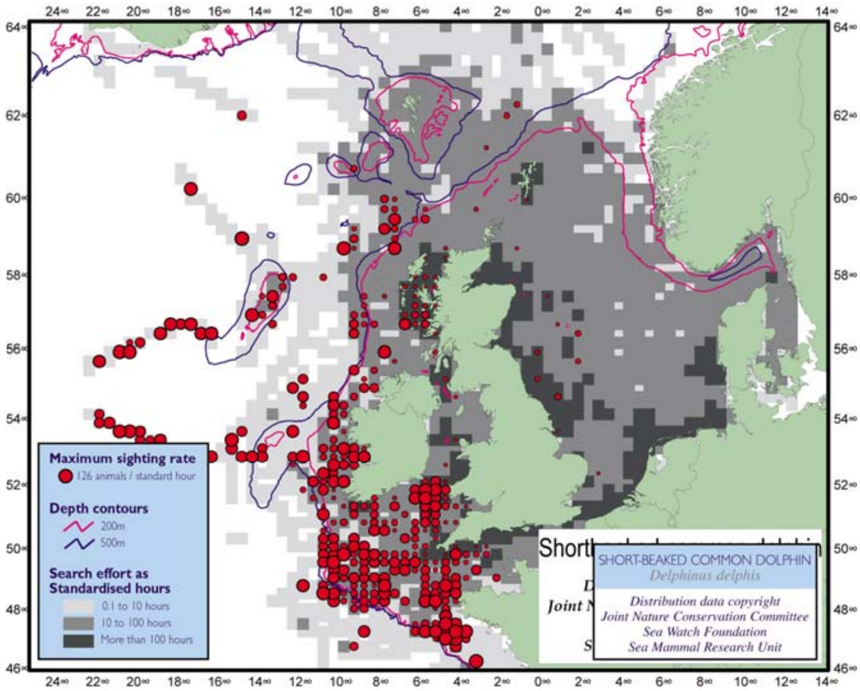


Figure 19. Distribution of common dolphins in western European waters (data obtained from 1978–1998). Taken from Reid *et al*. (2003).

In July 2005, an estimated 50 507 (CV = 0.29) *D. delphis* were reported in continental shelf and slope waters (Hammond *et al*., in prep).

North Sea

Low numbers of sightings were reported in the North Sea between 1978 and 1998 (Reid *et al*., 2003). During SCANS-II in July 2005, no common dolphins were observed within the North Sea (Figure 20). However strandings and sightings of common dolphins have been reported in Danish and Dutch waters in more recent years; six common dolphins stranded along the Danish coastline between 2001 and 2003 and smaller schools containing up to ten individuals have been sighted (ICES WGMME, 2005). Further since 1993, common dolphins have been reported (sightings and strandings) in Swedish, Norwegian, German, Polish and Finnish waters (ICES WGMME, 2005; Øien and Hartvedt, 2009). 13 groups of common dolphins (ranging from 2 to 450+ individuals) have been observed during summertime within the outer Moray Firth/Scottish North Sea waters since 2006 (Robinson *et al*., in press).

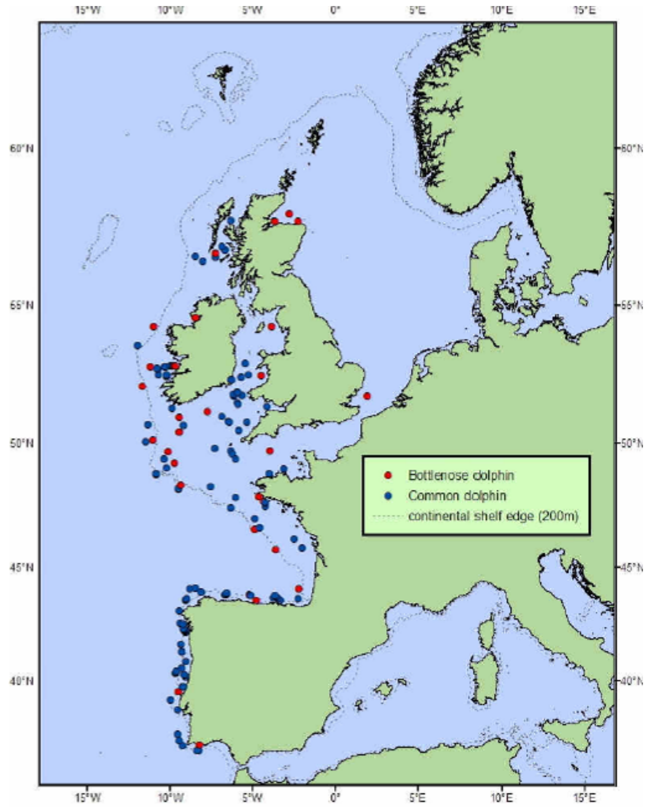


Figure 20. Sightings of common dolphins during SCANS-II in July 2005 (SCANS-II 2008).

West of Scotland and Ireland

Macleod *et al*. (2005) reported an increase in the abundance (sighting and stranding records) of common dolphins off the northwest Scottish coast during the period 1948 to 2003, and attributed this to an increase in the sea surface temperature (SST) in the region.

SCANS-II estimated 11 661 *D. delphis* in near shore waters off the west coast of Ireland, 2199 in near shore waters off the Scottish west coast, and an additional 1454 individuals were sighted in continental shelf waters off western Ireland and western and northern Scotland during July 2005 (Table 3).

Celtic Sea, western approaches, western English Channel and Irish Sea

SCANS-II reported 11 141 (CV = 0.61) common dolphins in the Celtic Sea and contiguous shelf waters, 4919 (CV = 0.82) individuals in the western English Channel and 825 (CV = 0.78) in the Irish Sea in July 2005 (Table 11). Highest densities were reported in the Celtic Sea and extending into St George's Channel/southern Irish Sea, along the continental shelf off southwest Ireland and west of Brittany, and in the western English Channel.

Table 11. Results from SCANS-II, estimates of group abundance, mean group size, animal abundance and animal density (individuals.km^{-2}) for (a) *D. delphis* (Hammond *et al.*, in prep). CVs are given in parentheses. Figures in square brackets are 95% confidence intervals. There were no sightings of *D. delphis* in blocks H, J, L, M, S, T, U, V and Y.

Block	Group abundance	Mean group size	Animal abundance	Animal density
B	378 (0.73)	13.0 (0.36)	4919 (0.82)	0.040 (0.82)
N	1256 (0.58)	1.8 (0.14)	2199 (0.60)	0.072 (0.60)
O	375 (0.69)	2.2 (0.36)	825 (0.78)	0.018 (0.78)
P	999 (0.31)	11.2 (0.57)	11 141 (0.61)	0.056 (0.61)
Q	505 (0.85)	2.9 (0.39)	1454 (0.81)	0.010 (0.81)
R	1266 (0.70)	9.2 (0.19)	11 661 (0.73)	0.302 (0.73)
W	1434 (0.26)	12.5 (0.17)	17 916 (0.22)	0.129 (0.22)
Z	314 (0.84)	1.3 (0.20)	392 (0.86)	0.012 (0.86)
Total	6527 (0.26) [3970–10 732]		50 507 (0.29) [28 742–88 751]	

Strong seasonal movements have been reported within this region, with dolphins being more widely dispersed in deeper offshore waters during summer (May–October) compared with the winter period (November–April); when there is a pronounced concentration in the shelf waters of the western English Channel and further offshore parts of the Celtic Sea (WGMME 2005). Using sightings data from platforms of opportunity, Brereton *et al*. (2005) reported that the shallow Brittany coast and western English Channel supports large numbers of common dolphins during winter (December to February), with a reported tenfold increase in sightings of dolphins in the western English Channel at that time. Kiszka *et al*. (2007) also analysed sightings data obtained opportunistically on board ferries operating, predominately, between July and October. During this period (summertime), aggregations were largest in the northern Bay of Biscay compared with the western English Channel. It has been suggested that these seasonal movements may be correlated with prey availability and distribution (ICES WGMME, 2005).

Between 1996 and 2006 an increase in the occurrence of common dolphins was noted in the English Channel during winter (MacLeod *et al*., 2009). This increase is correlated with an increase in reported strandings of this species along the southwest coast of the UK during winter (Jepson *et al*., 2005).

For a large number of common dolphins that strand along the UK coastline, predominantly in the southwest, cause of death has been attributed to incidental capture (see Figure 21). However in 2008, out of 43 autopsied stranded *D. delphis* only two individuals were diagnosed as bycatch; the slight increase in strandings observed in 2008 in Figure 22 is attributed to a mass stranding event in Cornwall (n = 26). This is a marked reduction compared with the previous 18 year period where bycatch was the most common cause of death for a large number of common dolphins that, predominantly, stranded in southwest England (Cornwall and Devon) between January and April (Deaville and Jepson, 2010). The reason/s for the reduction in numbers of stranded common dolphins (and harbour porpoises) that were diagnosed as bycatch in 2008 (mainly in southwest England) is not known.

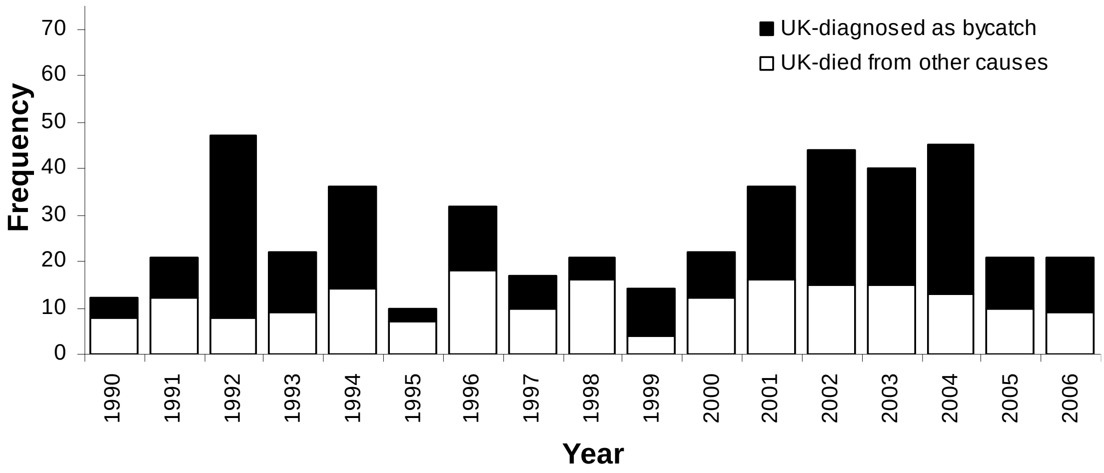


Figure 21. Cause of death for necropsied common dolphins from the UK (n= 461, 1990-2006). (Taken from Murphy, 2008).

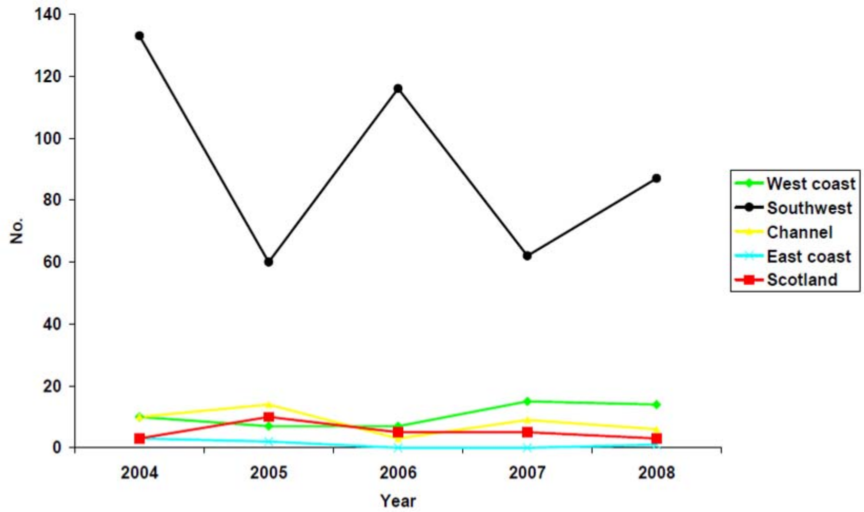
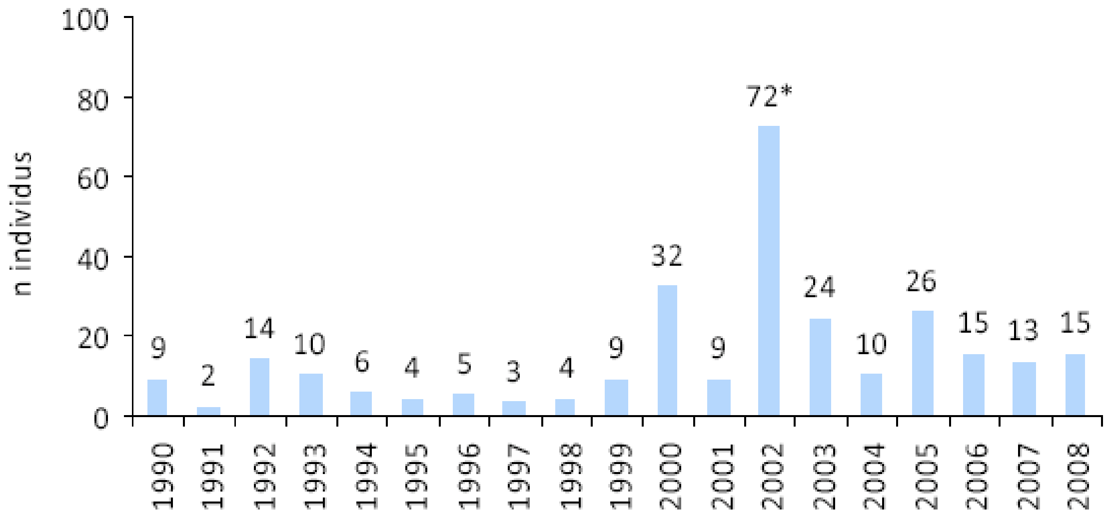


Figure 22. Inter-annual variation in UK regional strandings of common dolphins (2004–2008). (Taken from Deaville and Jepson, 2009).

French and Iberian waters

SCANS-II estimated 17 916 common dolphins in continental shelf waters in the southern Bay of Biscay (south of Brittany) and off Iberia in July 2005.

The number of common dolphin strandings along the Channel and Atlantic coast of France are presented in Figure 23. The peak in strandings along the Channel coast in 2002 was attributed to a mass live stranding event.



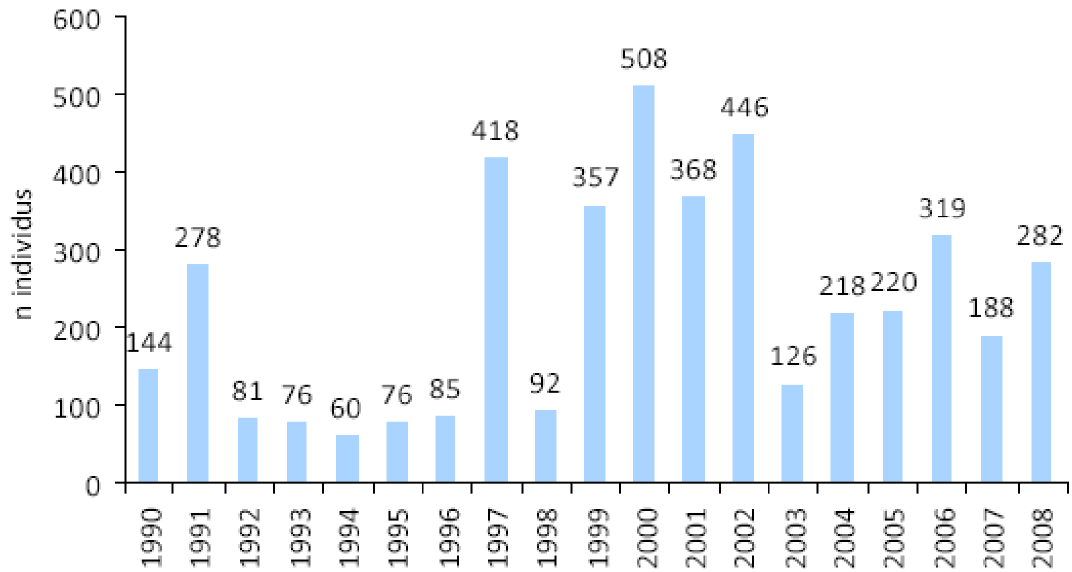


Figure 23. Number of strandings of common dolphins along the (a) Channel (n=282) and (b) Atlantic coasts of France (n=4342) (Van Canneyt *et al.*, 2009).

The spatial distribution of common dolphins in May in the Bay of Biscay is shown in Figure 24. Data were obtained over a six year period from 2003 and 2008 by PELGAS surveys coordinated by Ifremer to assess pelagic fish spawning. Although it appears that common dolphins are distributed throughout the Bay of Biscay at this time of year, areas of highest abundance occur between the upper Gironde river plume to waters off the Vendée coast, the canyons areas in the south of the Bay (Cap Ferret and around), and in coastal waters off Brittany (G. Certain, unpublished data).

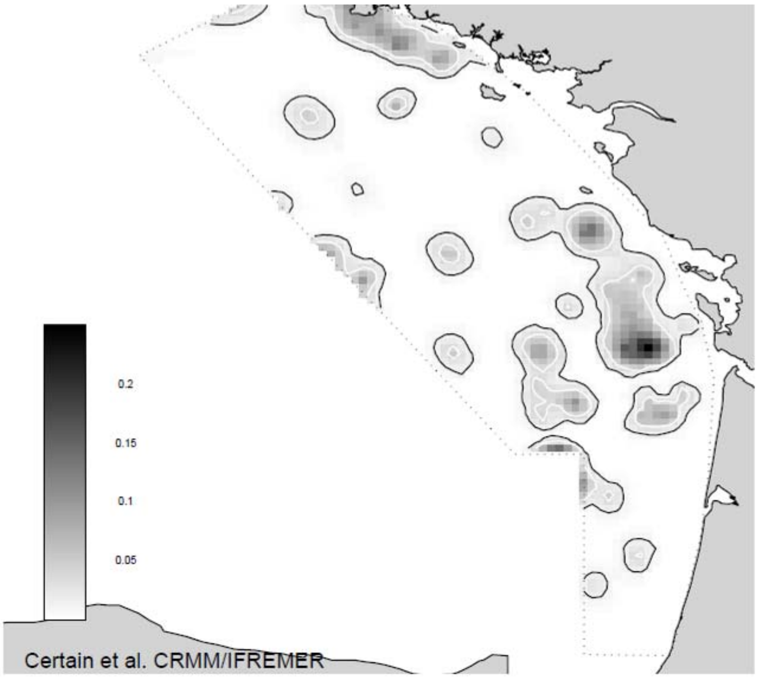


Figure 24. Surface maps of smoothed predicted abundance of common dolphins in the Bay of Biscay in May (G. Certain, unpublished data). Data obtained between 2003 and 2008.

Along the Galician coast, an increase in the number of stranded *D. delphis* has been observed since 2000, with a peak of 174 individuals reported stranded in 2005 (see Figure 17c).

Offshore waters

The recent CODA survey reported 116 709 (CV = 0.34) *D. delphis* in European offshore waters (beyond the continental shelf) in July 2007 (CODA 2009). Highest densities were observed along the continental shelf slope, west of France (see Table 12, Figure 25). 4216 *D. delphis* were reported off the west coast of Scotland and northwest coast of Ireland; 52 749 individuals off the southwest coast of Ireland and further offshore waters off the west of France; 21 071 individuals in the southern Bay of Biscay; and 38 673 off the northwest coast of Spain.

Table 12. Results from CODA, model-based (DSM) abundance estimates. Figures in parentheses are CVs. Figures in square brackets are 95% confidence intervals. From CODA (2009).

Species	Block	Abundance of animals (CV)	95% Confidence Interval
Common dolphin	1	4216 (0.57)	1478–12 027
	2	52 749 (0.39)	25 054–111 059
	3	21 071 (0.51)	8270–53 689
	4	38 673 (0.46)	16 464–90 839
	Total	116 709 (0.34)	61 397–221 849

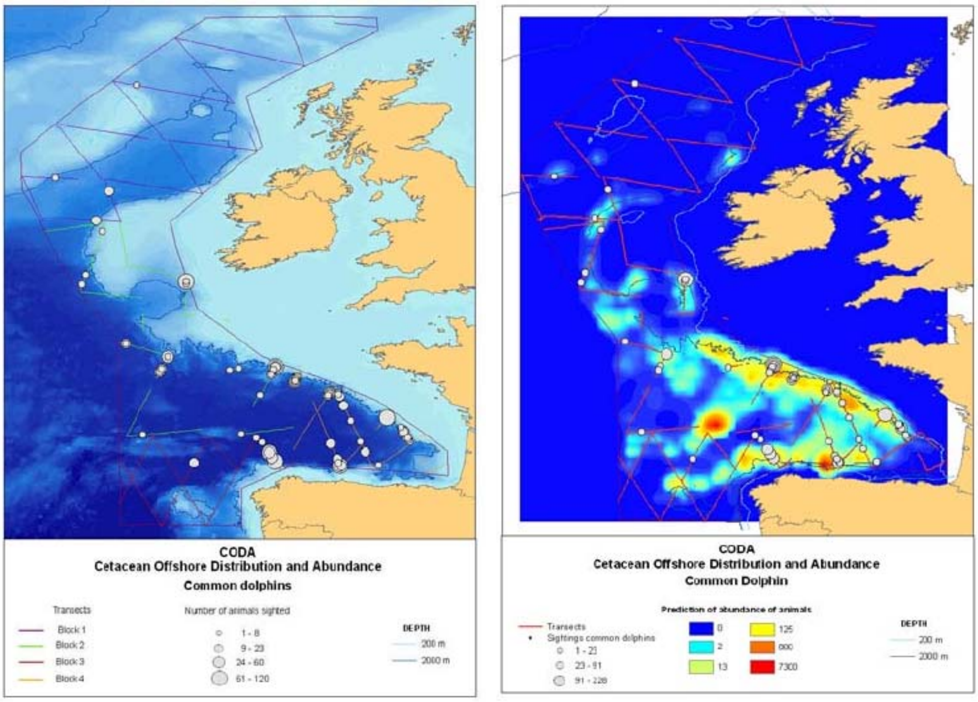


Figure 25. (a) Distribution of sightings (circles proportional to group size) of common dolphins and (b) surface maps of smoothed predicted abundance of common dolphins in offshore waters (CODA, 2009).

The small numbers sighted off the west coast of Ireland during the CODA survey are similar to those reported in the SIAR survey undertaken in 2000. SIAR, which surveyed waters over the shelf break north and west of Ireland during summertime, estimated only 4496 individuals for this region (Ó Cadhla *et al.*, 2003). However, both these results are in contrast to the large numbers sighted off the west of Ireland during the 1990s. 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. The estimated abundance in the W Block of the NASS-95 Faroese survey, which is beyond the CODA survey region, was 273 159 (CV=0.26; 95% CI=153 392–435 104) short-beaked common dolphins (Cañadas *et al.*, 2009). Unfortunately, an estimate is not available for NASS block east (west of Ireland).

T-NASS surveyed waters further offshore during the same period as the CODA survey in July 2007, and the distribution of sightings for common dolphins in the North Atlantic is shown in Figure 26. Very small numbers of common dolphins were sighted in areas where animals were seen in high abundance during the NASS 1995 surveys. In 2007, no sightings of common dolphins were made south of 57°N in the central strata with the southern border at 52.5°N and 27° and 37°W (Lawson *et al.*, 2009). In 2009, the IWC Sub-Committee on Small Cetaceans noted that based on these results there did seem to be a change in density of short-beaked common dolphins west beyond the CODA area. Several potential reasons for this were identified: i) differences in sighting conditions, e.g. sea state, ii) uncertain species identification (as other dolphin species were sighted), iii) a true reduction in common dolphin density, iv) ship effect and v) interannual distributional shifts. In addition, due to poor weather conditions, some of the survey tracks were not covered in these areas (IWC, 2009).

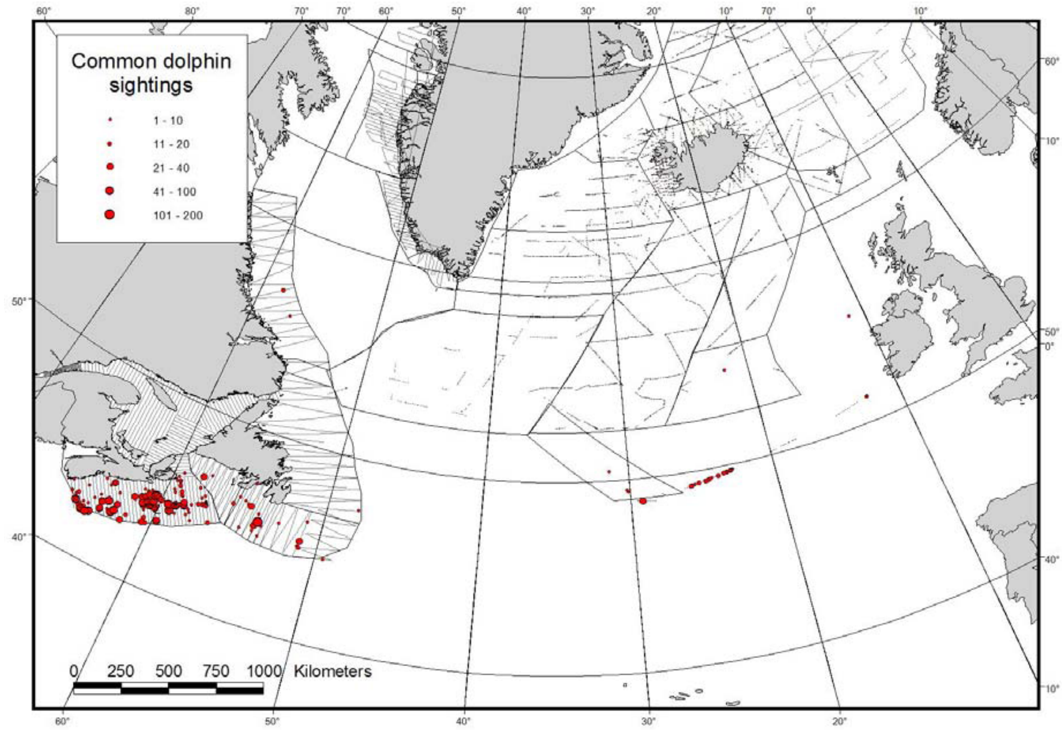


Figure 26. Sightings of short-beaked common dolphins, *Delphinus delphis*, made during T-NASS in July 2007 (Taken from Lawson *et al.*, 2009).

Summary

The distribution and abundance data available for *D. delphis* cannot be used to evaluate the impact of this regulation because:

a) No large-scale survey of coastal waters has taken place since 2005.

b) Due to the way SCANS-I collected sightings data for common dolphins in 1994 (single platform method used, did not correct for animals missed on the trackline, or responsiveness), we do not have baseline data to compare SCANS-II results against. Further, the SCANS-II survey was undertaken in 2005, the year that implementation of the 812/2004 Regulation began (including use of pingers), and therefore any effects of the 812/2004 Regulation would not have been seen.

c) Available data suggest that the distribution (and habitat use) of the Northeast Atlantic common dolphin population may be changing, though additional analysis and investigations are required to investigate this further. At the current point in time, Hammond and Cañadas (pers. comm) are currently re-analysing all SCANS-II, CODA and T-NASS data (incorporating data from the Faroe Islands) to produce a current surface density map for the Northeast Atlantic *D. delphis* population.

d) The decline in stranded common dolphins diagnosed as bycatch in UK waters in 2008 is unknown. Whether or not this reflects a change in the distribution of animals, a reduction in fishing effort and/or due to effective mitigation needs to be assessed.

e) The nature of trends that would have occurred in the absence of the regulation is clearly unknown.

7.1.2 References

Berggren, P., Hiby, L., Lovell, P., and Scheidat, M. 2004. Abundance of harbour porpoises in the Baltic Sea from aerial surveys conducted in summer 2002. 16pp. Paper SC/56/SM7 submitted to the Scientific Committee of the International Whaling Commission. Available from www.iwcoffice.org.

Bjorge, A., and Oien, N. 1995. Distribution and abundance of harbour porpoise, *Phocoena phocoena*, in Norwegian waters. Report of the International Whaling Commission (Special Issue 16): 89–98.

Brereton, T., Williams, A., and Martin, C. 2005. Ecology and status of the common dolphin *Delphinus delphis* in the English Channel and Bay of Biscay 1995–2002. Proceedings of the workshop on common dolphins: current research, threats and issues, Special Issue April 2005, pp 15–22. Kolmarden, Sweden 1st April, 2004. K. Stockin, A. Vella and P. Evans (Eds).

Camphuysen, K. 2004. The return of the harbour porpoise (*Phocoena phocoena*) in Dutch coastal waters. Lutra 47(1): 135–144.

Cañadas, A., Donovan, G.P., Desportes, G., and Borchers, D.L. 2009. A short review of the distribution of short beaked common dolphins (*Delphinus delphis*) in the central and eastern North Atlantic with an abundance estimate for part of this area. North Atlantic Sightings Surveys. NAMMCO Scientific Publications, Volume 7: 201–220.

CODA. 2009. Cetacean Offshore Distribution and Abundance in the European Atlantic (CODA). 43pp without appendices; 164pp with appendices.

Deaville, R., and Jepson, P. 2009. UK Cetacean Strandings Investigation Program. Annual Report for the period 1st January–31st December 2008 (Contract number CR0364). 56pp.

Evans, P., Andersen, L.W., Bjørge, A., Fontaine, M., Galatius, A., Kinze, C.C., Lockyer, C., De Luna, C., Pierce, G.J., Sveegaard, S., Teilmann, J., Tiedemann, R., and Walton, M. 2009. Harbour porpoise *Phocoena phocoena*. Report of ASCOBANS/HELCOM small cetacean population structure workshop. 8–10 October 2007, UN Campus, Hermann-Ehlers-Str. 10, 53113 Bonn, Germany.

Evans, P.G.H., Lockyer, C.H., Smeenk, C.S., Addink, M., and Read, A.J. 2008. Harbour porpoise *Phocoena phocoena*. In Mammals of the British Isles. Handbook 4th Edition. Edited by S. Harris and D.W. Yalden. The Mammal Society. pp. 50–55.

Haelters, J., and Camphuysen, C.J. 2009. The harbour porpoise in the southern North Sea. Abundance, threats and research and management proposals. Report commissioned by IFAW (International Fund for Animal Welfare). 60pp.

Hammond, P.S., Berggren, P., Benke, H., Borchers, D.L., Collet, A., Heide-Jorgensen, M.P., Heimlich, S., Hiby, A.R., Leopold, M.F., and Oien, N. 2002. Abundance of harbour porpoise and other cetaceans in the North Sea and adjacent waters. Journal of Applied Ecology 39(2): 361–376.

Hammond, P.S., Bearzi, G., Bjørge, A., Forney, K., Karczmarski, L., Kasuya, T., Perrin, W.F., Scott, M.D., Wang, J.Y., Wells, R.S. and Wilson, B. 2008. *Phocoena phocoena (Baltic Sea subpopulation)*. In: IUCN 2010. IUCN Red List of Threatened Species. Version 2010.1. <www.iucnredlist.org>. Downloaded on 18 May 2010.

Hiby, L. and Lovell, P. 1996. Baltic/North Sea aerial surveys. Unpublished report.

ICES SGBYC. 2009. Report of the Study Group for Bycatch of protected species (SGBYC), 19–22 January 2009 ICES, Copenhagen, Denmark.

ICES WGMME. 2005. ICES. Report of the Working Group on Marine Mammal Ecology. 9–12 May 2005, Savolinna, Finland.

IWC. 2009. Report of the Sub-Committee on Small Cetaceans. Annex L. International Whaling Commission Report.

Jastarnia Group (ASCOBANS), and Bräger, S. 2009. Decline of the harbour porpoise (*Phocoena phocoena*) in the southwestern Baltic Sea. HELCOM Indicator Fact Sheets 2009. Online. [28 April 2010], http://www.helcom.fi/environment2/ifs/en_GB/cover/.

Jepson, P.D., Sabin, R.C., Spurrier, C.J.H., Chimonides, P.D.J., Jepson, P.D., Deaville, R., Perkins, M., Cunningham, A.A., Reid, R.J., Patterson, I.A.P., Foster, G., Barley, J., Penrose, R., and Law, R.J. 2005. Trends in cetacean strandings around the UK coastline and cetacean and marine turtle post-mortem investigations, 2000 to 2004 inclusive (Contract CRO 238), Cetaceans Strandings Investigation and Coordination in the UK. Report to Defra for the period 1st January 2000–31st December 2004.

Kiszka, J.J., Haelters, J., and Jauniaux, T. 2004a. The harbour porpoise (*Phocoena phocoena*) in the southern North Sea: a come-back in northern French and Belgian waters? Document AC11/Doc.24(P/R) presented at the 11th Advisory Committee Meeting to ASCOBANS, Jastrzebia Góra, Poland, 27–29 April, 2004. 4 pp.

Kiszka, J.J., Hassani, S., and Pezeril, S. 2004b Distribution and status of small cetaceans along the French Channel coasts: using opportunistic records for a preliminary assessment. *Lutra*, 47: 33–46.

Kiszka, J., Macleod, K., Van Canneyt, O., Walker, D., and Ridoux, V. 2007. Distribution, encounter rates, and habitat characteristics of toothed cetaceans in the Bay of Biscay and adjacent waters from platform-of-opportunity data. ICES J. Mar. Sci. 64(5): 1033–1043.

Lawson, J., Gosselin, J.-F., Desportes, G., Acquarone, M., Heide-Jørgensen, M.P., Mikkelsen, B., Pike, D., Víkingsson, G., Zabavnikov, V., and Øien, N. 2009. A note on the distribution of short-beaked common dolphins, *Delphinus delphis*, observed during the 2007 T-NASS (Trans North Atlantic Sightings Survey), International whaling Commission Report, SC/61/SM35.

MacLeod, C.D., Bannon, S.M., Pierce, G.J., Schweder, C., Learmonth, J.A., Herman, J.S., and Reid, R.J. 2005. Climate change and the cetacean community of north-west Scotland. Biological Conservation 124: 477–483.

MacLeod, C.D., Brereton, T., and Martin, C. 2009. Changes in the occurrence of common dolphins, striped dolphins and harbour porpoises in the English Channel and Bay of Biscay. Journal of the Marine Biological Association of the UK 89(Special Issue 05): 1059–1065.

Murphy, S. 2004. The biology and ecology of the common dolphin *Delphinus delphis* in the North-east Atlantic. PhD., University College Cork.

Murphy, S. 2008. Investigating biological parameters in common dolphins and harbour porpoises. Annex to Final Report to the UK Department for Environment Food and Rural Affairs, Project MF0736, Sea Mammal Research Unit. 38pp.

Murphy, S., Evans, P.G.H., and Collet, A. 2008. Common dolphin *Delphinus delphis*. In Mammals of the British Isles. Handbook 4th Edition. Edited by S. Harris and D.W. Yalden. The Mammal Society. pp. 719–724.

Murphy, S., Natoli, A., Amaral, A.R., Mirimin, L., Viricel, A., Caurant, F., Hoelzel, R., and Evans, P. 2009. Short-beaked common dolphin *Delphinus delphis*. Report of ASCOBANS/HELCOM small cetacean population structure workshop. 8–10 October 2007, UN Campus, Hermann-Ehlers-Str. 10, 53113 Bonn, Germany. 111–130.

Ó Cadhla, O., Mackey, M., Aguilar de Soto, N., Rogan, E., and Connolly, N. 2003. Cetaceans and Seabirds of Ireland's Atlantic Margin. Volume II – Cetacean distribution and abundance. Report on research conducted under the 1997 Irish Petroleum Infrastructure Programme (PIP): Rockall Studies Group (RSG) projects 98/6, 99/38 and 00/13. 83pp.

Øien, N., and Hartvedt, S. 2009. Common dolphins *Delphinus delphis* in Norwegian waters. International Whaling Commission Report. SC/61/SM 9.

Reid, J.B., Evans, P.G.H., and Northridge, S.P. 2003. Atlas of cetaceans distribution in north-west European waters. Joint Nature Conservation Committee (JNCC), Peterborough.

Robinson, K.P., Eisfeld, S.M., Costa, M., and M.P., S. in press. Short-beaked common dolphin (*Delphinus delphis*) occurrence in the Moray Firth, northeast Scotland. Marine Biodiversity Records 3.

SCANS-II. 2008. Small cetaceans in the European Atlantic and North Sea (SCANS II). Final report to the European Commission under contract LIFE04NAT/GB/000245.

Van Canneyt, O., Dars, C., Gonzalez, L., and Dorémus, G. 2009. Les échouages de mammifères marins sur le littoral français en 2008. Rapport CRMM pour le Ministère de l'Ecologie, de l'Energie, du Développement Durable et de la Mer, Direction de l'eau et de la biodiversité, Programme Observatoire du Patrimoine Naturel: 31pp.

7.1.3 Mediterranean Sea

Within the Mediterranean Sea the two main species of concern with regards to Regulation 812/2004 are the common dolphin and the striped dolphin, as these are the species most frequently reported as bycatch (ICES SGBYC, 2009). This assessment, which focuses on both species, was undertaken using reports provided by ACCOBAMS (Agreement on the Conservation of Cetaceans of the Black Sea, Mediterranean Sea and contiguous Atlantic area; including ACCOBAMS 2004; 2008; 2010); as well as the 2009 report of the IWC Sub-Committee on Small Cetaceans which assessed the status of all common dolphin populations, including the Mediterranean Sea population; and a report by Aguilar *et al*. (submitted) on striped dolphins for the IUCN Red List assessment. It should be noted that some information on bycatch of bottlenose dolphin in the Adriatic Sea by Italian pelagic pair trawls was provided last year as part of the Memorandum of Understanding between ICES and the European Commission.

In absence of a large-scale survey in the Mediterranean Sea, similar to that of SCANS-II and CODA, localized actions have been made to assess and monitor areas of interest for marine mammal conservation. An outline of all surveys undertaken for common and striped dolphins prior to 2004 is presented in Table 13 (taken from (Northridge and Fortuna, 2008). Since 2004, among others, the French Scientific Group of Interest for Mediterranean Marine Mammals (GIS3M) has carried out studies in the Pelagos Sanctuary in the northwestern Mediterranean Sea, which focused on assessing the distribution of both large whale species (Sperm whale and Fin whale) and small cetaceans, including striped dolphins. Italy conducted two aerial surveys within the Pelagos Sanctuary during summer and winter of 2008 and 2009, respectively. Whereas Spain has primarily concentrated efforts in the Alborán Sea and ongoing sightings surveys undertaken by ALNITAK (Cañadas *et al*., 2008) in this region have focused on a range of species, including the common dolphin and striped dolphin.

Due to the lack of a large-scale survey of the Mediterranean Sea we are unable to present distribution or density maps for the common and striped dolphin. Further, in the majority of cases, maps and data from studies undertaken since 2004 are not yet available.

7.1.3.1 Common dolphin

Population structure

Natoli *et al*. (2008) undertook a study to assess the population structure of common dolphins in the Mediterranean Sea and contiguous waters. Results from this study indicated a comparatively low rate of migration between the Mediterranean and the Black Seas with a separate subspecies (*D. d. ponticus* Barabasch, 1935) recognized in the Black Sea, and a clear population boundary between the western (Alborán Sea) and the eastern (Ionian Sea) Mediterranean; indicating the presence of discrete population in these two areas. These results suggest that the eastern and western regions of the Mediterranean Sea should be considered independently in further actions towards the conservation of this species (Natoli *et al*., 2008).

Distribution and abundance

The common dolphin Mediterranean Sea population is believed to have suffered a steep decline during the last 30–40 years, and in 2003 it was is listed as 'endangered' in the IUCN Red List. In 2006, it was included in Appendix I and II of the Convention on the Conservation of Migratory Species (Bonn Convention - CMS). The causes of their generalized decline in areas where this species was known to have inhabited, such as the central and eastern Mediterranean Sea, remain poorly understood but are thought to include prey depletion and bycatch. However, as large regions of the Mediterranean Sea have not been fully surveyed, i.e. regions of the southern Mediterranean, it is not known if some animals may have redistributed there (IWC, 2009). Several recent sightings off southern Israel (A. Scheinin, pers. comm. to G. Notarbartolo di Sciara) corroborate this view.

Bearzi *et al*. (2003) presented an overview of the ecology and conservation status of the common dolphin in the Mediterranean Sea. This species is now only relatively abundant in the westernmost portion of the basin (Alborán Sea), with sparse sightings records off Algeria and Tunisia, small concentrations around the Maltese islands and in parts of the Aegean Sea, and relict groups in the southeastern Tyrrhenian and eastern Ionian Seas (see Figure 27).

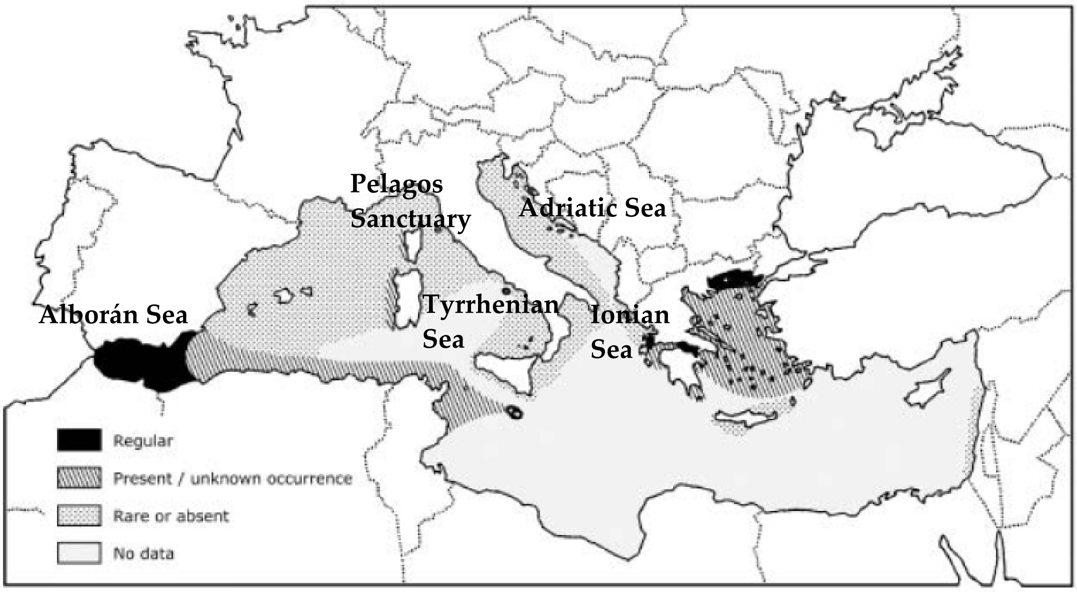


Figure 27. Approximate distribution of short-beaked common dolphins in the Mediterranean Sea (taken from Bearzi *et al*., 2003).

Western Mediterranean Sea

The Alborán Sea and Gulf of Vera are considered as the most important remaining western Mediterranean habitat for common dolphins (Cañadas and Hammond, 2008). Spatial modelling was used to estimate abundance and explore habitat use of common dolphins in this area using data obtained between 1992 and 2004. For this region (Gulf of Vera and the northern part of the Alborán Sea), the point estimate of abundance was 19 428 (95% CI = 15 277 to 22 804) dolphins. This abundance estimate is not directly comparable with an earlier estimate by Forcada and Hammond (1998) of 14 736 *D. delphis* for this region, as the later study only surveyed the northern part of the Alborán Sea which included continental shelf waters, the area where highest densities of *D. delphis* have been reported, whereas the Forcada and Hammond estimate was for the whole basin, excluding coastal/continental shelf waters.

Cañadas and Hammond (2008) reported that the average density was higher in summer than in winter, and higher in the northwestern Alborán Sea than in the eastern Gulf of Vera. No overall trend in abundance was observed in the northern Alborán area. However, a decline was observed in the Gulf of Vera, with a summer density threefold lower in the period from 1996 to 2004 than in 1992 to 1995 (see Figure 28). Groups with calves and feeding groups preferred more coastal waters. A new abundance estimate and maps for this region incorporating data collected between 2004 and 2009 will be available at the end of this year (Cañadas, pers. comm.).

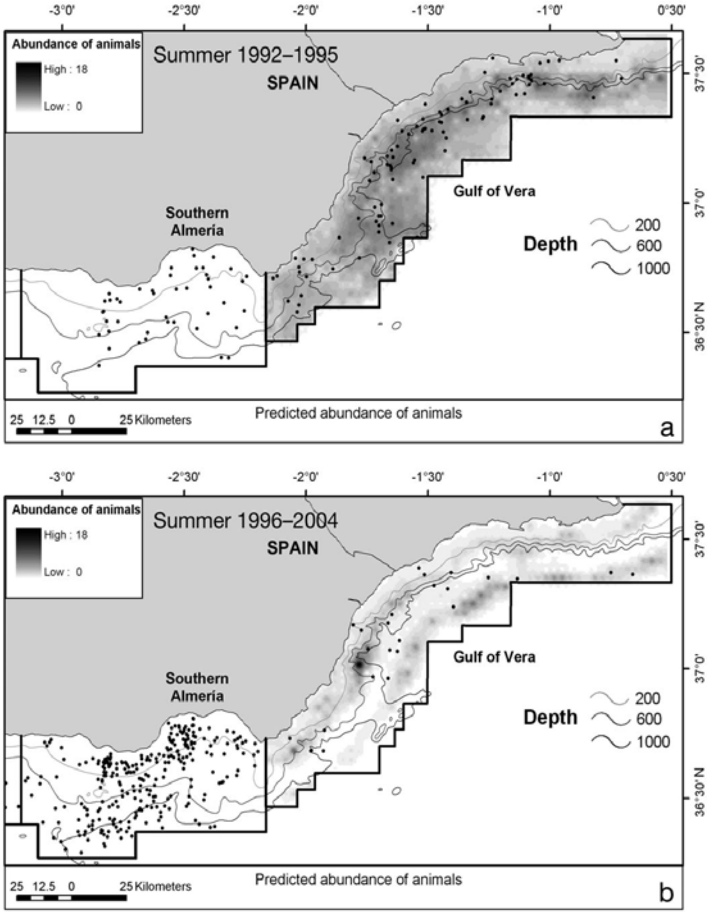


Figure 28. Surface maps of predicted abundance of *D. delphis* for the Gulf of Vera (a) between 1992 and 1995, and (b) between 1996 and 2004 (taken from Cañadas and Hammond, 2008).

Eastern Mediterranean Sea

Once common dolphins were relatively abundant in the area of Kalamos of western Greece, however numbers have declined dramatically over the past decade. A series of photo-identification mark-recapture abundance estimates calculated for common dolphins in Kalamos indicated a rapid decline from about 150 animals in 1996 to ca. 15 common dolphins in 2007 (Bearzi *et al*., 2008). During this study, five common dolphins with mutilations were found stranded near Paleros between December 2004 and April 2005. Of those, only one animal could be examined and based on mutilation of its tail flukes, it likely died in fishing gear. However, no other reports were received of common dolphin bycatch during the entire 15 year study (1993 to 2007) or during a 12-month fisheries monitoring programme (Bearzi *et al*., 2008).

The ACCOBAMS Scientific Committee (2008) and Bearzi *et al*. (2010) suggested that prey depletion could be a factor in the decline of the common dolphin in the waters of Kalamos (and is also suspected for the Gulf of Vera, Spain), which is a Natura 2000 site known as 'Inner Ionian Sea Archipelago' (or area GR2220003). It has been suggested that common dolphins may disappear from this region unless strict protection measures are enforced, such as the implementation of fishery management measures to reduce overfishing (Bearzi *et al.*, 2008; 2010).

7.1.3.2 Striped dolphins

Population structure

Analysis of skin samples collected from free range and stranded striped dolphins (*Stenella coeruleoalba*) in the Mediterranean Sea and Northeast Atlantic suggested very limited gene flow across the Straits of Gibraltar (Garcia-Martinez *et al.*, 1999; Gaspari *et al.*, 2007; Gaspari *et al.*, in prep.). Within the Mediterranean Sea there is evidence of population genetic structure based on genetic differentiation between the comparisons of putative populations and significant differences in tissue pollutant levels (Aguilar *et al.*, submitted). Separation of the eastern and western Mediterranean Sea basins occurs at the Italian Peninsula, where Europe is linked with Africa, and it has been suggested that this may have influenced the genetic separation of striped dolphins in this region (Gaspari *et al.*, 2007). Mitochondrial DNA analysis was undertaken using samples obtained from Gibraltar to Greece, and yielded 59 haplotypes (n=166). No haplotypes were shared between Gibraltar and Greece, thus supporting strong evidence of differentiation between the eastern and western Mediterranean Sea (Gaspari *et al.*, in prep.). Further, an earlier study reported small but significant differentiation between the Adriatic and Tyrrhenian Seas using 165 samples and eight microsatellite DNA loci, further suggesting low gene flow between basins (Gaspari *et al.*, 2007).

Distribution and abundance

Although the striped dolphin is the most abundant cetacean in the Mediterranean Sea, in both the eastern and western basins, it is not found at uniform densities. It typically shows a preference for highly productive, open waters beyond the continental shelf (Forcada *et al.*, 1994; Frantzis *et al.*, 2003; Gannier, 2005; Notarbartolo di Sciara *et al.*, 1993). The population in the western Mediterranean, excluding the Tyrrhenian Sea, was estimated in 1991 to be 117 880 (CI = 68 379–214 800) individuals (Forcada *et al.*, 1994). There are no estimates available for the eastern Mediterranean basin, or current estimates for the whole western Mediterranean basin.

Aguilar *et al.* (submitted) have proposed listing the striped dolphin subpopulation in the Mediterranean Sea as 'vulnerable' on the IUCN Red List. It is suspected that striped dolphins have reduction in population size of >50% during the last ca. 60 years based on (a) a > 2/3 reduction in mean school size in the 1990s (here considered an index of abundance), (b) a decline in quality of habitat, particularly food availability, (c) past and current high levels of exploitation –incidental catches-, and (d) due to the effects of pathogens and pollutants (Aguilar *et al.*, submitted). Large-scale morbillivirus outbreaks were reported between 1990–1992 and 2006–2007, producing many 1000s of deaths during the former period and ca. 200 during the latter period (Aguilar and Borrell, 1994; Aguilar *et al.*, submitted).

Pelagos Sanctuary

Recent aerial surveys were conducted in the Pelagos Sanctuary (northwestern Mediterranean, see Figure 27) to estimate winter (January, February) and summer (August) abundance of striped dolphins. During winter 2009, 114 sightings of striped dolphins were reported. Using the multiple covariate method, MCDS, the uncorrected striped dolphin population size was estimated to be 19 578 (95% CI=12 318–27 039), with a density of 0.2218 individuals km-1 (95% CI=0.1395–0.3063) (Panigada *et al.*, 2009). Whereas in August 2008, 13 232 (95% CI=6640.0–26 368) striped dolphins were estimated, with a density of 0.23 individuals km-1 (95% CI=0.11–0.45). The central value of the summer 2008 estimate was almost half of that of a survey conducted in 1992 in the same area with comparable effort and platform (N=25 614; %CV=25.3; 95% CI=15 377–42 658) (Lauriano *et al.*, 2009). These results raise concern that the abundance of striped dolphins in the Pelagos Sanctuary may be declining (Lauriano *et al.*, 2009).

Western Mediterranean Sea

The absolute abundance for striped dolphins in the northern Alborán Sea and Gulf of Vera (1992–2008) was 14 220 (CV=0.18, CI = 8827–17 764, density = 0.72 dolphins/sq km) (Höschle and Cañadas, unpublished data). Relative abundance was highest in the northwestern Alborán Sea. Highest densities were reported in deeper waters, with relatively low densities observed in shallow waters; where striped dolphins often formed mixed groups with common dolphins (Hoschle, 2008). No trend in abundance over years could be observed, although this work/analysis is still in process.

Summary

The distribution and abundance data available for common and striped dolphins cannot be used to evaluate the impact of the Regulation 812/2004 due to a lack of contemporary population abundance estimates for both species within the Mediterranean Sea.

As reiterated in successive meetings of the ACCOBAMS Scientific Committee it is imperative that baseline population estimates and distributional information is obtained for cetaceans in the Mediterranean Sea (ACCOBAMS, 2008). Further in 2009, and previous years, the IWC Sub-Committee on Small Cetacean recommended that a survey be carried out to obtain basin-wide estimates of abundance for cetaceans in the Mediterranean Sea. The SC subcommittee recommended that the planning and implementation of such a survey proceeded as quickly as possible.

Within the Mediterranean Sea there are problems posed in conserving both species due to large-scale incidental capture in some regions. These problems are well known and have been documented by the ACCOBAMS Scientific Committee since its inception. Within the Mediterranean Sea in recent years, highest bycatch rates have been reported in Moroccan driftnets, and it has been estimated that over a 12-month period (December 2002 and September 2003) the estimated bycatch rate by the whole driftnet fleet in the Alborán Sea targeting swordfish (*Xiphias gladius*) was 3110–4184 dolphins (including both the common and striped dolphin) (Hassani *et al.*, 1997, Tudela *et al.*, 2005).

7.1.4 References

ACCOBAMS. 2004. Final Report of the Workshop on obtaining baseline cetacean abundance information the ACCOBAMS area (Valsain 17–97 December) 31 pp.

ACCOBAMS. 2008. Chair's Report of the 2008 Workshop on surveying the ACCOBAMS Area (Monaco 15–17 May) 19 pp.

ACCOBAMS. 2010. Report of the Sixth Meeting of the Scientific Committee (Casablanca 11–13 January) 48 pp.

Aguilar, A., and Borrell, A. 1994. Abnormally high polychlorinated biphenyl levels in striped dolphins (*Stenella coeruleoalba*) affected by the 1990–1992 Mediterranean epizootic. Science of The Total Environment 154(2–3): 237–247.

Aguilar, A., Gaspari, S., Notarbartolo di Sciara, G., and Perrin, W.F. submitted. Striped dolphin, *Stenella coeruleoalba*, Mediterranean subpopulation. Checklist for IUCN Red List Assessment. Mediterranean and Black Sea cetacean assessment workshop.

Bearzi G., Agazzi S., Gonzalvo J., Bonizzoni S., Costa M., Petroselli A. 2010. Biomass removal by dolphins and fisheries in a Mediterranean Sea coastal area: do dolphins have an ecological impact on fisheries? Aquatic Conservation: Marine and Freshwater Ecosystems 11 p. DOI: 10.1002/aqc.1123.

Bearz, G., Agazzi, S., Gonzalvo, J., Costa, M., Bonizzoni, S., Politi, E., Piroddi, C., and Reeves, R.R. 2008. Overfishing and the disappearance of short-beaked common dolphins from western Greece. Endangered Species Research 5: 1–12.

Bearzi, G., Reeves, R.R., Notarbartolo di Sciara, G., Politi, E., Cañadas, A., Frantzis, A., and Mussi, B. 2003. Ecology, status and conservation of short-beaked common dolphins (*Delphinus delphis*) in the Mediterranean Sea. Mammal Review 33(3): 224–252.

Canadas, A., and Hammond, P.S. 2008. Abundance and habitat preferences of the short-beaked common dolphin *Delphinus delphis* in the southwestern Mediterranean: implications for conservation. Endangered Species Research 4(3): 309–331.

Forcada, J., Aguilar, A., Hammond, P.S., Pastor, X., and Aguilar, R. 1994. Distribution and numbers of striped dolphins in the western Mediterranean sea after the 1990 epizootic outbreak. Marine Mammal Science 10(2): 137–150.

Forcada, J., and Hammond, P. 1998. Geographical variation in abundance of striped and common dolphins of the western Mediterranean. Journal of Sea Research 39(3–4): 313–325.

Forcada, J., Notarbartolo di Sciara, G., and Fabbri, F. 1995. Abundance of fin whales and striped dolphins summering in the Corso-Ligurian Basin. Mammalia 59(1): 127–140.

Fortuna, C.M., Canese, S., Giusti, M., Revelli, E., Consoli, P., Florio, G., Greco, S., Romeo, T., Andaloro, F., Fossi, M.C., and Lauriano, G. 2007. An insight into the status of striped dolphins (*Stenella coeruleoalba*) of the southern-Tyrrhenian sea. Journal of the Marine Biological Association of the UK 87: 1321–1326.

Frantzis, A., Alexiadou, P., Paximadis, G., Politi, E., Gannier, A., and Corsini-Foka, M. 2003. Current knowledge of the cetacean fauna of the Greek Seas. Journal of Cetacean Research and Management 5(3): 219–232.

Gannier, A. 2005. Summer distribution and relative abundance of delphinids in the Mediterranean Sea. Rev. Ecol. (Terre Vie), 60: 223–238.

Garcia-Martinez, J., Moya, A., Raga, J.A., and Latorre, A. 1999. Genetic differentiation in the striped dolphin *Stenella coeruleoalba* from European waters according to mitochondrial DNA (mtDNA) restriction analysis. Molecular Ecology 8(6): 1069–1073.

Gaspari, S., Azzellino, A., Airoldi, S., and Hoelzel, A.R. 2007. Social kin association and genetic structuring of striped dolphins (*Stenella coeruleoalba*) population in the Mediterranean Sea. Molecular Ecology 16: 2922–2933.

Gaspari, S., Marsili, L., Fossi, C., Ciofi, C., and Chelazzi, G. in prep. Identification of conservation units of the striped dolphin *Stenella coeruleoalba* in the Mediterranean Sea by population genetic analysis.

Gomez de Segura, A., Crespo, E.A., Pedraza, S.N., Hammond, P.S., and Raga, J.A. 2006. Abundance of small cetaceans in the waters of the central Spanish Mediterranean. Marine Biology 150(1): 149–160.

Hassani, S., Antoine, L., and Ridoux, V. 1997. Diets of albacore, *Thunnus alalunga*, and dolphins, *Delphinus delphis* and *Stenella coeruleoalba*, caught in the Northeast Atlantic albacore driftnet fishery: a progress report. Journal of Northwest Atlantic Fishery Science 22: 119–124.

Hoschle, C. 2008. Abundance and habitat preferences of striped dolphins (*Stenella coeruleoalba*) in the northern Alborán Sea, Spain, University of Hamburg.

ICES SGBYC. 2009. Report of the Study Group for Bycatch of protected species (SGBYC), 19–22 January 2009 ICES, Copenhagen, Denmark.

IWC. 2004. Annex L. Report of the Sub-committee on Small Cetaceans. Journal of Cetacean Research and Management 6: 315–334.

Lauriano, G., Panigada, S., Canneri, R., and Zeichen, M.M. 2009. Abundance estimate of striped dolphins (*Stenella coeruleoalba*) in the Pelagos Sanctuary (NW Mediterranean) by means of line transect surveys. International Whaling Commission SC/61/SM25.

Natoli, A., Cañadas, A., Vaquero, C., Politi, E., Fernandez-Navarro, P., and Hoelzel, A. 2008. Conservation genetics of the short-beaked common dolphin (*Delphinus delphis*) in the Mediterranean Sea and in the eastern North Atlantic Ocean. Conservation Genetics 9(6): 1479–1487.

Northridge, S., and Fortuna, C. 2008. A Standardised Methodology for Use in the Collection of Data on Cetacean Bycatch and Depredation of Nets. Draft Protocol for data collection on bycatch and depredation in the ACCOBAMS Region. International workshop on bycatch within the ACCOBAMS region (Rome 17-18 September) 37 pp.

Notarbartolo di Sciara, G., Venturino, M.C., Zanardelli, M., Bearzi, G., Borsani, F.J., and Cavalloni, B. 1993. Cetaceans in the central Mediterranean Sea: distribution and sighting frequencies. Boll. Soc. Zool. Ital.(60): 131–138.

Panigada, S., Burt, L., Lauriano, G., Pierantonio, N., and Donovan, G. 2009. Winter abundance of striped dolphins (*Stenella coeruleoalba*) in the Pelagos Sanctuary (northwestern Mediterranean Sea) assessed through aerial survey. International Whaling Commission SC/61/SM7.

Tudela, S., Kai Kai, A., Maynou, F., El Andalossi, M., and Guglielmi, P. 2005. Driftnet fishing and biodiversity conservation: the case study of the large-scale Moroccan driftnet fleet operating in the Alborán Sea (SW Mediterranean). Biological Conservation 121(1): 65–78.

Table 13. Abundance estimates within the ACCOBAMS area (taken from Northridge and Fortuna (2008). All surveys were undertaken prior to 2004.

WESTERN MEDITERRANEAN	GFCM AREA CODE	STUDY AREA (KM²)	SAMPLED AREA	YEARS	N	CV	95% CI	ESTIMATION METHOD	SOURCE
Striped dolphins (*Stenella coeruleoalba*)									
Western Mediterranean (Tyrrhenian Sea excluded)	1 to 9, 11	889 400	in- & offshore	1991	117 880	0.22	68 379–214 800	Distance sampling	(Forcada *et al*., 1994)
Corso-Ligurian basin	8, 9, 11	58 269	in- & offshore	1992	25 614	0.25	15 377–42 658	Distance sampling	(Forcada *et al*., 1995)
Balearic Sea (1)	5, 6	64 733	in- & offshore	1991–1992	5826	0.36	2193–15 476	Distance sampling	(Forcada and Hammond, 1998)
Provencal basin (2)	6 to 8, 11	133 800	in- & offshore	1991–1992	30 774	0.36	17 433–54 323	Distance sampling	(Forcada and Hammond, 1998)
Ligurian Sea (3)	8, 9	46 677	in- & offshore	1991–1992	14 003	0.35	6305–31 101	Distance sampling	(Forcada and Hammond, 1998)
Liguro-Provencal basin (2+3)	6 to 9, 11	177 517	in- & offshore	1991–1992	42 604	0.26	24 962–72 716	Distance sampling	(Forcada and Hammond, 1998)
North-western Mediterranean (1+2+3)	5 to 9, 11	240 490	in- & offshore	1991–1992	48 098	0.24	29 388–78 721	Distance sampling	(Forcada and Hammond, 1998)
Alboran Sea (4)	1 to 4	88 640	in- & offshore	1991–1992	17 728	0.33	9507–33 059	Distance sampling	(Forcada and Hammond, 1998)
Central Spanish Mediterranean sea	6	32 270	in- & offshore	2001–2003	15 778	0.19	10 940–22 756	Distance sampling	(Gomez de Segura *et al*., 2006)
South Balearic area (5)	4 to 6, 11	235 125	in- & offshore	1991–1992	18 810	0.34	8825–35 940	Distance sampling	(Forcada and Hammond, 1998)
South-western Mediterranean (4+5)	1 to 6, 11	333 025	in- & offshore	1991–1992	39 963	0.38	18 206–87 721	Distance sampling	(Forcada and Hammond, 1998)

Aeolian Islands (Italy)	10	13 200	in- & offshore	2002–2003	4030	0.30	2239–7253	Distance sampling	(Fortuna *et al.*, 2007)
Common dolphin (*Delphinus delphis*)									
Alboran Sea	1 to 4	92 100	in- & offshore	1991–1992	14 736	0.40	6923–31 366	Distance sampling	(Forcada and Hammond, 1998)
TURKISH STRAIT SYSTEM	**GFCM AREA CODE**	**STUDY AREA (KM^2)**	**SAMPLED AREA**	**YEARS**	**N**	**CV**	**95% CI**	**ESTIMATION METHOD**	**SOURCE**
Common dolphin (*Delphinus delphis*)									
Turkish Strait	28	~100	inshore	1997	773	-	292–2059	Distance sampling	(IWC 2004)
Turkish Strait	28	~100	inshore	1998	994	-	390–2531	Distance sampling	(IWC 2004)

7.2 Harbour porpoise management units in the North-east Atlantic

7.2.1 Introduction

In 2009, the WGMME reviewed available literature on population structure in harbour porpoises within the North-east Atlantic. Harbour porpoises are not distributed throughout western European waters; low sighting rates have been reported for the eastern English Channel and also for the southern Bay of Biscay (Basque area). Further, harbour porpoises are predominately confined to shelf waters, though animals have been sighted in deep offshore waters between Faroe Islands and Iceland (Reid *et al.*, 2003). Fontaine (2008) reported that two separate populations exist in this region, the North-east Atlantic (France to Norway) and Iberian populations. Results suggested that north of the Bay of Biscay both genetic and ecological approaches converged toward a similar conclusion: harbour porpoises form a continuous system under isolation by distance (i.e. the greater the distance the smaller the genetic correlation) displaying regional habitat-related variation in genetic, demographic and ecologic properties. The Iberian population, which displayed a very small population size, was further qualified as an independent demographically significant unit, although belonging to the Atlantic evolutionary significant unit. Based on these results, the WGMME recommended that the Iberian harbour porpoise population and the North-east Atlantic harbour porpoise continuous system population (France to Norway) are managed separately (ICES WGMME, 2009).

One of the main problems identified in this assessment was how to categorize continuous process for management and conservation purposes. The dilemma in defining management units in a continuous system will essentially lead to problems in deciding boundaries for Management Units (MUs). Especially in a species that has shown evidence of large-scale changes in distribution, i.e. the southern shift in distribution from the northern North Sea to the southern North Sea between the 1990s and 2000s (see Section 7.1).

Prior this assessment, a number of studies previously proposed the existence of separate stocks within the North Atlantic (Andersen 2003; Gaskin 1984; IWC 1996; 2000; Lockyer, 2003). Gaskin (1984) proposed 14 stocks for porpoises in the North Atlantic and later the IWC (1996) revised this to 13 (with one more in the Black Sea), lumping together as one unit, the English Channel, NW French, Spanish and Portuguese waters, including the Bay of Biscay (see Figure 29). In 1999, the joint IWC-ASCOBANS Working Group recognized extra subdivisions, and proposed five stocks within the ASCOBANS area; (1) Baltic Sea, (2) Kattegat, inner Danish waters and German Baltic Sea, (3) northern North Sea, (4) central and southern North Sea and the (5) Celtic Shelf (IWC 2000).

7.2.2 Management Units

The ASCOBANS-HELCOM small cetacean population structure Working Group reviewed all available literature, and assessed unpublished data, in order to decipher contemporary existing stock structure with the North-east Atlantic (see (Evans and Teilmann, 2009). The WG reviewed data from studies on genetics (mtDNA and microsatellites), skull morphometrics, fatty acids and diet, stable isotopes, parasite loads, contaminant loads and telemetry. The WG only recommended minor changes to earlier divisions, and these mainly in the light of recent more comprehensive genetic studies (such as (Fontaine, 2008; Fontaine *et al.*, 2007) and the combining of information from other approaches (e.g. Danish telemetry studies) so as to derive Management Units that were not so heavily based upon genetics (Evans *et al.*, 2009).

This therefore enabled the identification of "ecological stocks" i.e. a group of individuals of the same species that co-occur in space and time and have an equal opportunity to interact with each other, and not just purely base MUs on separate evolutionary differences i.e. "populations;" a group of individuals of the same species living in close enough proximity that any member of the group can potentially mate with any other member.

The recommended MUs proposed by the ASCOBANS-HELCOM WG include (see Figure 30):

1) **WGR** = West Greenland;
2) **ICE** = Iceland;
3) **FAR** = Faroe Islands;
4) **NOR** = Northwest/Centralwest Norway and the Barents Sea;
5) **NENS** = North-eastern North Sea and Skagerrak;
6) **SWNS** = Southwestern North Sea and the Eastern Channel;
7) **IDW** = Inner Danish Waters;
8) **BAL** = Baltic Sea proper;
9) **CES** = Celtic Sea (plus South-west Ireland, Irish Sea and the Western Channel);
10) **NWIS** = North-west Ireland and West Scotland;
11) **BoB** = Bay of Biscay (West France);
12) **IBNA** (NW Spain, Portugal and NW Africa).

The main changes from earlier stock divisions include: dividing the North Sea into two MUs along a median (at this stage arbitrary) line, running NNW–SSE; inclusion of the Shetland Islands, Skagerrak and northern Kattegat within the North-eastern North Sea MU; northern boundary shift of the North-eastern North Sea MU along the Norwegian coast; Inner Danish Waters MU to include part of the Kattegat, all of the Danish Belt seas, and the Western Baltic; the Baltic Sea proper to form a separate MU, with its western boundary being around the Darss/Gedser underwater ridge or Rügen; the coasts of Portugal and NW Spain forming a separate MU (which was placed tentatively with NW Africa), from that of the Bay of Biscay and English Channel.

Results from very recent genetic analysis undertaken by Fontaine *et al.*, 2007 suggests further separating the IBNA MU, which incorporates northwest Spain, Portugal and northwest Africa, into two separate MUs; (a) Iberian and (b) northwest Africa MU's (Fontaine, pers. comm.).

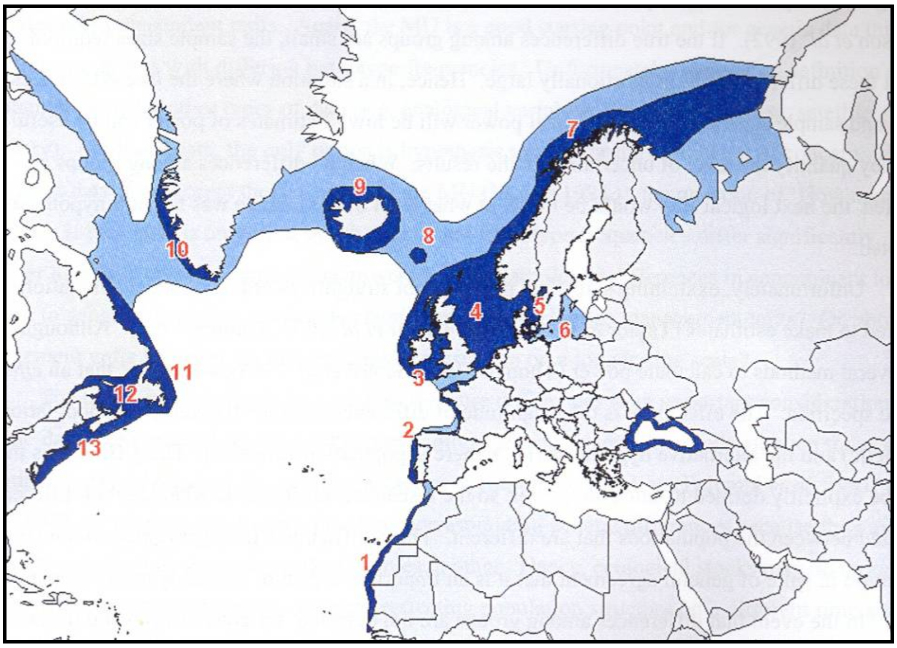


Figure 29. The IWC designated 13 stocks within the North Atlantic including: (1) Northwest Africa, (2) Iberia/Bay of Biscay, (3) Ireland/Western UK, (4) North Sea, (5) Kattegat and IDW, (6) Baltic Sea, (7) North Norway/Barents Sea, (8) Faroe Islands and (9) Iceland.

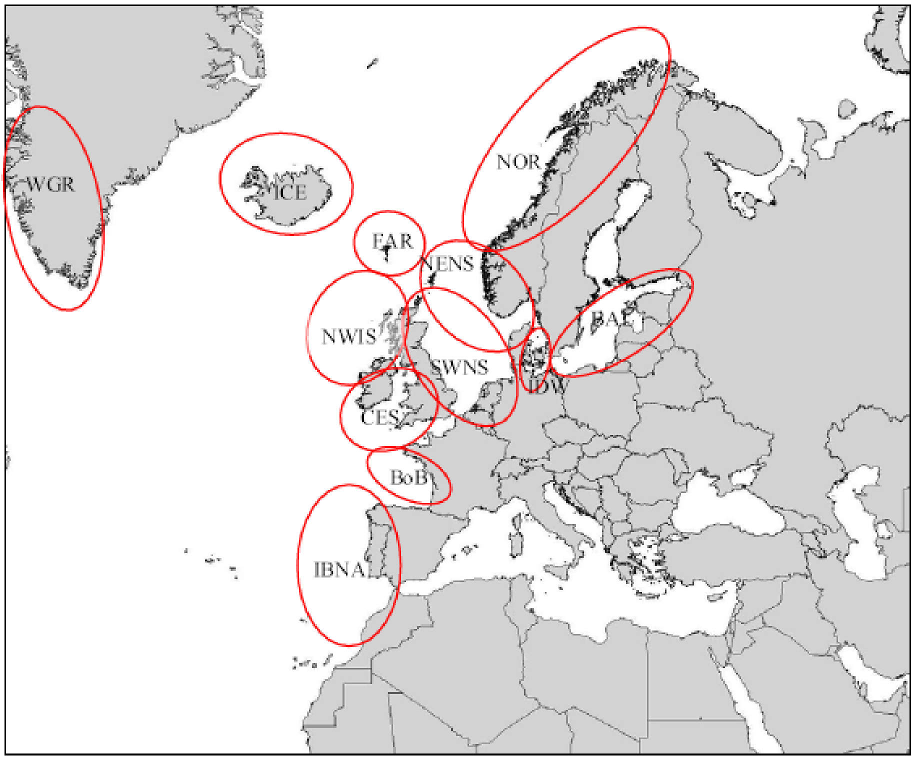


Figure 30. Recommended Management Units for harbour porpoise proposed by the ASCOBAN-HELCOM small cetacean population structure workshop (taken from Evans and Teilmann, 2009).

Within the North-east Atlantic (France to Norway) harbour porpoise continuous system population, seven separate Management Units have been defined. Although a number of studies justified the existence of these separations, these are not closed systems and movements of animals between Management Units will occur. The level at which is currently unknown. It should be noted that for the Bay of Biscay stock, very low sighting rates have also been reported. Within SCANS-II, no animals were observed, or detected acoustically in summertime in the Bay of Biscay, west of France (see Section 7.1). However, this is not reflective of the actual distribution of the species, as harbour porpoises have been reported as bycatch in fisheries operating off the coast (see Section 7.1).

A recent genetic study also suggested splitting the Skagerrak and the Belt Sea (inner Danish waters) with a transition zone at the Kattegat (Wiemann *et al.*, 2010), thus backing up recent telemetry studies and designating the inner Danish waters as a separate MU. Very few porpoises remain in the Baltic proper.

7.2.3 Recommendations

1) The WGMME reiterates its recommendation from last year and **strongly recommends** that the Iberian harbour porpoise population should be given a high priority for conservation, as a consequence of its presumed small population size, low genetic diversity and likely susceptibility to habitat degradation.
2) The WGMME again **strongly recommends** immediate action by the Spanish and Portuguese governments in monitoring and conserving the Iberian harbour porpoise population.
3) Based on the newly described harbour porpoise Management Units, the WGMME **recommends** to ASCOBANS the establishment of a separate conservation plan for the harbour porpoise Inner Danish Waters MU.
4) The WGMME also **recommends** to ASCOBANS to take into account the existence of the two newly designated harbour porpoise Management Units in the North Sea, North-eastern North Sea and Skagerrak and Southwestern North Sea and Eastern Channel, within their harbour porpoise North Sea conservation plan; with the inclusion of the Shetland Islands, Skagerrak and northern Kattegat within the North-eastern North Sea MU.

7.2.4 References

Andersen, W.L. 2003. Harbour porpoise (*Phocoena phocoena*) in the North Atlantic: distribution and genetic population structure. *In* Harbour Porpoises in the North Atlantic. NAMMCO Scientific publications, Volumn 5. *Edited by* T. Haug, G. Desportes, G. Vikingsson and L. Witting.

Evans, P.G.H., and Teilmann, J. 2009. Report of ASCOBANS/HELCOM small cetacean population structure workshop, ASCOBANS, Bonn, Germany.

Fontaine, M.C. 2008. Structure et évolution des populations de marsouins communs des eaux Paléarctiques occidentales (Structure and evolution of harbour porpoise populations in Western Palearctic waters) Université de Liège, Belgium.

Fontaine, M.C., Baird, S.J.E., Piry, S., Ray, N., Tolley, K.A., Duke, S., Birkun Jr, A., Ferreira, M., Jauniaux, T., Llavona, A., Ozturk, B., Ozturk, A.A., Ridoux, V., Rogan, E., Sequeira, M., Siebert, U., Vikingsson, G.A., Bouquegneau, J.-M., and Michaux, J.R. 2007. Rise of oceanographic barriers in continuous populations of a cetacean: the genetic structure of harbour porpoises in Old World waters BMC Biology 5(30).

Gaskin, D.E. 1984. The harbour porpoises (*Phocoena phocoena* L.): regional populations, status, and information on direct and indirect catches. Reports of the International Whaling Commission 34: 569–586.

ICES WGMME. 2009. Report of the ICES Working Group on Marine Mammal Ecology. February 2–6 2009, Vigo, Spain.

IWC. 1996. Report of the Sub-Committee on Small Cetaceans. . Reports of the International Whaling Commission 46: 160–179.

IWC. 2000. Report of the Scientific Committee. Annex O. Report of the IWC-ASCOBANS Working Group on Harbour porpoises. Journal of Cetacean Research and Management 2 (Suppl.): 297–305.

Lockyer, C. 2003. A review of methods for defining population structure in the harbour porpoise (*Phocoena phocoena*). *In* Harbour Porpoises in the North Atlantic. NAMMCO Scientific publications, Volumn 5. *Edited by* T. Haug, G. Desportes, G. Vikingsson and L. Witting. pp. 41–69.

Reid, J.B., Evans, P.G.H., and Northridge, S.P. 2003. Atlas of cetaceans distribution in northwest European waters. Joint Nature Conservation Committee (JNCC), Peterborough.

Wiemann, A., Andersen, L., Berggren, P., Siebert, U., Benke, H., Teilmann, J., Lockyer, C., Pawliczka, I., Skóra, K., Roos, A., Lyrholm, T., Paulus, K., Ketmaier, V., and Tiedemann, R. 2010. Mitochondrial Control Region and microsatellite analyses on harbour porpoise (*Phocoena phocoena*) unravel population differentiation in the Baltic Sea and adjacent waters. Conservation Genetics 11(1): 195–211.

7.3 Update on Cetacean Offshore Distribution and Abundance in the European Atlantic (CODA)

In 2009, the WGMME presented the abundance estimates produced by the CODA survey undertaken in European offshore waters during July 2007 (see Section 7). As part of this study, surface maps of smoothed predicted abundance e were produced, and these are presented in Figures 31 and 32 (Taken from CODA 2009).

Density surface modelling provided information on the spatial distribution of abundance and habitat use. **Common and striped dolphins** displayed a similar distribution, with higher densities predicted to occur in the southern part of the surveyed area (Bay of Biscay), and associated with the shelf break (Figure 31a, b, c). Highest densities of **long-finned pilot whales** were predicted to occur in the northwestern part of the surveyed area; associated with deep waters, seabed slopes with a southeast orientation and warmer temperatures (Figure 31d). **Sperm whale** predicted density was highest in northwestern waters of the Iberian Peninsula, the inner part of the Bay of Biscay, and off the northwest coast of the Hebrides (Figure 32a). Two main areas of distribution were predicted for **beaked whales** in the surveyed area: the inner part of the Bay of Biscay in association with the deep underwater canyons; and in the northwestern part of the surveyed area, west of the Hebrides, Scotland (Figure 32b). Higher densities of **fin whales** (and large whales) were predicted in the southern part of the surveyed area, in areas of sea surface temperature ranging from 16–19˚C, and depths between 1000–3000 m (Figure 32c).

7.3.1 References

CODA. 2009. Cetacean Offshore Distribution and Abundance in the European Atlantic (CODA).

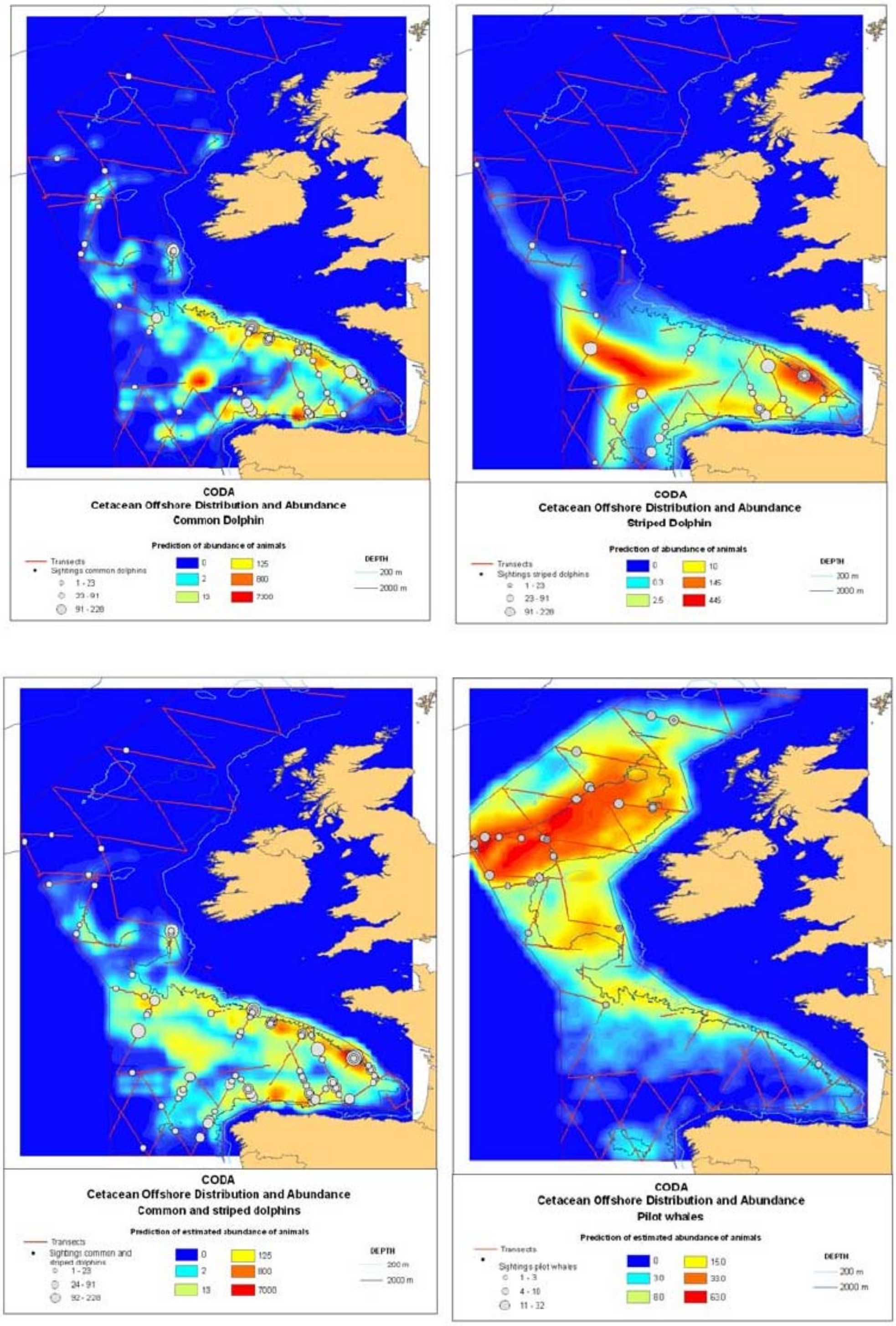


Figure 31. Surface maps of smoothed predicted abundance for (a) common dolphins, (b) striped dolphins, (c) common and striped dolphins and (d) pilot whales in offshore waters in July 2007 (taken from CODA, 2009).

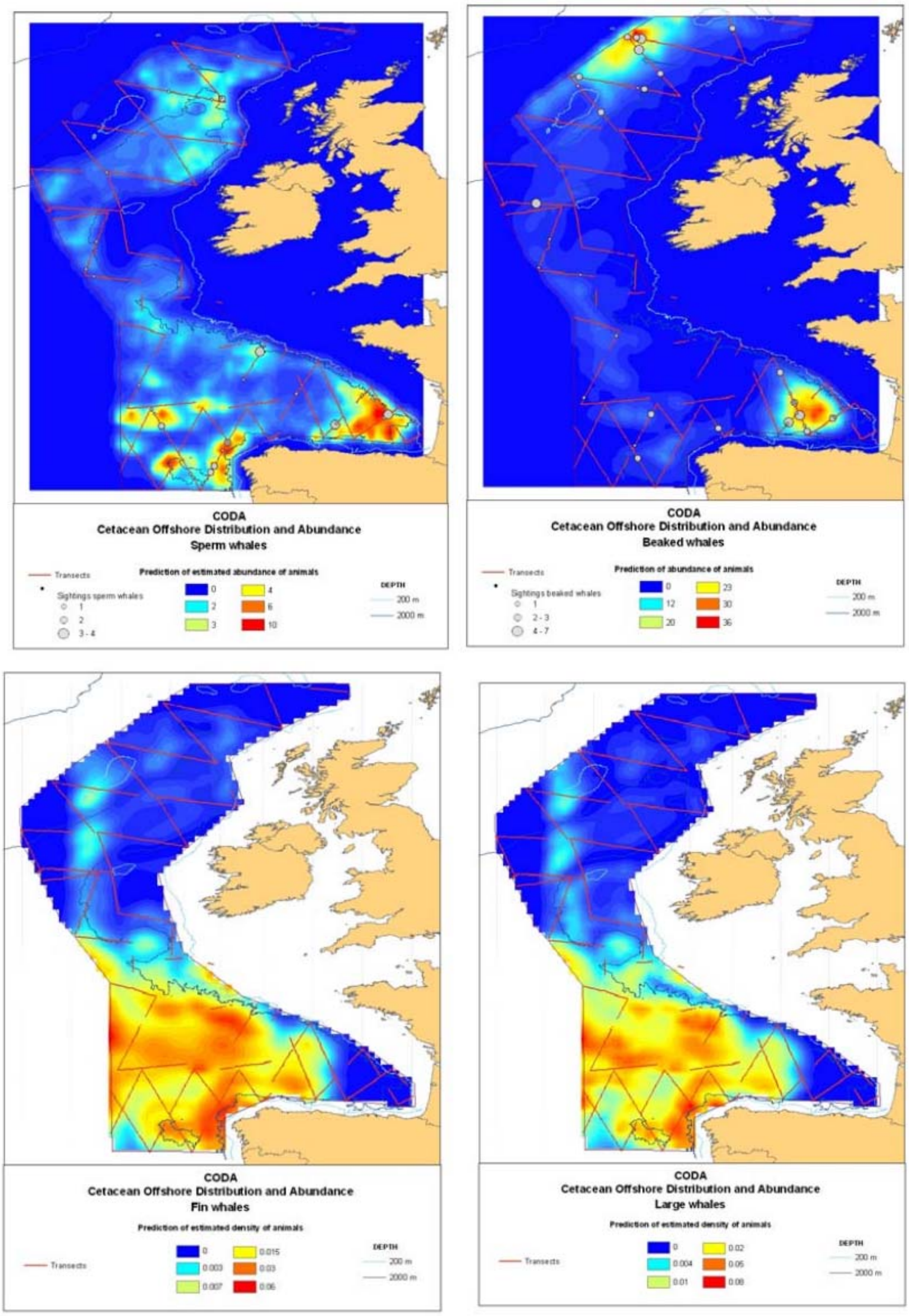


Figure 32. Surface maps of smoothed predicted abundance for (a) sperm whales, (b) beaked whales, (c) fin whales, and (d) large whales in offshore waters in July 2007 (taken from CODA, 2009).

7.4 Distribution and preliminary abundance estimates for cetaceans seen during Canada's marine megafauna survey; a component of the 2007 TNASS

7.4.1 Introduction

The Canadian Department of Fisheries and Oceans (DFO) conducted a large-scale aerial survey of marine megafauna in the Northwest Atlantic during summer 2007, as a component to the multinational Trans North Atlantic Sightings Survey (TNASS). The Canadian survey was flown following a systematic line-transect design and covered the Labrador Shelf and Grand Banks, the Estuary and Gulf of St Lawrence and the Scotian Shelf. Coverage was particularly high for the last two regions, with transects spaced 10 nautical miles apart (see Figure 33).

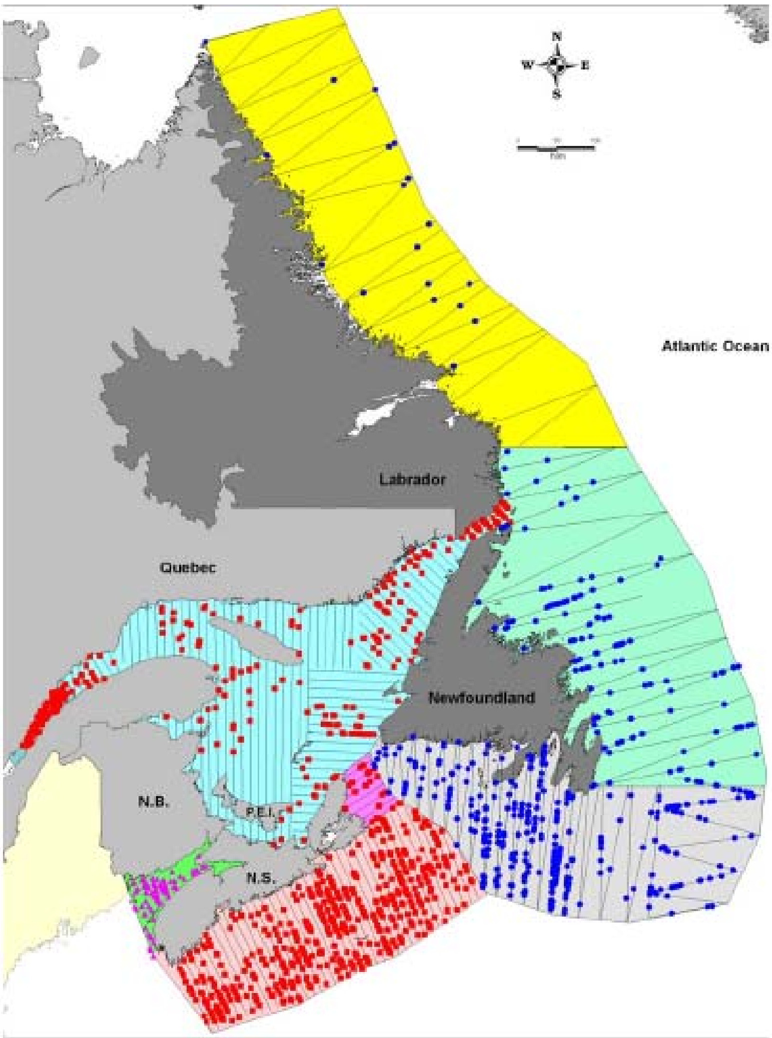


Figure 33. Aerial survey effort. Newfoundland and Labrador (yellow, light green, and light grey), Cape Breton (purple), Scotian Shelf (pink), and Gulf (light blue) blocks. Marine megafauna sightings made during the Canadian aerial survey effort are indicated with blue circles (Newfoundland and Labrador) and red squares (Cape Breton, Scotian Shelf, and Gulf). The darker green survey block in the Bay of Fundy was flown by NOAA; sightings collected during the NOAA survey are indicated with purple triangles. Taken from Lawson and Gosselin (2009).

7.4.2 Results

In all 1801 sightings of 11 494 individual cetaceans from twenty species were made during this survey. Within this region Lawson and Gosselin (2009) estimated population indices for species with >20 sightings. The most common species were dolphins, with an estimated 53 625 (CI: 35 179–81 773) common dolphins, 5796 (CI: 2, 681–13 088) Atlantic white-sided dolphin, 4862 (CI: 2204–8801) harbour porpoises, and 34 462 (95% CI: 20 560–57 862) unknown dolphin species. Long-finned pilot whales were the most abundant medium-sized species (6134; 95% CI: 2774–10 573), while there were an estimated 3242 minke whales (95% CI: 2051–4845), 2080 humpback (95% CI: 1337–3172), and 1352 fin whales (95% CI: 821–2226). These abundance estimates are uncorrected for perception and availability biases (Lawson and Gosselin, 2009).

Lawson and Gosselin (2009) reported that the Scotian Shelf had a higher diversity (27 species) and higher encounter rate (0.13 sighting/km) of marine mammals than the Gulf of St Lawrence (16 species, 0.04 sightings/km), northern or southern Newfoundland (0.0005 and 0.002 sighting/km), and the Labrador Shelf (0.0001 sighting/km). The St Lawrence Estuary is the only stratum within the Gulf that provided a higher encounter rate value (0.14 sighting/km) than the Scotian Shelf, primarily due to the presence of beluga whales (0.13 sighting/km). Abundance was lower for a number of species than expected, particularly common dolphin, harbour porpoises, fin whales, minke whales and white-sided dolphins. Anecdotal reports from fisheries officers, fishermen, and tour boat operators suggest that marine mammals in 2007 arrived later than usual in the study area. This delay would explain why the number of sightings in the Labrador stratum and the Gulf of St Lawrence, which were surveyed first, was lower than observed during previous surveys (Kingsley and Reeves, 1998).

Data from these surveys will have a significant impact on our understanding of cetaceans in the Northwest Atlantic, some which are listed under the Canadian Species At Risk Act. However, first the estimates will have to be adjusted by incorporating sightings made by the US team in the Bay of Fundy, and by correcting estimates for perception and availability biases. Distribution data will also be analysed to determine habitat use and preferences of the various species.

7.4.3 References

Lawson, J.W., and Gosselin, J.-F. 2009. Distribution and Preliminary Abundance Estimates for Cetaceans Seen During Canada's Marine Megafauna Survey - A Component of the 2007 TNASS. Canadian Science Advisory Secretariat. Research Document 2009/031. 34 pp.

Kingsley, M.C.S., and R. R. Reeves. 1998. Aerial surveys of cetaceans in the Gulf of St Lawrence in 1995 and 1996. Canadian Journal of Zoology 766: 1529–1550.

7.5 Present status of the Saimaa ringed seal population

The WGMME last reported on the status of the Ladoga and Saimaa ringed seals in 2008. Since then, no new information is available on the Ladoga ringed population in Lake Ladoga in Russia.

In Lake Saimaa in Finland only 44 pups were born in 2009, which is exceptionally low compared with previous years. Six of those pups were either still-born or died soon after birth (during the lactation period). In the years 2000 to 2005, the mean number of pups alive after the lactation period was ca. 50/year. Whereas the corresponding figure for the period 2006 to 2009 was only ca. 40 pups/year (see Figure 34). The high mortality rate observed for newborns in 2006 and 2007 (27% and 31%, respectively) was attributed to a lack of suitable snowdrifts on the shorelines for ringed seal to dig the lairs (Sipilä and Kokkonen, 2008; WGMME, 2008).

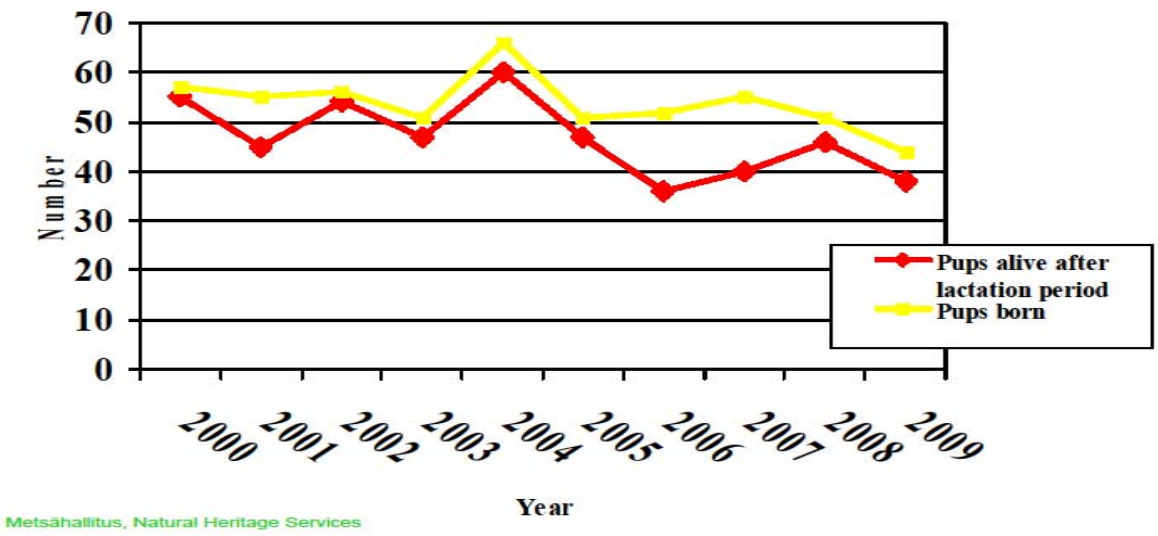


Figure 34. Number of pups born in 2000–2009 and number of pups alive at weaning.

Between 1990 and 2004 the population size was slowly increasing (growth rate 1.026) (Sipilä and Kokkonen, 2005) and the (highest) estimate of population size was ca. 280 seal in 2005 (Sipilä and Kokkonen, 2008). The present population size is ca. 260 seals, which suggest the population is in decline. The main threat to Saimaa ringed seal is incidental capture in fishing gear. After the weaning period in May and June, young seal easily get entangled in fishing gears (Figure 35). The combination of mortality due to entanglement in fishing nets with an abnormally small number of living pups found on Lake Saimaa may be fatal to the Saimaa seal population (WGMME, 2008).

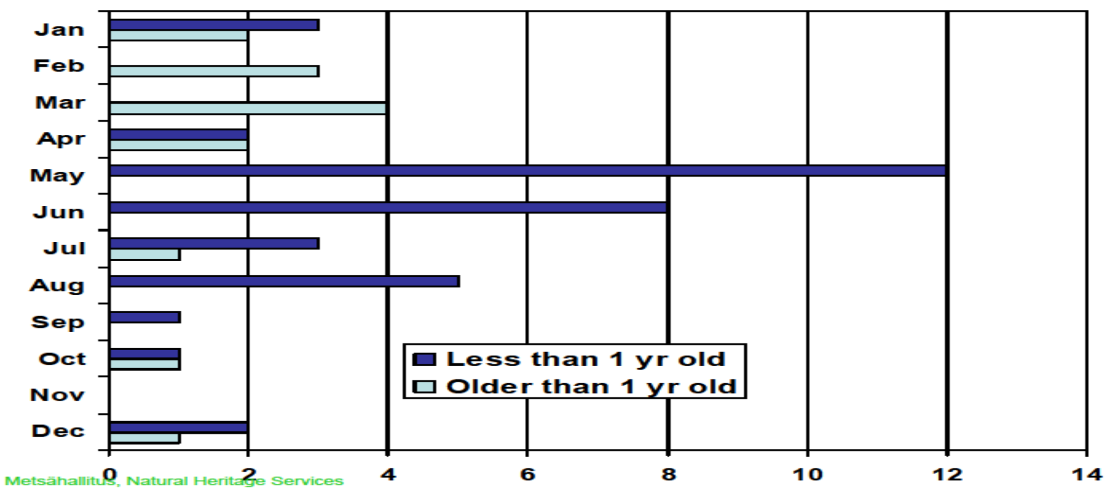


Figure 35. Monthly distribution of bycatch of seals in Lake Saimaa 2000–2009. Carcasses were retrieved and post-mortem examinations revealed that the cause of death was incidental capture.

According to telemetry studies the mean area used by young Saimaa seal in May and June is ca. 90 km^2, whereas the total area used by pups is ca. 1900 km^2 (Figure 36, area 15.4–30.6) (Anon., 2010). To decrease the mortality in the Saimaa ringed seal population, especially of weaned pups, there is an aim to restrict net fishing by fishermen in core areas. In 2010 net fishing is banned in ca. 1500 km^2 of the area from mid April to end of June.

7.5.1 Recommendation

The WGMME agrees with the actions of the Finnish Government, and recommends a ban on fishing within the area 15.4–30.6 in Lake Saimaa from mid April to mid June.

7.5.2 References

Anon. 2010: Saimaannorppa ja kalastus – seurantaryhmä II. – Maa- ja Metsätalousministeriön työryhmämuistio. MMM 2010:2, 25 s. + liit. (In Finnish) http://www.mmm.fi/attachments/kalariistajaporot/5nDNjY0Tr/NORPPARAPORTTI_100201.pdf.

Sipilä, T., Koskela, J.T. and Kokkonen, T.S. 2005. Spatial differences in the changes of population size of the Saimaa ringed seal. - In: Symposium on Biology and Management of Seals in the Baltic Area, E. Helle, O. Stenman and M. Wikman (eds,) . Kala- ja riistaraportteja 346:38–40.

Sipilä, T. and Kokkonen, T. 2008: Saimaannorppakannan tila vuonna 2007. Ilmaston muutoksen vaikutus sekä sen aiheuttaman haitan kompensoinnista. Metsähallitus, Etelä-Suomen Luontopalvelut, julkaisematon asiakirja nro 657/41/2008.17 s.

ICES WGMME. 2008. ICES. Report of the Working group on marine mammal ecology. February 25–29 2008, St Andrews, UK.

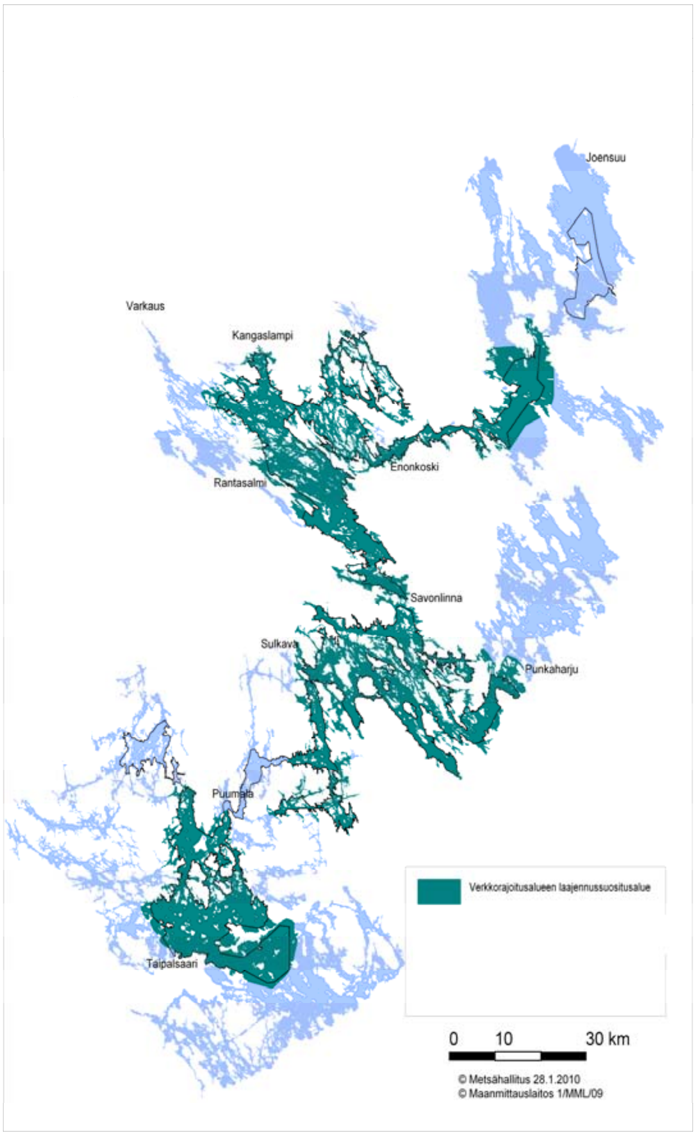


Figure 36. Recommendation of net fishing restriction area 15.4–30.6 in Lake Saimaa (highlighted in green).

7.6 Grey seal prey consumption

UK North Sea

Hammond and Grellier (2006) reported on grey seal diets and provided estimates of grey seal prey consumption in the North Sea based on scat samples collected in 1985 and 2002. They estimated that in 2002, grey seals consumed a total of 116 000 tonnes of commercially important fish species. Their equivalent figure for 1985 was 39 000 tonnes. These values were produced by scaling proportions of the energy represented by different prey species found in scats by the estimated energy consumption of the population of grey seals associated with the UK North Sea breeding colonies (27 000 seals in 1985; 67 000 seals in 2002).

Sandeel was the main prey species for North Sea grey seals. Consumption increased from 29 000 tonnes in 1985 to 69 000 tonnes in 2002, representing approximately 2.7% of the estimated stock size. In 1985 grey seals were estimated to have consumed 4150 tonnes (95% CI: 2484–5760) of cod. This figure rose to 8344 tonnes (95% CI: 5000–15 000) in 2002, 3.7% of the estimated stock size. Between 2002 and 2007, the total estimated annual removal of cod by the North Sea fisheries (landings and discards) was much larger, varying between 52 000 tonnes and 81 700 tonnes. This information on grey seal cod consumption has been considered unlikely to substantially change estimates of North Sea cod stock dynamics (ICES Advice, 2008).

West coast of Scotland and Shetland

Grey seal prey consumption estimates off western Scotland and Shetland were provided by Hammond and Harris (2006), based on scat samples collected in 1985 and 2002. They detected significant differences in grey seal diet in different areas, but found limited evidence of changes in diet between 1985 and 2002. The main changes between the two surveys were a relative decrease in sandeel consumption and an increase in herring consumption. Grey seal abundance increased from around 29 000 to 42 000 seals over this period, along with the proportion of prey stock size estimates they consumed. For most prey species the changes in consumption by grey seals were much smaller than the declines in both the total stock sizes and the landings by fisheries. In a modelling study, Pope and Holmes (2008) reported that the contribution of grey seal predation to total cod mortality is likely to be significant in the ICES Division VIa (west of Scotland) and may impair the ability of the cod stock to recover; though they emphasized that the limited data available makes including grey seal predation in the cod assessment problematic and their conclusions tentative.

A study of the potential impact of harbour seals on a small cod population in the Skagerrak/Kattegat area (Hansen and Harding, 2006) found that the impact of harbour seal predation on the cod population was negligible compared with human harvesting, and concluded that the predation pressure from those 14 000 animals was too small to affect the growth rate of that cod population. A new diet study has recently been started that covers all the Scottish populations of grey and harbour seals (http://www.sealdietscotland.co.uk). Work is also underway to refine the grey seal population models (SCOS, 2008). Once completed, these should hopefully improve our understanding of the current impact of grey seal predation on cod recruitment. Overall, it seems unlikely that cod is a fixed proportion in the seal diet, and that it is more likely that the proportion of cod in the diet would reduce as cod abundance declines.

ICES Advice (2009) evaluated impacts of the environment on cod stocks and reported on the general warming trend of the Northern Shelf waters. A negative impact on recruitment with rising sea temperature has been shown for cod in the warmer waters of this species' range, including cod off the west coast of Scotland (Brunel and Boucher, 2007). The balance between these effects, and the other environmental changes, including the effects of intraspecific predation, that are occurring is uncertain and makes the importance of grey seals to the determination of future fish stocks difficult to predict.

7.6.1 References

Brunel, T., and J. Boucher, 2007. Long-term trends in fish recruitment in the north-east Atlantic related to climate change. Fisheries Oceanography, 16(4): 336–349.

Hammond, P. S., and K. Grellier. 2006. Grey seal diet composition and prey consumption in the North Sea. Final report to Department for Environment Food and Rural Affairs on project MF0319.

Hammond, P. S., and R. N. Harris. 2006. Grey seal diet composition and prey consumption off western Scotland and Shetland. Final report to Scottish Executive Environment and Rural Affairs Department and Scottish Natural Heritage.

Hansen, B. J. L., and K. C. Harding. 2006. On the potential impact of harbour seal predation on the cod population in the eastern North Sea. *Journal of Sea Research*, **56** (4): 329–337.

ICES Advice. 2008. Cod in Subarea IV (North Sea), Division VIId (Eastern Channel), and Division IIIa (Skagerrak). ICES Advice 2008, *Book 6*.

ICES Advice. 2009. Cod in Division VIa (West of Scotland). *ICES Advice 2009, Book 5.*

Pope, J. G., and Holmes, S. J. 2008. Length-based Approaches compared to Age-based Approaches to Determining the Significance of Grey Seal Feeding on Cod in ICES Division VIa. ICES CM 2008/F:08.

SCOS. 2008. Scientific Advice on Matters Related to the Management of Seal Populations: 2008. Sea Mammal Research Unit. 98pp. http://www.smru.st-andrews.ac.uk/documents/SCOS_2008_v1.pdf.

7.7 Target population sizes for marine mammals

Scientific research into marine mammal ecology provides information that guides the management and conservation of these species. Local and total abundance estimates are an important part of this process and particularly relevant to assessing the potential and actual effects of developments such as offshore wind farms and other marine renewable energy devices. However, management targets and conservation goals are also required, and it is not obvious how these should best be set. At a very minimum, these targets need to be sufficient to limit extinction risks but generally they are intended to go beyond that point.

Currently, under the EU Habitats Directive, these targets are set by Member States (DocHab-04-03-03:http://www.lcie.org/Docs/Legislation/DocHab-04-03-03 rev3.pdf), and appears to be implemented differently in different countries. Potential Biological Removal, a method incorporated in the US Marine Mammal Protection Act (Wade, 1998), is being adopted in many countries. It was developed primarily to assess the sustainability of fisheries bycatch, but is now beginning to be used in other situations. PBR is simple to calculate, and generally aims to keep a population above the level at which maximum sustainable yield occurs. ASCOBANS adopted a modified form of PBR as a way to aim for their interim goal of harbour porpoises reaching 80% of their carrying capacity in the Baltic (Berggren *et al*., 2002). The IWC uses a different algorithm in its Revised Management Procedure, but also aims to maintain cetacean populations above a fixed proportion (72%) of their carrying capacity (IWC, 1999). These methods therefore assume that a fixed proportion of carrying capacity is an appropriate management goal for these species. The situation is further complicated for methods, such as PBR, where carrying capacities are not explicitly calculated, or necessarily calculable, from the available data. HELCOM has attempted a more flexible solution to this issue for Baltic seal populations as it attempts to eventually return them to their "natural" carrying capacities (HELCOM Recommendation 27–28/2 http://www.helcom.fi/Recommendations/en_GB/rec27–28_2). They use two thresholds for deciding applications to shoot seals around fishing gear: no shooting is permitted from populations below a lower threshold, and strong evidence is required that the population will continue to increase before permission is granted to shoot animals from a population between this and the maximum sustainable yield level, while weaker evidence is sufficient above this higher level. A broadly similar philosophy, of increased precaution for more depleted populations, is also being applied to management of commercial seal catches in Canada (Hammill and Stenson, 2007).

There are two issues with most of these approaches to setting management targets: it is not clear what fixed general proportion of the carrying capacity is most suitable, or even if such a general target exists, and the choice of target is often treated as a scientific decision. In practice the targets are often implicit and automated, and are assumed as apparently objective values, when they should be considered as societal choices and follow from the balancing of competing interests. These interests need to include, but not be limited to, conservation goals.

7.7.1 Recommendation

The WGMME **recommends** ICES to encourage a move away from implicit and automated conservation targets and towards the explicit definition and justification of target population sizes and management objectives.

7.7.2 References

Berggren, P., P.R. Wade, J. Carlström, and A.J. Read. 2002. Potential limits to anthropogenic mortality for harbour porpoises in the Baltic region. Biological Conservation 103:313–322.

Hammill, M. O., and Stenson, G. B. 2007. Application of the precautionary approach and conservation reference points to management of Atlantic seals. ICES Journal of Marine Science 64: 702–706.

International Whaling Commission. 1999. Report of the Scientific Committee. Annex N. The Revised Management Procedure (RMP) for Baleen Whales. J. Cetacean Res. Manage. (Suppl.) 1:251–258.

Wade, P.R. 1998. Calculating limits to the allowable human caused mortality of cetaceans and pinnipeds. Marine Mammal Science 14:1–37.

8 ToR e. Provide information on abundance, distribution, population structure and incidental capture of marine mammals off the Azores

8.1 Cetaceans

8.1.1 Species distribution, abundance and migration

The following overview of cetacean research in the Archipelago of the Azores was extracted from peer reviewed publications and unpublished data provided by the Department of Oceanography and Fisheries of the University of the Azores (DOP/UAç) to WGMME.

Prior to recent studies little was known regarding the spatial and temporal patterns of distribution and abundance of cetaceans around the Azores Archipelago (Figure 37) (Silva *et al*., 2003). The exception was the sperm whale (*Physeter macrocephalus*), due to studies associated with commercial whaling (Clarke, 1956; Ávila de Melo and Martin, 1985). Strandings and opportunistic sighting records were the principal sources of information for other species (Clarke, 1981; Martin, 1988; Gordon *et al*., 1990; Gonçalves *et al*., 1992; Reiner *et al*., 1993; Steiner, 1995; Gonçalves *et al*., 1996).

Twenty-eight cetacean species have been documented in Azorean waters (Table 14). Reiner *et al*. (1993) provided an updated checklist of cetaceans found in Azorean waters, which included two new stranding records of Ziphiidae: Cuvier's beaked whale (*Ziphius caviorstris*) and Gervais' beaked whale (*Mesoplodon europaeus*). Their list included 22 species, but species records for harbour porpoise (*Phocoena phocoena*), long-finned pilot whale (*Globicephala melaena*), northern right whale (*Eubalaena glacialis*), and blue whale (*Balaenoptera musculus*) were considered "dubious". The latter designation was based on either animals recorded in the historical data (e.g. right whales) or uncertainty regarding the species identification (e.g. harbour porpoise). Subsequent publications have increased the number of cetaceans stranded or sighted in the archipelago, and confirmed the occurrence of "dubious" species. Based on strandings, Gonçalves *et al*. (1996) added fin whale (*B. physalus*) and dwarf sperm whale (*Kogia sima*) to the checklist; Steiner (1995) presented the first confirmed sighting of rough-toothed dolphin (*Steno bredanensis*); Barreiros *et al*. (2006) reported a stranded harbour porpoise; Steiner *et al*. (2007) reported seven individually identified Bryde's whales (*B. brydei cf.*) in summer 2004, including a mother-calf pair; blue whales have been confirmed by numerous sightings, and the first photograph was taken in April 1997 (Simas *et al*., 1999); Prieto and Fernandes (2007) confirmed the presence of long-finned pilot whales based on sightings off Pico in May 2003 and May 2006; Silva *et al*. (in prep.) reported the only sighting of a North Atlantic right whale within the last 80 years; Fraser's dolphins (*Lagenodelphis hosei*) were sighted by a whale-watching company in August 2008 (Serge Viallele, pers. comm.); and the True's beaked whale (*M. mirus*) and Blainville's beaked whale (*M. densirostris*) have been confirmed by a few sightings and stranding records.

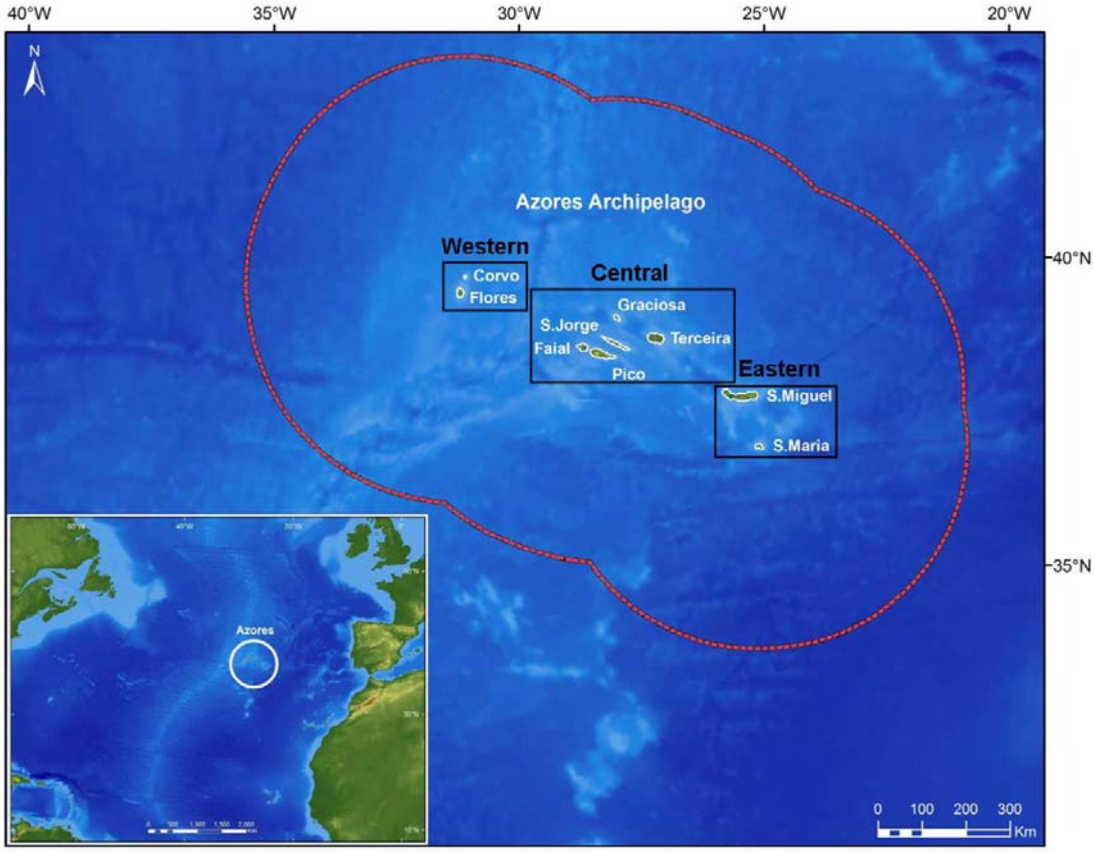


Figure 37. Map showing the Archipelago of the Azores in the Atlantic Ocean and the western, central and eastern island groups. The dashed line shows the limits of the Economic Exclusive Zone of the Azores (map by Ricardo Medeiros / *Imag*DOP).

Since, the last quarter of the 20th century a substantial level of cetacean ecological studies has been conducted in the Azores. The following is an overview of the breadth and scope of that research.

8.1.1.1 Distribution and abundance

The only boat-based line-transect survey of the cetacean complex in the Archipelago of the Azores was conducted in July–December 1999 and May–September 2000 (Silva *et al*., 2003). The survey region was portioned into island groups (i.e. eastern, central, and western) and within each group by coastal (to 9 km from shore) and offshore (9 to 28 km) waters. While all island groups were surveyed, most of the effort was in the central group (Figure 1). Within each group of islands, cetaceans were more abundant in the coastal vs. the offshore areas, and species diversity was highest in the central group of islands. The most frequently sighted small cetaceans (number of schools/100 km) were: spotted dolphins (*Stenella frontalis*) (n = 47, 0.87), common dolphins (*Delphinus delphis*) (n = 33, 0.61) and bottlenose dolphins (*Tursiops truncatus*) (n = 28, 0.52). Although there was considerable overlap in spatial distribution among these species, common dolphins and bottlenose dolphins were more frequent in coastal areas, while spotted dolphins were more common in offshore and deeper waters. Sperm whales (n = 14, 0.26) and *Mesoplodon* spp. (n = 12, 0.22) were the most frequently sighted large and medium size odontocetes.

Apart from this, no other dedicated survey was ever conducted in the waters around the Azores. As a result, there are no estimates of abundance for any cetacean species and published information on their distribution at sea is scarce. Therefore, published information presented in this report was supplemented with data collected by DOP/UAç during boat-based surveys and land-based observations. DOP/Uaç has been carrying out opportunistic surveys in all islands of the Azores since 1999 (Silva *et al*., 2008). Often these surveys did not follow predetermined tracks nor did they ensure equal probability of coverage within the study area. Nevertheless, cetacean sighting and effort data were collected following the same protocol and can be used to derive information on the occurrence and relative abundance of several species. The sighting database available at DOP/UAç also includes sighting and effort information collected by trained observers of the Azorean Fisheries Observer Programme (POPA) (Silva *et al*., 2002). Distribution of boat-based survey effort is shown in Figure 38. In addition to these, stranding records maintained by the Azores Cetacean Stranding Network (RACA) were used to document the occurrence of species that were rarely observed or difficult to identify at sea. A summary of the species found in Azorean waters and current knowledge of their frequency and seasonal occurrence is presented in Table 14.

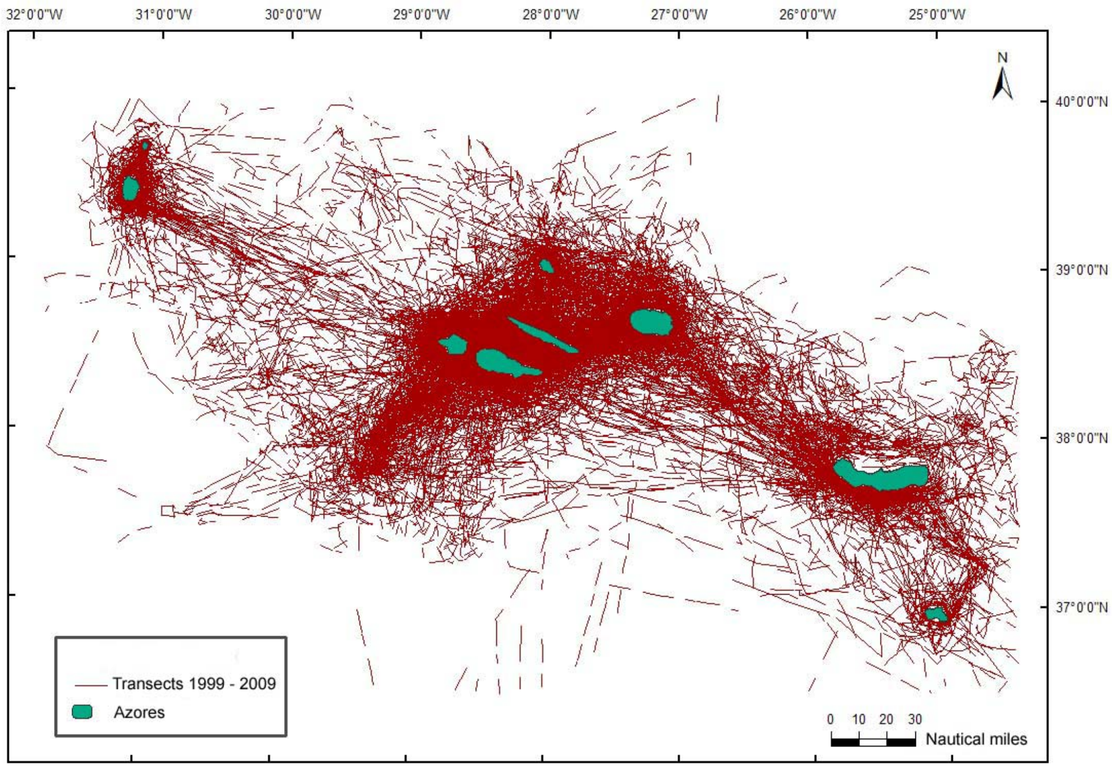


Figure 38. Boat-based transects conducted in the Azores between 1999 and 2009.

Mysticetes

Sightings of humpback whales (*Megaptera novaeangliae*) were rare in the Azores although the species visited the area almost every year. Apart from a single whale seen in October, sightings occurred between March and July, with an evident peak in May. A small number of minke whales (*Balaenoptera acutorostrata*) were observed annually from April to September, thus were considered to be uncommon in the area. However, minke whales were the most frequently stranded baleen whale, implying that it may be more common in the region than suggested by the sightings data. Stranding records also indicate a wider period of occurrence spanning from February to October. Bryde's whales were first observed in 2004. In all seven individuals, including a mother and calf pair, were photo-identified and seen repeatedly in July and August foraging around the islands of Pico and Faial (Steiner *et al*., 2007).

Sei whales were first documented in the Azores in 1989 (Gordon *et al*., 1990). Data from boat-based surveys carried out by DOP/UAç suggests sei whales were the second most frequent baleen whale species in the Azores from 1999–2009. Sei whales were usually present from April to September, with a peak in sighting frequency in June. Unlike other baleen whales, sei whales were more frequently found in groups than singly. Groups of three or more whales comprised 47% of the sightings, whereas single animals were reported <20% of the times. Calves were sighted twice in early May. Gordon *et al*. (1995) conducted multiyear observations of fin whales using boat-based observations and observers stationed at *vigia* stations (look-outs) on the islands of Faial and Pico. These authors reported that fin whales were present in the Azores from April–June, but absent from July–October. However, recent information collected by DOP/UAç indicates fin whales were the most abundant baleen whale species in the Azores and were present from January to October, with higher sighting frequencies from May to July. Aggregations of five to eleven individuals were reported in 11% of the sightings but on most occasions animals were observed as singleton (43%) or pairs (31%). There is a single observation of a calf in May. Data from DOP/UAç suggests blue whales were frequent in the Azores in late winter and spring but rare in the rest of the year. Blue whales were usually seen from February to June, with a pronounced peak in the mean sighting rate in March and a few scattered sightings throughout summer and autumn. About 44% of the sightings consisted of single individuals and 24% of two individuals, but up to 15 blue whales were observed in the study area at the same time. Calves were observed in only three occasions, in February, March and April.

The photo-id catalogue (1997–2007) maintained at DOP/UAç contains a minimum of 46 individually identified sei whales, 43 blue whales and 52 fin whales. The large majority of these individuals were sighted once but there were several resightings with 5–50 days intervals and a few resightings between years.

Odontocetes

Physeteroidea

DOP/UAç data shows that sperm whales were among the most frequent and abundant cetaceans in the area, being recorded in every month, although sighting rates were slightly lower during winter. The relative abundance of sperm whales was very consistent between years. Both adult males and groups of females with calves and immature animals were observed year-round in the Azores. Habitat preference modelling suggests that sperm whales preferentially used high relief areas between 1000–1500 m or >3000 m depth and their occurrence in the Azores was positively correlated with sea surface temperature (Seabra *et al*., 2005). Photo-identification data collected from 1997 to 2004 by DOP/UAç were used to estimate the residence rate of individual sperm whales in the Azores by calculating lagged identification rates (Whitehead, 2001). Data were best described by an emigration–re-immigration model. According to this model, average residence time of individual whales in the area was 15 days. The model also estimated a population of about 700 sperm whales visiting the Azores (Silva *et al*., 2005). A sperm whale photo-ID catalogue was initiated in the late 1980s (Matthews *et al*., 2001) using data collected in the Central Group of islands and in the island of São Miguel. Closed population (Petersen) estimates, using data from within summers suggest a population of 300–800 female or immature sperm whales in summer in the study area. Estimates of the population that visits the study area in summer were made using a model selected from the Jolly-Seber family. The open population visiting the study area appeared to vary between about 400–700 individuals between the years 1988 to 1990, increasing by a factor of three to about 1600–2200 individuals between the years 1991 to 1994.

Kogia spp. were rarely encountered during boat-based surveys and were never observed from land. However, *Kogia* spp. accounted for 4% of all strandings in the Azores. The stranding database contained one record of dwarf sperm whale (*K. sima)* from June 1996 (Gonçalves *et al*., 1996). From 2002 to 2006, four pigmy sperm whales (*K. breviceps*) were found stranded in July, August, October and December. Another six unidentified *Kogia* were reported from January–March, June, September and November. Combining all records of *Kogia* spp., members of this genus were present year-round in the Azores and possibly in larger numbers than revealed by the sighting information.

Beaked whales

In 10 years of DOP/UAç boat-based surveys, Cuvier's beaked whales were seen 30 times. The species was sighted nearly every year, from June to September, with single sightings in March and May. Stranding data indicates the species also occurred in winter (January–March) and late fall (November), suggesting Cuvier's beaked whales may be present in the region throughout the year.

Northern bottlenose whales (*Hyperoodon ampullatus*) were frequently sighted during late spring and summer. According to data from DOP/UAç boat-based surveys, the species occurred in the Azores from late May to early September, with well pronounced peaks in sighting rate in July and August. All sightings made during land-based observations occurred in July and August, suggesting the species may have a more offshore distribution during most of the year and come near the islands during summer. In 2009, aggregations of over 60 northern bottlenose whales were detected in the study area. Based on pictures collected opportunistically by the Nova Atlantis Foundation, in all 135 bottlenose whales were identified south of Pico Island. So far, one matching was found between years (Weilermann *et al*., 2009).

Beaked whales of the genus *Mesoplodon* were common in the Azores accounting for almost 3% of all sightings. *Mesoplodon* spp. were observed every month between February and October but seemed to be more common during summer. Sighting rates calculated from boat-based data increased drastically from June to July, and gradually decreased from July to October. Average school size was 4.4 individuals but the majority of schools sighted were composed of six animals. From 1994 to 2009, there were eight strandings of Sowerby's beaked whale (*M. bidens*) all of which took place between June and August. Five of these strandings involved between 2 and 5 individuals, and in three strandings the individuals were alive. There was a single stranding of two Blainville's beaked whales in March, one Gervais beaked whale in August, one True's beaked whale in July and four unidentified *Mesoplodon* spp. in March, July and August.

Delphinids

Bottlenose dolphin was the third most frequent and abundant species in the Azores, comprising nearly 10% of all sightings. Despite slight variations in their mean monthly sighting rate, likely caused by lower observation effort in autumn and winter, bottlenose dolphins appeared to be equally abundant throughout the year. Silva (2007) studied the relationship between the distribution of bottlenose dolphins in the Azores and several physiographical and physical variables, using generalized linear models. Dolphins significantly preferred shallower depths (between 100 and 600 m) and areas with higher slopes, and tended to avoid depths greater than 1000 m, and slopes smaller than four degrees. Bottlenose dolphins showed a clear preference for areas within 2–5 km from the islands, and areas >25 km away, which is likely related to the presence of a large shallow-water seamount complex in the central group of islands. The models also indicated persistence of dolphin-habitat associations, suggesting the possibility of identifying important areas of habitat for this species based on static bathymetric features, which would be clearly important in an oceanic and dynamic ecosystem such as the Azores.

Silva et al. (2008) conducted a six-year boat-based study on movements, home ranges and site fidelity of bottlenose dolphins around the islands of Faial, Pico and S. Jorge (central group of islands) (Figure 37). Only 44 individuals out of 966 identified were frequently sighted within and between years. The remaining individuals were either temporary migrants from within or outside the archipelago, or transients. Resident dolphins showed strong geographic fidelity to the area but estimates of home range size were three times larger than previously reported for this species, possibly as a result of the lower availability of food resources and lack of predation pressure. The population size of dolphins using this area was estimated using open-population models and Pollock's robust design (Silva *et al*., 2009a). There were wide interannual variations in the number of bottlenose dolphins that occurred in the area but most of these used the area only temporarily, as suggested by the high emigration rates produced by the models. The annual abundance of adults dolphins varied between 114 (95% CI: 85 to 152) and 334 (95% CI: 237 to 469) and the number of subadult individuals varied from 300 (95% CI: 232 to 387) to 434 (95% CI: 316 to 597).

The seasonal occurrence of spotted dolphins in the Azores was very noticeable: first sightings occurred in early May and the highest relative abundance was reached in July or August, depending on the years. By October the species disappeared from the region. Between July and September, spotted dolphins usually were the most frequently sighted species and also outnumbered common dolphins. Overall, spotted dolphins were the second most frequent species in the Azores, comprising 17% of all sightings. They preferred medium depth waters (750–1250 m) at 2–6 nm from the islands (Quérouil, in prep.).

Sightings of striped dolphins (*Stenella coeruleoalba*) from land-based observations were rare and, with the exception of a sighting in October, limited to summer, from May to July. Stranding records contradict these results and reveal that striped dolphins occurred regularly in the area and in most months. Striped dolphins were encountered 157 times during sighting surveys with an average of 0.05 schools/100 km. Striped dolphins were sighted every month, except January, February and November.

Common dolphins were the most frequently sighted and abundant cetacean in the region. It accounted for 38% of all sightings made during boat-based surveys and had a mean sighting rate of 1.1 schools per 100 km of daily effort. Animals were recorded year-round in the Azores but showed a significant reduction in relative abundance during summer and autumn (June to October), although this pattern was more evident in some years. In contrast, common dolphin relative abundance varied only slightly between years. They were predominantly found in medium-depth water (500–1000 m) all year-round, and tended to avoid deep waters (>1500 m) in summer and shallow waters (<250 m) during the rest of the year. Common dolphin sighting rates were predominantly found either within 2–8 nm from shore or more than 35 nm offshore during most of the year, with a tendency for areas closer to the islands during summer (Quérouil, in prep.).

Quérouil *et al*. (2008) examined mixed-species associations involving common dolphin, bottlenose dolphin, striped dolphin and spotted dolphin based on 4369 sightings of at least one species between 2001 and 2004. Data sources were dedicated cetacean surveys from DOP/UAç and from fisheries observers. The percentage of sightings in association was low (2–7%) for all species except striped dolphin, which was observed up to 31% of the time in the observer programme data. Overall, the study found that mixed species associations are rare, but those that do occur are associated with foraging activities.

Morato *et al*. (2008) examined spotted dolphin, common dolphin, bottlenose dolphin, and sperm whale and other marine predators (e.g. tuna, sea turtles, and seabirds) association with Azorean seamounts. Common dolphin was the only cetacean that was significantly more abundant around shallow (<400 m) seamounts.

Risso's dolphin (*Grampus griseus*) was the fifth most common species in the Azores, comprising nearly 6% of all sightings. Boat-based sighting data showed gaps in the presence of Risso's dolphins in the area, but combining all sources of information it becomes clear that the species occurred in the Azores year-round, with slightly higher relative abundances in spring and summer. Pereira (2008) reported on the ecology of Risso's dolphins south of Pico Island. This author determined that Risso's preferred areas between 500 and 1200 m, with slopes around 30%. Most group sizes were composed of 20 or fewer animals, which is consistent with other regions. However, large socializing groups of ~175 individuals have been observed. Based on photo-identification data collected from 2004–2006, Hartman *et al*. (2008) identified 1028 Risso's dolphins at Pico Island. High resighting rates suggest strong site fidelity for a part of this population. These authors found that dolphins formed strong and stable bonds and suggested that dolphins identified in this area likely composed a single social network.

False killer whales (*Pseudorca crassidens*) were regular visitors in the region and were sighted every year. Information from sighting surveys suggests the species only occurred in spring and summer, but land-based observers reported false killer whales in every month except January and December. However, about 50% of the sightings made from land occurred in August and September, which is in agreement with boat-based data that also showed higher mean sighting rates in those months. In 40% of the land-based sightings, false killer whales were seen in association with bottlenose dolphins.

Short-finned pilot whales (*Globicephala macrorhynchus)* were frequently sighted and were recorded in every month, with the exception of December. Similar to false killer whales, land-based data indicates a wider period of occurrence for this species but pilot whales seemed to be more frequent in late spring and early summer. The recent confirmation of the occurrence of long-finned pilot whales in the Azores (Prieto and Fernandes (2007) may raise doubts about the identity of a few of these sightings.

Killer whales (*Orcinus orca*) were uncommon in the Azores, being sighted two or three times per year. Sightings occurred between February and August. Groups ranged from five to seven individuals.

8.1.2 Population/stock structure

Mysticetes

Some work on the ecology of baleen whales has been developed in the last years by the cetacean research team at the DOP/UAç, using several techniques, including satellite telemetry. Data on sei whale movements from satellite telemetry indicate a link between the Azores and the Labrador Sea, during spring/summer migration. Olsen *et al*. (2009) tagged one sei whale in 2005 that went to the Charlie Gibbs Fracture Zone and from there to the Labrador Sea. Subsequently, another seven whales were tagged off the Azores during spring and early summer in 2008 and 2009, and were all tracked to the Labrador Sea (Prieto, pers. comm.). The timing of the movements into the Labrador Sea is coincident with the known season of sei whale presence in Nova Scotia and Gulf of Maine, giving some support to the idea of a stock separation between the Labrador Sea and Nova Scotia proposed by the International Whaling Commission [IWC] (Mitchell and Chapman, 1977; Perry *et al*. 1999). On the other hand, the recorded movements indicate a link between the Labrador Sea stock and the southern part of the Iceland-Denmark Strait stock, implying that a revision of the delimitations and ecological meaningfulness of these stocks should be considered.

Blue whale photo-identification samples have been compared with the catalogue for the Northwest Atlantic, and no matches have been made so far. Skin samples collected by DOP/UAç from blue, fin and sei whales are being analysed by the University of Stockholm to examine the genetic structure of these species at the scale of the North Atlantic and Mediterranean basins.

An adult North Atlantic right whale was observed less than one mile south of Pico island in January 2009 and was identified as a female from the western Atlantic population known as #3270 in the North Atlantic Right Whale Catalogue (NARWC), maintained at the New England Aquarium. This whale was renamed as *Pico,* accordingly. *Pico*'s sighting in the Azores is the only record of the species in the area within the last 80 years. Most of the whales known from the NARWC are regularly seen at least in one of the five well-studied critical habitats. Yet, several catalogued whales show sighting frequencies well below the average of the rest of the population (Hamilton *et al*., 2007). *Pico*, on the other hand, was seen consistently in the population's foraging habitats and would probably be classified as a regular whale. *Pico*'s documented excursion to the Azores and back means that long-distance movements are not restricted to individuals with lower site fidelity to the population's critical habitats (Silva *et al*., in prep).

Wenzel *et al*. (2009) reported a resighting of a humpback whale photographed in the Azores in June 2006 that was recaptured in Cape Verde on April 2009. The authors suggest that the Azores are part of the migratory corridor for the migrating Northeast Atlantic humpback whales during spring/summer migration.

Odontocetes

Physeteroidea

Sperm whale population and social structure in the Azores were evaluated by Pinela *et al*. (2009) using microsatellite analyses. Their findings suggest that sperm whales visiting the Azores are part of a large single population, showing site fidelity to the Azores, and both high genetic diversity and an absence of inbreeding. The authors suggest that the social structure is similar to other regions, where primary units of sperm whales and secondary social groups are mainly, but not exclusively, composed of members of the same family. Likewise, fatty acid analyses did not discern any differences throughout the archipelago (Walton *et al*., 2008). Genetic analyses of a larger number of individuals and a complete sampling of social groups is required to provide a North Atlantic wide understanding of sperm whale social organization (Pinela *et al*., 2009). To date, sperm whale stock division has only been detected between the Mediterranean Sea and the North Atlantic (Drout, 2003). Steiner *et al*., (2009) used photo-ID data to examine the long distance movements of male sperm whales, and made three matches of animals moving between the Azores and Norway (~2400 nm). In addition to these, a two-way movement between the Azores and the Canary Islands was documented for one female (Steiner, pers. comm.). Recoveries of harpoons and tags from the whaling period documented movements from the Azores to Iceland and Spain (Steiner *et al.*, 2009).

Delphinids

Photo-identification data collected over a period of six years showed that bottlenose dolphins have the capability of undertaking movements of almost 300 km between the island groups (Silva *et al.*, 2008). Also, dolphins from different genders and age classes showed similar ranging patterns. The high mobility of individuals and the varying patterns of residence in any single area suggest that bottlenose dolphins in the Azores constitute a single and open population, composed of several geographic communities that maintain social interactions with neighbouring communities and groups from within and outside the archipelago. These interactions are facilitated by the extensive ranging behaviour of some individuals and groups and by an apparent lack of habitat partitioning.

Analysis of mitochondrial DNA and microsatellite DNA markers of bottlenose dolphins from the Azores showed a high genetic variability, similar to that obtained for the Mediterranean Sea and Northwest Atlantic pelagic populations, and a lack of population structure within the Azores (Quérouil *et al.*, 2007). This information is in agreement with results from fatty acid analyses showing no differences in fatty acid profiles between genders or age classes and between dolphins living in different groups of islands of the Azores (Walton *et al.*, 2007). MtDNA indicated significant differentiation between the Azores and Madeira, but not between the two archipelagos and mainland Portugal, although the latter result could be due to small sample size for the mainland (Quérouil *et al.*, 2007). Microsatellites showed no population differentiation between the three study sites. MtDNA sequences indicated that the population of the Azores was significantly differentiated from all the Atlantic Basin populations except the Northwest Atlantic pelagic population. The population of the Azores would thus be of the pelagic type, despite the fact that bottlenose dolphins are primarily encountered within 9 km from the shore in Azorean waters (Silva *et al.* 2003).

In the North-east Atlantic, common dolphins are one of the most abundant and widely distributed small cetaceans. To date, one population has been reported within this region, ranging from Scotland to Portugal, and with separate populations in the Mediterranean Sea and North-west Atlantic (Murphy *et al.*, 2009; ICES WGMME, 2009). The actual distributional range of the North-east Atlantic population is unknown. Amaral *et al.* (2007) investigated stock structure in the North-east Atlantic, which included four genetic samples collected in the Azores. Their study found evidence of the existence of a sex-biased population in this region (although sample sizes were small), which supports previous findings based on morphometrics (Murphy *et al.*, 2006). The overall mitochondrial DNA genetic variability estimates for the North-east Atlantic were similar to studies conducted in other regions. Further, Amaral *et al.* (2007) detected some shared haplotypes between samples from the Canary Islands and North-east Atlantic which indicate some level of gene flow may exist. There were no haplotypes shared between the Azores samples and other regions but that may be due to the small sample size (n = 4).

Quérouil *et al.* (submitted) examined the population genetic structure of common dolphins in the Azores and Madeira, sequencing part of the mitochondrial hyper-variable region, screening a dozen microsatellite loci from 147 individuals. The results did not unravel any population structure at the scale of the study area, either within or between archipelagos. The authors consider that the obtained values must be regarded as raw estimates, nevertheless, most of them exceeded the "one migrant per generation" threshold that was formerly used to assess connectivity between populations, as well as the more realistic "up to ten migrants per generation" threshold that is currently preferred definition for conservation management units (Mills and Allendorf, 1996). Owing to that criterion, the authors argue that the populations of the Azores and Madeira should be regarded as a single management unit. Nonetheless, these authors caution about the possibility of undetected subpopulations.

Quérouil *et al*. (submitted) also examined the population genetic structure of Atlantic spotted dolphins in the Azores and Madeira, using the same methodology described above on 191 samples. The results were similar to those found for the common dolphin, pointing towards a lack of genetic population structure at the scale of the study area, either within or between archipelagos. For this species, though, lack of population structure could be expected, given that they are temporary visitors in the Azores and Madeira and tend to prefer offshore waters. The authors argue that the populations of the Azores and Madeira should be regarded as a single management unit within each species, for the same reason presented above for the common dolphin.

8.1.3 Bycatch

There are four main fisheries in the Azores: i) a fishery for small pelagics (*Trachurus picturatus, Scomber japonicus, Sardina pilchardus*) using small seinenets, dipnets and liftnets; ii) a seasonal pole-and-line tuna fishery, iii) a multispecific demersal fishery that uses handlines and bottom longlines and iv) a swordfish (*Xiphias gladius*) fishery using surface longlines. There is also a small coastal gillnet fishery that catches a variety of pelagic and benthic fish species. The use of gillnets is limited to an area <500 m from the coastline and to depths <30 m. The fishery for cephalopods and crustaceans is a small-scale, mostly seasonal activity carried out by snorkel divers and handpickers, or using bottom traps, iron traps and jigs. Purse seinenets for tuna, trammelnets, drift gillnets, driftnets, bottom trawling and other deep-sea net are banned from the Azorean EEZ (Silva *et al*., submitted).

Silva *et al*. (2002; submitted) examined interactions between cetaceans and several fisheries in the Azores. The tuna pole-and-line fishery is one of the most valuable fisheries in the Azores. The fishery operates from May to October and trip duration is from five to six days (Silva *et al*. 2002). Vessels search for tuna schools using seabirds and floating objects as cues. From 1998 to 2006, 1526 trips were monitored, during which 14 851 fishing events were recorded. Observer coverage (tonnage of tuna landed by vessels with observers divided by total fleet landing) ranged from a minimum of 32% in 2003 to 67% in 1999 (Silva *et al*., submitted). Cetaceans were present in about 7% (973/14 851) of the fishing events, including baleen whales, sperm whales, and a variety of delphinids. Common dolphins, spotted dolphins, and bottlenose dolphins, respectively, accounted for 73%, 14% and 7% of total interactions. Cetacean interference with the fishing activity (i.e. tuna schools sank and competition for bait) was noted in 452 (46%) fishing events. Common dolphins (73%) were responsible for most of the interferences, followed by spotted dolphins (16%), bottlenose dolphins (10%). A couple of interactions were observed with striped dolphins, false killer whales, and Risso's dolphin (Silva *et al*., 2002, submitted).

Incidental hooking of 59 dolphins was observed (48 common dolphins, nine spotted dolphins, one bottlenose dolphin, and one small unidentified dolphin) (Silva *et al*., 2002). All animals were released alive by cutting the fishing line, thus the level of injury or subsequent mortality is unknown. The majority (80%) of these interactions were observed during the first three years (1998–2000) of the study, and in 2003–2004 there were no observed incidental captures (Silva *et al.*, submitted). The estimated numbers of captures (95% CI in parentheses) were: 38 (16.91–59.06) in 1998, 37 (22.78–51.79) in 1999, 16 (11.74–20.19) in 2000, 2 (0.12–4.12) in 2001, 2 (1.14–5.56) in 2002, 0 in 2003 and 2004, 11 (2.71–20.17) in 2005, and 3 (1.25–6.29) in 2006 (Silva *et al.*, submitted). Reasons for the decline in the capture rate are unknown but could be related to the substantial decrease in fishing effort from 2001 onwards (Silva *et al.*, submitted) and to a change in fish species used as live bait.

Silva *et al*. (submitted) reviewed interactions with demersal fisheries that use handlines and bottom longlines to catch more than 20 species. Observer monitoring of these fisheries began in 2004 and observer coverage, calculated as observed landing/total landings, was 1% or less from 2004 to 2006. Short-term monitoring was also conducted on four commercial longline boats between August and September 2004, and a single handline boat was monitored between May 2002 and August 2004. Cetacean interactions were restricted to depredation. Depredation was noted in 25%, 16% and 2% of the sets observed in 2004, 2005, and 2006, respectively. Bottlenose dolphins (n=68), common dolphins (n=10), and Risso's dolphins (n=1) were the only cetaceans observed in the vicinity of the fishing operations, but bottlenose dolphins were responsible for all depredation events.

Short-term monitoring efforts were also conducted in several other fisheries. In 1998 and from 2000 to 2004 a low level (i.e. 0.6% of the sets and 0.5% of the hooks) of observer coverage was conducted on swordfish longline boats (Silva *et al.*, in press). Cetaceans were recorded in the vicinity of the gear on 5% (20/384) of the observed sets. Bottlenose dolphins were seen three times; Risso's dolphin and killer whales were each seen twice; common dolphins, spotted dolphins, pilot whales (*Globicephala* spp.), false killer whales, and sperm whales were recorded once. Cetaceans damaged blue shark (*Prionace glauca*) catch in three sets, and the damage was consistent with killer whale or false killer whale attacks. No cetaceans were bycaught.

Between 1999 and 2005, observers were placed aboard six commercial boats in the deep-water (1000–2000 m) drifting bottom longline fishery for black scabbard (*Aphanopus carbo*). The level of observer coverage could not be determined, but there were no cetacean interactions in 240 observed sets (Silva *et al.*, submitted).

Three experimental fisheries were also monitored: 1) In April–June 2001, and December 2001–January 2002 observers monitored 246 hauls in the orange roughy (*Hoplosethus atlanticus*) trawl fishery; 2) In 2003 and 2004, observers monitored 200 sets in the deep-water crab (*Chaceon affinis*) trap pot fishery; and 3) In November 2006, 23 sets in the deep-water pandalid shrimp (*Plesionika edwardsii*) trap fishery were observed. There were no cetacean interactions recorded in any of the experimental fisheries (Silva *et al.*, submitted).

Since 2008, Azorean fishermen have been complaining about cetacean interactions in the squid jig fishery. In July 2009, DOP/UAç began monitoring this fishery through interview surveys of fishermen and by placing observers on board fishing boats. There were no reports of cetacean incidental mortality or injury associated with this fishery. In 74% of the 127 interviews, fishermen stated that dolphins interfered in the fishery by removing the whole squid or the squid mantle from the jigs. Dolphins were also responsible for damages to the fishing gear in 11% of the events reported during interviews. Cetacean interaction was reported in 68% of the interviews conducted in the island of S. Miguel (n= 82) and in 100% of the interviews in S. Jorge (n=15). The Risso's dolphin was the most frequently sighted species during fishing (60%) and was also responsible for most depredation events. The bottlenose dolphin was the second most sighted species during fishing (22%), and responsible for 7% of all depredation events. Between July and December 2009, in all ten fishing trips in five fishing boats were monitored. On-board observations were carried out in S. Jorge (n= 6) and in S. Miguel (n=4). In S. Jorge no cetaceans were sighted during fishing operations. Risso's dolphins were present and interfered in all fishing trips monitored in S. Miguel. Catch loss resulting from depredation varied from 3.5–34%. To avoid depredation fishermen changed the fishing gear to target other species, suspended fishing until the dolphins abandoned the area, or searched for another fishing ground.

8.1.4 Whale watching and dolphin swim programmes

Oliveira *et al*. (2007) reviewed the whale watching management in the Azores. The process of creating a whale watching regulation started in 1995 when a proposal from a whale watching operator was submitted to the Regional Directorate of Tourism (DRT). Both parties with the DOP/UAç agreed to start developing the whale watching regulation. In 1996, DOP/UAç initiated a project to determine the best methodologies to evaluate the impact of whale watching on cetaceans, develop a survey form to obtain tourist opinions on whale watching and to draft the whale watching regulations. The refinement of this proposed legislation was based on fieldwork (Gaspar and Gonçalves, 1997), worldwide regulations, guidelines and recommendations (IFAW *et al*., 1995; IFAW 1996) and NGO's and specialists' opinions. Following several revisions based on stakeholder input a final law was created in 1999. The law was revised in 2003 and implemented in 2005. Presently the activity is regulated by both the law order ('DL 10/2003/A') and the Governmental order ('Portaria 5/2004'). The regulation currently limits the number of licences by zone (25 licences for the waters around Faial, Pico, S. Jorge, in the Central group, Flores and Corvo islands, the Western group and 20 licences for São Miguel and Santa Maria islands (Eastern group). The regulation limits boat manoeuvres and speed around the animals. Vessels operators are not allowed to pursue animals, cause group separation, engage in feeding of the animals, allow people to dive with scuba gear or use motor swimming aids, pollute, and conduct night-time operations. Approaches should be made from the rear, leaving a 180º sector free ahead of the animals, to a distance no closer than 50 m from the animal and for no longer than 30 minutes. Engines must be idled at the sight of an approaching animal, and boats must depart area moving away from the animals, when detecting avoidance behaviour. Sailing boats must have the engine turned on at all times. Boats should coordinate among themselves to avoid having more than three boats within 300 m radius of a group, leaving priority to the first vessel that arrived. Swimming with most species is forbidden, although licences can be issued for scientific research and multimedia professionals. Swimming is permitted for some species of dolphins, but under controlled conditions: no more than 2 persons are allowed to be in the water simultaneously, swimmers cannot stay in the water for >15 minutes, boats are required to have a crew member dedicated to surveillance of swimmers.

Magalhães *et al*. (2002) conducted detailed land-based and boat-based observations of sperm whale behaviour in the presence and absence of whale watching activities. Feeding was the main activity, and the presence of calves suggests that it is an important foraging site for female sperm whales with calves. Changes in feeding or socializing/resting behaviour were not detected in the presence of boats. However, from boat based observations, as opposed to land-based ones, sperm whale swimming speed and aerial displays increased when boats made manoeuvres considered inappropriate in the code of conduct. The presence of swimmers led to a significant increase in aerial displays. In the presence of boats, mature females and immature individuals significantly increased mean blow interval when accompanied by calves.

The authors note that potential for disruption of breeding is an issue of concern, and needed to be monitored in future studies.

Magalhães *et al*. (2007) further investigated the reactions of sperm whales and bottlenose dolphins using land-based observations of interactions with whale watching activities south of Pico Island. Sperm whale movements were significantly more linear in the presence of boats. The presence of boats did not affect significantly the magnitude of the overall tracks. This implied that even if an individual changed course several times during a track, the deviation from its initial course was not substantial. Mean swimming speed or diving patterns (fluke-up) of sperm whales were not significantly altered in presence of boats, nor were they related to distance of approach. However, sperm whales moved more frequently away from boats at shorter distances of approach, or when changes of speed were performed by boats.

In contrast, bottlenose dolphin movements, swimming speed, spatial arrangement, aggregation type or degree were unaffected by boats or their distance of approach. Orientation of bottlenose dolphins relative to boats was not correlated with distance of approach, but was affected by course changes of vessels, but not their speed changes were unaffected by boats (Magalhães *et al*., 2007).

Behavioural responses of Risso's dolphins to whale-watching vessels were documented using land-based observations off Pico Island (Visser *et al*., in press). Whale watching vessels affected the resting behaviour of the animals. The peak resting activity of Risso's dolphin during the high whale watching season was shifted to the hours of lowest vessel activity. Resting rate was negatively, but not linearly, related to number of vessels, which explained the difference observed between low and high whale watching seasons. The presence of more than four vessels had a significant negative effect on resting behaviour and positive effect on travelling, indicating a behavioural shift. The incidence of foraging and socializing behaviour were unrelated to whale-watching intensity, indicating that these behaviours are less sensitive to vessel presence. Based on these results the authors suggested that the number of vessels should be limited by area and that time windows should be introduced in the whale watching management procedures to create resting opportunities for Risso's dolphins.

8.2 Pinnipeds

Silva *et al*. (2009b) summarized historical and recent occurrences of pinnipeds in the Azores. Historical records denote the presence of Mediterranean monk seal (*Monachus monachus*) colonies, and the authors suggest that sealing by the early colonizers likely extirpated the population. Since the beginning of the 20th century, the extralimital presence of thirteen individual seals has been recorded. This includes one ringed seal (*Pusa hispida*), three Mediterranean monk seals, two harbour seals (*Phoca vitulina*), two grey seals (*Halichoerus grypus*), one harp seal (*Pagophilus groenlandicus*), two hooded seal (*Cystophora cristata*), and two unidentified seals.

8.3 Summary

The review documents provided to WGMME clearly delineates the broad scope of the Azorean cetacean research programme. Since the latter quarter of the 20th century information on cetaceans inhabiting the Azores has moved beyond historical whaling and strandings data to a comprehensive international programme. Research has been published in a number of peer reviewed journals and presented at scientific conferences (e.g. European Cetacean Society, Society for the Biology of Marine Mammals).

Ongoing studies of large baleen whales suggest that Azorean waters might represent an important stopover during migration to high latitude summer feeding grounds.

Overall, at least seven cetacean species occur year-round in Azorean waters. Several other cetacean species seasonally occupy these waters to meet important life-history requirements (i.e. feeding, calving, nursing, and breeding, etc.). These proportions are likely to increase as more information is acquired through ongoing research programmes.

Table 14. List of cetaceans recorded in the Azorean EEZ (T- strandings; S- sightings; C- catches (Reiner *et al.*, 1990; Gonçalves *et al.*, 1992; 1996; Prieto and Fernandes, 2007; Steiner *et al.*, 2007; Unpublished sighting and stranding records from DOP/UAç and the Azores Stranding Network (marked with an asterisk "*"). Frequency: A- Accidental, R – Rare, O – Occasional, C- Common, U – Undetermined; Seasonality: Y-Year-round; S-Seasonal; U – Undetermined.

Latin name	Common name	Type of record	Frequency	Seasonality
Eubalaena glacialis (Muller, 1776)	Northern right whale*	S, C	A	---
Megaptera novaeangliae (Borowski, 1781)	Humpback whale	T, S	R	S (Spring–Summer)
Balaenoptera acutorostrata Lacépède, 1804	Minke whale	T, S	R	U
Balaenoptera brydei cf. Anderson, 1878	Bryde's whale	S	U	U
Balaenoptera borealis Lesson, 1828	Sei whale	T, S	C	S (Spring–Summer)
Balaenoptera physalus (Linnaeus, 1758)	Fin whale	T, S	C	S (Spring–Summer)
Balaenoptera musculus (Linnaeus, 1758)	Blue whale	S	C	S (Spring–Summer)
Physeter macrocephalus (= *catodon*) Linnaeus, 1758	Sperm whale	T, S, C	C	Y
Kogia sima Owen, 1866	Dwarf sperm whale	T	U	U
Kogia breviceps (de Blainville, 1838)	Pygmy sperm whale	T, C	U	U
Ziphius cavirostris Cuvier, 1823	Cuvier's beaked whale	T, S	O	Y
Hyperoodon ampullatus (Forster, 1770)	Northern bottlenose whale	S	O	S (Spring–Summer)
Mesoplodon bidens (Sowerby, 1804)	Sowerby's beaked whale	T, S, C	U	U
Mesoplodon europaeus Gervais, 1855	Gervais' beaked whale	T	U	U
Mesoplodon mirus True, 1913	True´s beaked whale*	T, S	U	U
Mesoplodon densirostris (de Blainville, 1817)	Blainville´s beaked whale*	T, S	U	U
Tursiops truncatus (Montagu, 1821)	Bottlenose dolphin	T, S, C	C	Y
Steno bredanensis (Lesson, 1828)	Rough toothed dolphin	S	A	---
Stenella frontalis (Cuvier, 1829)	Spotted dolphin	T, S, C	C	S (Spring–Summer)
Stenella coeruleoalba (Meyen, 1833)	Striped dolphin	T, S, C	C	S (Spring–Autumn)
Lagenodelphis hosei Fraser, 1956	Fraser's dolphin*	S	A	---
Delphinus delphis Linnaeus, 1758	Common dolphin	T, S, C	C	Y
Grampus griseus (Cuvier, 1812)	Risso's dolphin	T, S, C	C	Y
Pseudorca crassidens (Owens, 1846)	False killer whale	T, S, C	O	Y
Orcinus orca (Linnaeus, 1758)	Killer whale	S, C	R	U
Globicephala melas (= *melaena*) (Trail, 1809)	Long-finned pilot whale	S, C	U	U
Globicephala macrorhynchus Gray, 1846	Short-finned pilot whale	T, S	O	Y
Phocoena phocoena (Linnaeus, 1758)	Harbour porpoise	S	A	---

8.4 Recommandations for quantitative conservation criteria

1) Conduct systematic cetacean surveys of the Azores archipelago every 3 to 5 years-implement survey design established in Faustino *et al*. (in press);
2) Bycatch monitoring-existing Azorean observation programmes should be expanded to increase observation effort of some Azorean fisheries (e.g. demersal) and allow monitoring of other fisheries (e.g. the swordfish fishery);
3) Bycatch monitoring of the European deep-water longline fleet that fish in the outer 100 nm of the Azores archipelago needs to be implemented;
4) Research on cetacean stock structure should be continued and expanded within the Azores archipelago;
5) Research on contaminants should be initiated using good condition stranded cetaceans within the Azores archipelago;
6) Maintain/enhance monitoring of whale watching and dolphin swim programmes to develop long-term data on the potential impact of these activities within the Azores archipelago;
7) Ensure compliance of existing whale-watching regulations in the Azores archipelago through the establishment of an efficient law-enforcement scheme;
8) Maintain collaborative research programmes with international cetacean researchers.

8.5 References

Amaral, A. R., Sequeira, M., Martínez-Cedeira, J., and Coelho, M.M. 2007. New insights on population genetic structure of *Delphinus delphis* from the North-east Atlantic and phylogenetic relationships within the genus inferred from two mitochondrial markers. Marine Biology 151:1967–1976.

Ávila de Melo, A., and Martin. A. 1985. A study of male sperm whale length data from the Azorean and Madeiran catches, 1947–1982. Report of the International Whaling Commission: 209.

Barreiros, J., Teves, M., and Rodeia. J. 2006. First record of the Harbour Porpoise, *Phocoena phocoena* (Cetacea: *Phocoenidae*) in the Azores (NE Atlantic). Aqua 11: 45–46.

Clarke, R. 1956. Sperm whales of the Azores. Discovery Reports 28:237–298.

Clarke, R. 1981. Whales and dolphins of the Azores and their exploitation. Report of the International Whaling Commission 31:607–615.

Drout, V. 2003. Ecology of sperm whales (*Physeter macrocephalus*) in the Mediterranean Sea. . University of Wales, Bangor.

Faustino, C.E.S., Silva, M.A., Marques, T.A., and Thomas, L. in press. Designing a shipboard line transect survey to estimate cetacean abundance off the Azores Archipelago. Arquipélago. Life and Marine Sciences.

Gaspar, R.H., and Gonçalves, J.M. 1997. Relatório de actividades de 1996 do projecto: Regulamentação da Observação de Cetáceos (‘Whale Watching’) nos Açores: Impacto das Embarcações nos Animais. Departamento de Oceanografia e Pescas. Universidade dos Açores. Arquivos do DOP, Série de Estudos N.º4/97. 53 pp.

Gonçalves, J., Galhardo, L., and Brum, J. 1992. Marine mammals stranded in the Azores during 1990–1991. Arquipélago: 113–118.

Gordon, J.C.D., Steiner, L., and H.R. Martins. 1995. Observations of fin whales (*Balaenoptera physalus* L., 1758) around the central North Atlantic islands of the Azores and Madeira. Arquipélago. Life and Earth Sciences: 79–84.

Gonçalves, J., Barreiros, J., Azevedo, J., and Norberto, R. 1996. Cetaceans stranded in the Azores during 1992–1996. Arquipélago. Life and Marine Sciences 14: 57–65.

Gordon, J., Steiner, L., and Gonçalves. J. 1990. Sei whales (*Balaenoptera borealis*) encountered in the Azores: a new record for the region. Arquipélago. Life and Earth Sciences 8:97–100.

Hamilton, P. K. and L.A. Cooper. in press. Changes in North Atlantic right whale (*Eubalaena glacialis*) cow–calf association times and use of the calving ground: 1993–2005. Marine Mammal Science.

Hartman, K.L, Visser, F., and Hendriks A.J.E. 2008. Social structure of Risso's dolphins (*Grampus griseus*) at the Azores: a stratified community based on highly associated social units. Canadian Journal of Zoology 86(4): 294–306.

ICES WGMME. 2009. Report of the ICES Working Group on Marine Mammal Ecology. February 2–6 2009, Vigo, Spain.

IFAW. 1996. Report of the workshop on the Special Aspects of Watching Sperm Whales. Roseau, Commonwealth of Dominica. 36 pp.

IFAW, Téthys Research Institute and Europe Conservation. 1995. Report of the workshop on the Scientific Aspects of Managing Whale Watching. Montecastello di Vibio. Italy. 40 pp.

Magalhães, S., Prieto, R., Silva, M., Gonçalves, J., Afonso-Dias, M., and Santos. R.S. 2002. Short-term reactions of sperm whales (*Physeter macrocephalus*) to whale-watching vessels in the Azores. Aquatic Mammals 28: 267–274.

Magalhães, S., Prieto, R., Silva, M.A., and Santos, R.S. 2007. Impact of whale watching on cetaceans: assessing the appropriateness of existing regulations. International Whaling Commission Scientific Committee Document SC/59/WW9.

Martin, A. 1988. Cetaceans around the central Azores Islands in the summers of 1985 and 1986. Mammalia (Paris) 52: 541–547.

Matthews, J., Steiner, L., and Gordon. J. 2001. Mark-recapture analysis of sperm whale (*Physeter macrocephalus*) photo-id data from the Azores (1987–1995). Journal of Cetacean Research and Management 3: 219–226.

Mills, L.S., and Allendorf, F.W. 1996. The one-migrant-per-generation rule in conservation and management. Conservation Biology 10: 1509–1518.

Mitchell, E., and Chapman, D.G. 1977. Preliminary assessment of stocks of Northwest Atlantic sei whales (*Balaenoptera borealis*). Report of the International Whaling Commission (Special Issue) 1: 117–120.

Morato, T., D. A. Varkey, C. Damaso, M. Machete, M. B. Santos, R. Prieto, R. S. Santos, and T. J. Pitcher. 2008. Evidence of a seamount effect on aggregating visitors. Marine Ecology Progress Series 357: 23–32.

Murphy, S., Herman, J., Pierce, G., Rogan, E., and Kitchener. A. 2006. Taxonomic status and geographical cranial variation of common dolphins (*Delphinus*) in the eastern North Atlantic. Marine Mammal Science 22: 573–599.

Murphy, S., Natoli, A., Amaral, A.R., Mirimin, L., Viricel, A., Caurant, F., Hoelzel, R., and Evans, P. 2009. Short-beaked common dolphin *Delphinus delphis*. Report of ASCOBANS/HELCOM small cetacean population structure workshop. 8–10 October 2007, UN Campus, Hermann-Ehlers-Str. 10, 53113 Bonn, Germany. 111–130.

Oliveira, C., Gonçalves, J.G., Magalhães, S., Prieto, R., Silva, M.A., Santos, R. S. 2007. Whale watching management in the Azores: an updated review of the regulations. International Whaling Commission Scientific Committee Document SC/59/WW7.

Olsen, E., Budgell, W. P., Head, E., Kleivane, L., Nottestad, L., Prieto, R., Silva, M. A., Skov, H., Vikingsson, G. A., and Waring, G. 2009. First Satellite-Tracked Long-Distance Movement of a Sei Whale (*Balaenoptera borealis*) in the North Atlantic. Aquatic Mammals 35: 313–318.

Pereira, N. J. D. S. G. 2008. Field notes on Risso's dolphin (*Grampus griseus*) distribution, social ecology, behaviour, and occurrence in the Azores. Aquatic Mammals 34:426–435.

Perry, S. L., DeMaster, D. P., and Silber, G. K. 1999. The great whales: history and status of six species listed as endangered under the US Endangered Species Act of 1973. Marine Fisheries Review 61(1): 1–74.

Pinela, A., Querouil, S., Magalhães, S., Silva, M., Prieto, R., Matos, J., and Santos R. 2009. Population genetics and social organization of the sperm whale (*Physeter macrocephalus*) in the Azores inferred by microsatellite analyses. Canadian Journal of Zoology 87: 802–813.

Prieto, R., Silva, M. A., Magalhães, S., Cabecinhas, R. and Santos, R. S. 2000. Desenvolvimento do 'Whale Watching' nos Açores. Departamento de Oceanografia e Pescas. Universidade dos Açores. Arquivos do DOP, Série Relatórios Internos N.º1/2000. 21 pp.

Prieto, R., and Fernandes, M. 2007. Revision of the occurrence of the long-finned pilot whale *Globicephala melas* (Traill, 1809), in the Azores. Arquipélago - Life and Marine Sciences 24: 65–69.

Quérouil, S., Silva, M. A., Freitas, L., Prieto, R., Magalhães, S., Dinis, A., Alves, F., Matos, J., Mendonça, A.D., and Hammond P. S. 2007. High gene flow in oceanic bottlenose dolphins (*Tursiops truncatus*) of the North Atlantic. Conservation Genetics 8: 1405–1419.

Quérouil, S., Silva, M. A., Cascão, I., Magalhães, S., Seabra, M. I., Machete, M. A., and Santos R. S. 2008. Why do dolphins form mixed-species associations in the Azores? Ethology 114: 1183–1194.

Quérouil, S., Freitas, L., Cascão, I., Alves, F., Dinis, A., Almeida, J. R., Prieto, R., Borras, S., Matos, J. A., Mendonça, D., and Santos, R.S. submitted. Molecular insight on the population structure of common and spotted dolphins inhabiting the pelagic waters of the North-east Atlantic. Marine Biology.

Quérouil, S, Seabra M. I., Silva, M. A., Magalhães, S., Prieto, R., Machete, M. A., Lafon, V., Santos, R. S. in prep. Spatio-temporal niche segregation between two dolphin species (*Stenella frontalis* and *Delphis delphis*) around the Azores.

Reiner, F., Gonçalves, J., and Santos. R. 1993. Two new records of Ziphiidae (Cetacea) for the Azores with an updated checklist of cetacean species. Arquipélago. Life and Marine Sciences 11A: 113–118.

Seabra, M.I., Silva, M.A., Magalhães, M., Prieto, R., and Santos, R.S. 2005. Desenvolvimento de um SIG para analisar a influência das características do habitat na ecologia dos cetáceos. Relatório da Bolsa de Investigação realizada no âmbito do Projecto CETAMARH (FCT-POCTI/BSE/38991/2001). Arquivos do DOP, Série Estudos, nº 5/2005: 48 p.

Silva, M.A. 2007. Population biology of bottlenose dolphins in the Azores Archipelago. PhD thesis, University of St Andrews, 266p.

Silva, M. A., Magalhães, S., Prieto, R., Cruz, A., Cabecinhas, R., Gonçalves, J., and Santos, R.S. 2001. Desenvolvimento do 'Whale-watching' nos Açores. Departamento de Oceanografia e Pescas. Universidade dos Açores. Arquivos do DOP, Série Relatórios Internos N.º9/2001. 27 pp.

Silva, M.A., Feio, R., Prieto, R., Gonçalves, J. M., and Santos. R. S. 2002. Interactions between cetaceans and the tuna fishery in the Azores. Marine Mammal Science 18: 893–901.

Silva, M., Prieto, R., Magalhães, S., Cabecinhas, R., Cruz, A., Gonçalves, J., and Santos, R. 2003. Occurrence and distribution of cetaceans in the waters around the Azores (Portugal), Summer and Autumn 1999–2000. Aquatic Mammals 29: 77–83.

Silva, M.A., Magalhães, S., Prieto, R., Querouil, S., Pinela, A., Seabra, M.I., and Santos, R.S. 2005. Ecologia e estrutura populacional dos roazes e cachalotes nos Açores: relação com as características do habitat. Relatório Final do Projecto CETAMARH (FCT-POCTI/BSE/38991/2001). Arquivos do DOP, Série Estudos, nº 4/2005: ii+40 p.

Silva, M. A., Prieto, R., Magalhaes, S., Seabra, M. I., Santos, R. S., and Hammond, P. S. 2008. Ranging patterns of bottlenose dolphins living in oceanic waters: implications for population structure. Marine Biology 156:179–192.

Silva, M.A., Magalhães, S., Prieto, R., Ricardo, R.S, and Hammond, P.S. 2009a. Estimating survival and abundance in a bottlenose dolphin population taking into account transience and temporary emigration. Marine Ecology Progress Series, 392: 263–276.

Silva, M. A., Brito,C., Santos, S., and Barreiros, J. P. 2009b. Historic and recent occurrences of pinnipeds in the Archipelago of the Azores. Mammalia 73(1): 60–62.

Silva, M.A., Feio, R., Prieto, R., Gonçalves, J.M., and Santos, R.S. submitted. Interactions between cetaceans and the tuna-fishery in the Azores Peer Aquatic Conservation: Marine and Freshwater Ecosystems.

Silva, M.A., Steiner, L., Cascão, I., Cruz, M.J., Prieto, R., Cole, T., Hamilton, P.K. and Baumgartner, M. in prep. Winter sighting of a western North Atlantic right whale in the Azores.

Simas, E.M., Herbert, B., Thompson, W., and Azevedo, J.H. (editors). 1999. New cetacean observations for the Azores. European Cetacean Society 13: 78.

Steiner, L. 1995. Rough-toothed dolphin, *Steno bredanensis*: a new species record for the Azores, with some notes on behaviour. Arquipélago. Life and Marine Sciences 13A: 125–127.

Steiner, L., Silva, M. A., Zereba, J., and Leal. M.J. 2007. Bryde's whales, *Balaenoptera edeni*, observed in the Azores: a new species record for the region. Marine Biodiversity Records 1: e66.

Steiner, L., Lamoni, L., Acousta Planta, M., Lewis, T., Beer, C., Gordon, J., Pettesson, L., Domingo, M., Lettevall, E., and Jensen, S.-K. 2009. Long distance movements of male sperm whales, *Physeter macrocephalus*, in the North Atlantic: including new matches between the Azores and Norway. European Cetacean Society.

Visser, F., Hartman, K., Rood, E., Hendriks, A., Zuult D., Wolff, W., Huisman, J., and Pierce, G. (in press) Risso's dolphin alters daily resting pattern in response to whale watching at the Azores. Marine Mammal Science.

Walton, M., Silva, M., Magalhaes, S., Prieto, R., and Santos, R. 2007. Using blubber biopsies to provide ecological information about bottlenose dolphins (*Tursiops truncatus*) around the Azores. Journal of the Marine Biological Association of the UK 87:223–230.

Walton, M., Silva, M., Magalhães, S., Prieto, R., and Santos, R. 2008. Fatty acid characterization of lipid fractions from blubber biopsies of sperm whales *Physeter macrocephalus* located around the Azores. Journal of the Marine Biological Association of the UK 88: 1109–1115.

Weilermann, D., Hartman, K., Cosentino, A., Garcia, L.G., and Visser, F. (editors). 2010. First ID- catalogue of Northern bottlenose whales (*Hyperoodon ampullatus*) off Pico island, Azores. European Cetacean Society. 35 p.

Wenzel, F.W., Allen, J., Berrow, S., Hazevoet, C.J., Jann, B., Seton, R.E., Steiner, L., Stevick, P., Suárez, P.L., and Whooley, P. 2009 Current knowledge on the distribution and relative abundance of humpback whales (*Megaptera novaeangliae*) off the Cape Verde Islands, Eastern North Atlantic. Aquatic Mammals 35(4): 502–510.

Whitehead, H. 2001. Analysis of animal movement using opportunistic individual identifications: application to sperm whales. Ecology 92: 1417–1432.

9 ToR f. Review of the scope, objectives and technical issues of the initiative for a European Marine Mammal Tissue Bank

9.1 Introduction

Since 1990, marine mammals stranded on the Belgian coast and bycaught animals are systematically necropsied. During these post-mortem examinations various tissue samples are collected and preserved. They form the **Belgian Marine Mammal Bio-bank (BMMB)**. Since 1995, through collaborations with researchers in France and the Netherlands, the geographical coverage of the collection extends to the entire continental coastline of the southern North Sea. Some samples from Ireland are also included in the BMMB. At the end of 2009, more than 23 000 samples were accumulated by the BMMB.

The tissues are collected, fixed and stored following standard protocols and the collection is kept at the Royal Belgian Institute of Natural History and the University of Liege (Belgium). Samples from the following species are currently available at the BMMB; *small cetaceans*: harbour porpoise, white-beaked dolphin, white-sided dolphin, bottlenose dolphin and striped dolphin; *large cetaceans*: sperm whale, humpback whale, minke whale, fin whale and Sowerby's beaked whale; and *pinnipeds*: harbour seal, grey seal and hooded seal. Samples are preserved by different methods: formalin-fixed tissue, formalin-fixed paraffin embedded tissue, frozen tissues (-20°C) and ethanol-fixed tissues. For each sample from a stranded or bycaught animal, the following documentation is available: species, age, sex, date and place of the stranding or incidental capture, morphometric data, conservation code, and results of post-mortem investigations (relevant lesions, cause of death, etc.).

As one of the goals of the BMMB is to make samples available for research at national and international levels, a specific database accessible through a web portal was designed and developed to facilitate data and tissue exchange. These samples are available for research purposes only and specific access rules have been designed in order to promote non-profit scientific collaboration.

For the database design, the data flow has been identified in its broadest sense by also taking into account the information on living animals, i.e. sightings data. A group management system and an authentication system have been developed to secure the web application. In addition, a collaboration agreement form has been established and should be accepted bilaterally before access is given to data or samples. The web interface presents a flexible tool for incrementally searching the recorded information. It will be possible to sort the samples and research data, filtering by species, organ, conservation code, place and date of stranding and also by relevant lesion and cause of death.

The resulting integrated information system replaces several, more or less independent, databases, datasets and paper archives. It also allows various views of the stored information; including public access to marine mammal stranding and sighting records, and a more detailed view to registered users for scientific use on individual animals with their associated tissue samples. The setup permits an easy incorporation of observations or tissue samples at any location. The system, developed and managed at the Belgian Marine Data Centre, will be available at www.marinemammals.be from June 2010 onwards.

Up to now within the North-east Atlantic, no coordination and centralization of post-mortem investigation results has been developed and even if samples are available in different countries and institutes, there is no catalogue to identify and locate them.

9.2 Scope

In Europe, there are numerous different stranding networks in charge of post-mortem investigations. In general, the procedures of necropsy and sampling are very similar and based on published protocols (Kuiken and García Hartmann, 1991; Geraci and Lounsbury, 2005). However there are no common procedures for sharing information and/or samples. Since 1990, for some countries bordering the North Sea, post-mortem investigations are comparable and have included investigations into parasitology, virology, bacteriology, histology, toxicology, etc. To date, the results from these post-mortem investigations, and information on samples collected during necropsy, have been documented and stored in different ways; e.g. paper archives and reports, publications, datasets or databases.

There are several marine mammal tissue banks in Europe accessible via a webportal: the BMMB discussed before, the Mediterranean Marine Mammal Tissue Bank (http://www.mammiferimarini.sperivet.unipd.it/eng/index.php), the BMA marine environmental tissue bank (http://www2.ub.edu/BMAtissuebank/home.htm) and the Irish Cetacean Genetic Tissue Bank (http://www.iwdg.ie/tissuebank/).

The Mediterranean Marine Mammal Tissue Bank was created in 2002 under the auspices of ACCOBAMS. It collects and preserves biological material sampled from marine mammals stranded along the Italian coast of the Mediterranean Sea, for gathering information on biology, genetics, anatomy, physiology, pathology and ethology.

The BMA was created in 2001 by the Faculty of Biology at the University of Barcelona, and the collection is composed mostly of endangered marine vertebrate samples from the Mediterranean Sea and neighboring areas; but also some samples from South America, Africa and Asia. Samples originate from a variety of sources: stranded specimens, commercial fishing, and bycatches in fishing operations or biopsies collected from free-ranging individuals. Tissues are preserved dry or frozen at -20ºC or at -80ºC. No samples are preserved in formalin, or embedded in paraffin wax.

The aim of the Irish Cetacean Genetic Tissue Bank is to establish a collection protocol and storage facility for tissues used in genetic analysis collected from cetaceans stranded, bycaught, or otherwise sampled around the Irish coast or within Irish waters.

Within Europe in general, as there is a gap in the coordination and centralization of post-mortem investigations and available samples, an initiative for a **European Marine Mammal Tissue Bank (EMMTB)** extending at least to whole the North-east Atlantic, can help improve knowledge of cetacean and seal biology at large. Samples can be used for investigations on anatomy, genetics, physiology, life history but also for pathology, parasitology, virology, bacteriology, histology, toxicology, etc.; all which will improve the evaluation of marine mammal health status in European waters. EMMTB can be considered as a common catalogue of available data and material. It will improve the use of samples and will allow a more general overview of the North-east Atlantic situation; including also geographical comparisons and temporal trends. A EMMTB can promote research on pathogens (Morbillivirus, Brucella) and contaminants of concern. A EMMTB will also help identify reference laboratories or institutions for specific investigations.

The EMMTB can also include previous analysed results to avoid duplication of investigations and ensure traceability of performed analyses. Last but not least, a EMMTB can provide some quality assurance/quality control to data and sample collection and preservation.

The present initiative can be considered as complementary to other existing initiatives (i.e. ASCOBANS and ICES WGMME 2010 ToR C: monitoring framework, see Section 5).

9.3 Objectives

The main objectives of the EMMTB will be:

- to identify laboratories and institutions involved in the post-mortem investigation (full necropsy and tissue sampling) of marine mammals in the North-east Atlantic;
- to identify laboratories and institutions qualifying as reference for specific investigations;
- to identify the location of samples in different laboratories and institutions in Europe;
- to create a web-portal catalogue of available samples from North-east Atlantic marine mammals, including formalin-fixed tissues, formalin-fixed paraffin embedded tissues, frozen tissues (-20°C and –80°C) and ethanol-fixed tissues;
- to provide samples that are adequately collected and preserved (Quality Assurance/Quality Control concept) for genetics, anatomy, physiology, life history, pathology, parasitology, virology, bacteriology, histology, toxicology, etc;
- to create a common collaboration agreement form between the EMMTB and scientific users of samples and data, with a possible embargo period on selected samples;
- to provide samples for research purposes only with specific access rules (detailed in the agreement form) in order to promote non-profit scientific collaboration;
- to create samples of reference for lesions and pathogens (parasites, virus, bacteria, etc.);
- to manage sample loans and data exchanges under the guidance of a steering committee composed of national coordinators of post-mortem investigations.

9.4 Technical issues

Establish a collaboration protocol between Belgium (T. Jauniaux, Dept of Veterinary pathology, Liege University), France (V. Ridoux, Centre de Recherche sur les mammifères marins), Germany (U. Siebert, Research and Technology Center Westcoast Christian-Albrechts-University Kiel), Ireland (E. Rogan, Cork University), UK (P. Jepson, Zoological Society of London), the Netherlands (A. Groene, Dept. of Veterinary Pathology, Utrecht University and M. Leopold, IMARES Texel) and other institutions or laboratories in charge of post-mortem investigations. The aim of such a protocol will be to select samples to share (following species, age, sex, cause of death, etc.) and decide how to share them (agreement).

Create a European Marine Mammal Tissue Bank website, similar to the BMMB website. Information concerning samples will remain stored in the existing national and local websites but links will be created between them and the EMMRB website (i.e. a meta-database with links and establishment of a common structure). The suggested website can be named: www.marinemammals.eu.

Further work is needed to define a strategy to fulfil the concept of quality assurance/quality control, and also to prevent duplication of investigations and to protect intellectual property of investigations. Also it is necessary to review and update necropsy and sampling procedures, in order to standardize methodologies for facilitating data comparisons.

9.5 Recommendations

1) Update marine mammal necropsy and sampling procedures, and standardize post-mortem procedures through international workshops.

2) Develop and promote the European Marine Mammal Tissue Bank and recognize its relevance in different fora.

3) As part of the EMMTB (a) identify laboratories and institutions involved in the post-mortem investigation (full necropsy and tissue sampling) of marine mammals in the North-east Atlantic, (b) collate information on the availability and location of samples, (c) develop bilateral collaborations between laboratories and institutes to fulfil the objectives of a tissue bank, including the establishment of a steering committee to manage sample loans and data exchanges, and (d) develop a website and meta-database for the EMMTB, with links to national websites and databases.

4) In future, the exchange of data and samples should be extended to include countries outside the North-east Atlantic, and also outside the ICES area.

9.6 References

Kuiken, T., and Garcia Hartmann. M. 1991. Proceedings of the first European Cetacean Society workshop on cetacean pathology: dissection techniques and tissue sampling. ECS Newsletter special issue 17.

Geraci, J. R., and Lounsbury. V. J. 2005. Marine mammals ashore: a field guide for strandings. National aquarium, Baltimore.

10 ToR g. Update on development of database for seals, and report on the status of any intersessional work

10.1 Requirement

To collate information from seal population monitoring programmes across the ICES Area and to populate a database so details on harbour and grey seal populations in different regions/countries can be more easily compared.

10.1.1 Security

The longevity of this seal database is entirely dependent on the frequency and extent to which it is populated with information from different countries. Many organizations that monitor seal populations are, understandably, very protective of their data, as it takes a lot of time, expense and effort to collect and collate. It is imperative that the database remains secure and that its contents are not accessible by anyone without the consent of the contributors. Some data are available annually on the Internet (e.g. Wadden Sea Trilateral Seal Expert Group http://www.waddensea-secretariat.org/QSR-2009/20-Marine-Mammals-(10-03-05).pdf; UK Special Committee on Seals http://www.smru.st-and.ac.uk/pageset.aspx?psr=411 for annual reports).

10.2 Area and species of relevance

The area covered is the North-east Atlantic and the North Sea, where the European species of the harbour (common) seal, *Phoca vitulina vitulina*, and the Atlantic grey seal, *Halichoerus grypus*, are found. Countries participating include: Norway, Sweden, Denmark, Germany, the Netherlands, Belgium, France, UK, and Ireland. In future, the area covered will extend to include the Baltic Sea in collaboration with the HELCOM Expert Group on Seals (i.e. to include the Baltic estimates of harbour and grey seals from Sweden, Finland, Russia, Estonia, Latvia, Lithuania and Poland and Russia), and also possibly the Faroe Islands, the Barents Sea (Russia) and the North-west Atlantic (Iceland, Greenland, Canada and the USA). For all areas, the database will be restricted to information on harbour and grey seals.

To date, Denmark, Germany, the Netherlands, Belgium, Ireland, Sweden, Norway and the UK have provided data. Data from France have been requested but may be problematic as they are collected independently by a number of different organizations and are not collated by one Governmental authority. Scientists in La Rochelle are attempting to collate the relevant information. Although France supports small populations of harbour and grey seals, both species are at the southern limit of their range so population information from this area is of particular interest.

10.3 Available data and survey methods

There is no standard survey methodology in use across all areas or for either species, although there are similarities, and different components of local populations of each species may be monitored in different areas. In many areas, surveys are carried out from either aircraft or helicopter, for instance. There is also variation in survey frequency in different countries. Survey frequency and intensity varies according to the degree of importance of each species in any country, the extent of coastline inhabited by seals, the complexity of that coastline and the substratum on which seals are normally found. Areas where seals are surveyed relatively easily are, unsurprisingly, surveyed more frequently.

There is also variation in reporting the results of surveys. Some are reported annually on publically accessible Internet sites (e.g. Wadden Sea Trilateral Seal Expert Group; Sea Mammal Research Unit for UK seals).

10.3.1 Harbour seals

Harbour seal surveys are carried out either during their summer breeding season or, some weeks later, during their annual moult. Both surveys report the minimum size of the local population. The Wadden Sea Trilateral Seal Expert Group, which collates the results of surveys in the Wadden Sea (the Netherlands to west Denmark), reported the maximum count for either of these periods as the count for the year between 1989 and 2002. Elsewhere, and in the Wadden Sea since 2003, surveys generally report the maximum counts for each season, breeding and moult, separately. Pups are counted during both the breeding season (June–July) and the moult (August) surveys. In the Wadden Sea, reported adult counts are from August surveys.

In the Netherlands, Germany, Denmark, southern Norway and southern Sweden, surveys are coordinated and normally occur on the same date in all areas. Three surveys are carried out in both the breeding season and the moult every year. Numbers reported are the trimmed mean i.e. the mean of the two highest counts from each period.

In the UK, harbour seals are not surveyed outwith their summer breeding and moulting seasons so information on their numbers and distribution outwith summer is very limited. Breeding and moult surveys are undertaken annually in the UK, though breeding season surveys are limited to two areas. The main population estimating surveys are carried out in August (moult) and only limited numbers of pups are seen during these surveys; they do not appear to be representative of the numbers that should be born. Harbour seals in east England (the Wash) are surveyed annually; once in the breeding season and twice in the moult. In Scotland, annual surveys are conducted in the Moray Firth (breeding and moulting seasons) and in the Firth of Tay (moult only). Elsewhere in Scotland, surveys are generally restricted to the moult and are repeated at approximately five-yearly intervals unless in response to specific requests for local information. In Northern Ireland, surveys (ground of boat counts) are carried out monthly throughout the year in a number of areas (e.g. Strangford Lough).

10.3.2 Grey seals

Data are mostly estimates of the numbers of pups born (pup production) in different areas. Information on grey seals is straightforward (if expensive) to collect in some areas and very difficult in others (e.g. SW England and Wales where most seals breed in caves or at the foot of cliffs). Relatively small numbers of pups are born in the Netherlands and these are widely dispersed over sandbanks and many are rescued into rehabilitation centres. On account of these anomalies, the Netherlands conducts additional aerial surveys during the grey seal moult, in April.

There are difficulties in inferring total populations size from pup production in areas where pup production is not increasing exponentially and is therefore subject to density-dependent effects. Some countries use a simple multiplier (e.g. Norway uses 4.0 and 4.7 to provide limits) others (e.g. UK) use more sophisticated models but these are limited by old and/or insufficient information on life-history parameters.

10.4 Database structure

To date, the current seal population database format is a simple MS Excel workbook. The database will be retained by the ICES database manager and updated annually as new information becomes available. There will be separate worksheets for the following:

- Harbour seal metadata;
- Regional harbour seal moult counts;
- Regional harbour seal pup counts;
- Regional harbour seal breeding counts;
- Overview of aggregated harbour seal data;
- Grey seal metadata;
- Regional grey seal pup production estimates;
- Regional grey seal moult/summer counts;
- Overview of aggregated grey seal pup production estimates;
- Overview of aggregated grey seal moult/summer counts

10.4.1 Harbour seal metadata

Very similar to grey seal metadata. Includes for each country: contact individual(s), e-mail address(es), Institute(s) and address(es), parameter(s) surveyed, year(s) of survey, frequency of survey, details of the methods used, area covered, comments. More detailed explanation of methods used during surveys including any limitations imposed to account for environmental factors e.g. numbers of hours from the time of low tide when surveys can be carried out; any other methods to minimize the effect of environmental variables. Describes the window of opportunity over which surveys are carried out for both breeding season and moult.

10.4.2 Harbour seal moult surveys

Contains the results of surveys carried out during the harbour seal annual moult. Numbers listed are the numbers of seals counted during the survey; they do not represent total populations size.

10.4.3 Harbour seal breeding surveys-pups

As above, but reporting numbers of pups counted during surveys. Includes information on whether the data represent pup counts (i.e. maximum number of pups counted), or whether counts are converted into an estimate of pup production.

10.4.4 Harbour seal breeding surveys-adults

Numbers of adults counted on surveys carried out during the breeding season. In some areas (Wadden Sea, UK Moray Firth) breeding season surveys are carried out annually. As with moult surveys, numbers presented are the number of seals counted; they do not represent an estimate of total population size.

10.4.5 Grey seal metadata

This worksheet contains information on:

The country, contact individual(s), e-mail address(es), Institute(s) and address(es), parameter(s) surveyed, year(s) of survey, frequency of survey, details of the methods used, the area covered, comments, indication whether pup production estimates are converted to total population size.

10.4.6 Grey seal pup production estimates

This worksheet contains the results of the grey seal pup production monitoring programmes. The data are organized by country, location within the country, ICES area, OSPAR area, whether an OSPAR EcoQO area. Data for each area is arranged by year of survey. Data from the UK represent estimates of total pup production for each area, either derived from direct ground counting or modelled from a series of counts through the breeding season.

10.4.7 Grey seal moult surveys

Some countries also monitor grey seal numbers during their moult between December and April e.g. Wadden Sea Trilateral Group (annual surveys) and the Republic of Ireland (one survey in 2007).

11 Future work and recommendations

11.1 Future work of the 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.

A list of the following recommendations can also be found at Annex 7 of this document.

Recommendation I

With regard to wind farm developments, establishment of means for efficient dissemination of results of common interest and means of making previous EIA reports and previously collected baseline data available for subsequent studies and assessments.

Recommendation II

Encourage multinational studies and encourage management decisions regarding offshore wind farms to be based on appropriate populations and/or management units for the relevant marine mammal species, irrespective of national borders.

Recommendation III

As the development of offshore wind farms extends further offshore and into new waters, monitoring should be extended to include all commonly occurring marine mammal species and marine mammal species of particular concern.

Recommendation IV

Geographical location of offshore wind farms should consider the distribution of marine mammals throughout the year, time of day and under typical weather and hydrographical conditions.

Recommendation V

Increase efforts to develop common measurement standards for both noise and marine mammal abundance.

Recommendation VI

Increase the effort to characterize sources of underwater noise related to the construction and operation of offshore wind farms. As part of this, common standards for measurement and characterization of underwater noise should be developed.

Recommendation VII

Develop methods to assess cumulative effects on marine mammals of the underwater noise level caused by the simultaneous construction and operation at nearby sites.

Recommendation VIII

Step up research on the behaviour of marine mammals as a consequence of increased underwater noise levels, in particular on how changes ultimately affect population parameters.

Recommendation IX

Increase efforts to characterize fundamental properties of the auditory system of marine mammals and the way noise affects physiology and behaviour.

Recommendation X

With regard to marine mammals to work towards common accepted tolerance limits for acute noise exposure and the development of common guidelines for mitigation in relation to pile driving.

Recommendation XI

To undertake studies to develop better marine mammal acoustic deterrent devices, including realistic trials in the field to demonstrate their effectiveness.

Recommendation XII

Attention should be given to improve efficient means of real-time detection of marine mammals during pile driving operations.

Recommendation XIII

Undertake other measures to prevent the exposure of marine mammals to high levels of underwater noise. This includes limiting the radiated energy during pile driving and the development of alternative methods for installation.

Recommendation XIV

Research should be continued and expanded to assess trends in contaminant exposure (PCBs and newer contaminants), population structure and health and reproductive effects in marine mammal species of highest risk (e.g. killer whales, St Lawrence belugas, polar bears, bottlenose dolphins, and Baltic marine mammals). The use of biopsy techniques would allow for simultaneous sampling for genetics and contaminant exposure.

Recommendation XV

In order to better detect future contaminant-related population level effects, there is a need for more robust population estimates for some marine mammal populations with low abundance and high pollutant (esp. PCB) exposure (e.g. killer whales and bottlenose dolphins).

Recommendation XVI

Adoption of an adaptive monitoring and surveillance framework for marine mammals under which objectives, monitoring (including surveys, strandings and observer bycatch programmes) and outcomes are regularly reviewed and updated by a Steering Group composed of representatives from all relevant bodies. While adaptive monitoring has the advantage that the monitoring programme can respond to changing requirements and constraints, the value of consistently collected long-term datasets should be taken into account. Further, this approach will improve the mechanisms for translating monitoring findings into appropriate management action for marine mammals.

Recommendation XVII

Adoption of a coordinated international approach to developing a single assessment for each marine mammal species at an appropriate biological scale when such assessments are required (e.g. the FCS reporting at six yearly intervals).

Recommendation XVIII

To further facilitate international coordination of monitoring, we recommend creation of ICES area/Europe-wide networks (e.g. for strandings, sightings, bycatch monitoring) and common databases and sample banks such as the European Marine Mammal Tissue Bank, and under which the unit of monitoring will be the natural population or (minimally) broad-scale spatial divisions that take into account the transboundary nature of most marine mammal populations (see also recommendations 26–27 below).

Recommendation XIX

The WGMME again strongly recommends immediate action by the Spanish and Portuguese governments in monitoring and conserving the Iberian harbour porpoise population.

Recommendation XX

Based on the newly described harbour porpoise Management Units, the WGMME recommends to ASCOBANS the establishment of a separate conservation plan for the harbour porpoise Inner Danish Waters MU.

Recommendation XXI

The WGMME also recommends to ASCOBANS to take into account the existence of the two newly designated harbour porpoise Management Units in the North Sea, Northeastern North Sea and Skagerrak and Southwestern North Sea and Eastern Channel, within their harbour porpoise North Sea conservation plan; with the inclusion of the Shetland Islands, Skagerrak and northern Kattegat within the Northeastern North Sea MU.

Recommendation XXII

The WGMME agrees with the actions of the Finnish Government, and recommends a ban on fishing within the area 15.4–30.6 in Lake Saimaa from mid April to mid June.

Recommendation XXIII

ICES to encourage a move away from implicit and automated conservation targets for marine mammals and towards the explicit definition and justification of target population sizes and management objectives.

Recommendation XXIV

Conduct systematic cetacean surveys of the Azores archipelago every 3 to 5 years; implement survey design established in Faustino *et al*.

Recommendation XXV

Existing Azorean observer bycatch monitoring programmes should be expanded to increase observation effort of some Azorean fisheries (e.g. demersal) and allow monitoring of other fisheries (e.g. the swordfish fishery).Further, bycatch monitoring of the

European deep-water longline fleet that fish in the outer 100 nm of the Azores archipelago needs to be implemented.

Recommendation XXVI

Ensure compliance of existing whale-watching regulations in the Azores archipelago through the establishment of an efficient law-enforcement scheme.

Recommendation XXVII

Develop and promote the European Marine Mammal Tissue Bank and recognize its relevance in different fora.

Recommendation XXVII

As part of the European Marine Mammal Tissue Bank; (a) identify laboratories and institutions involved in the post-mortem investigation (full necropsy and tissue sampling) of marine mammals in the North-east Atlantic, (b) collate information on the availability and location of samples, (c) develop bilateral collaborations between laboratories and institutes to fulfil the objectives of a tissue bank, including the establishment of a steering committee to manage sample loans and data exchanges, and (d) develop a website and meta-database for the EMMTB, with links to national websites and databases.

Annex 1: Legislation relating to monitoring and surveillance

1. European Legislation

Bern Convention and the Habitats Directive

The Convention on the Conservation of European Wildlife and Natural Habitats (or the Bern Convention) provides certain marine mammals with strict protection, while for others exploitation is allowed so long as their population numbers are not put in danger. For Member States of the European Community, the provisions of the Bern Convention are largely taken up in the 1992 Directive on the Conservation of Natural Habitats and of Wild Fauna and Flora (92/43/EEC), otherwise known as the 'Habitats Directive'.

Article 11 of the Habitats Directive requires that "*Member States shall undertake surveillance of the conservation status of the natural habitats and species referred to in Article 2 with particular regard to priority natural habitat types and priority species.*" This includes all species of cetacean and pinniped occurring in European waters. Additionally, Article 12 also requires that "*Member States shall establish a system to monitor the incidental capture and killing of the animal species listed in Annex IV (a) [which includes all cetaceans]. In the light of the information gathered, Member States shall take further research or conservation measures as required to ensure that incidental capture and killing does not have a significant negative impact on the species concerned.*"

For species, the favourable conservation status (FCS) was defined as '*the sum of the influences acting on the species that may affect the long-term distribution and abundance of its populations*'. A species status could be considered favourable if:

i) population dynamics data indicate that the species is maintaining itself on a long-term basis as a viable component of its natural habitats;

ii) the natural range of the species is neither being reduced nor is likely to be reduced in the foreseeable future; and

iii) there is, and will probably continue to be, a sufficiently large habitat to maintain its populations on a long-term basis.

Assessment of FCS therefore requires consideration of range and population (including trends), habitat availability, the main pressures and threats to the species (including bycatch), and future prospects of the species.

CMS and ASCOBANS

The Convention on Migratory Species (CMS or Bonn convention) sets out general provisions for the protection and conservation of certain migratory marine mammals, and also operates as a framework for a range of more specific multilateral agreements dealing with cetaceans, e.g. the Agreement on the Conservation of Small Cetaceans in the Baltic, North East Atlantic, Irish and North Seas (ASCOBANS).ASCOBANS includes a concise Conservation and Management Plan (CMP) that outlines the conservation and management measures to be implemented by signatories. This states that research '*shall be conducted in order to (a) assess the status and seasonal movements of the populations and stocks concerned, (b) locate areas of special importance to their survival, and (c) identify present and potential threats to the different species.*' Besides these requirements to monitor abundance and distribution of small cetacean species, the CMP also states that '*each party shall endeavour to establish efficient system for reporting and retrieving bycatches and stranding specimens and to carry out … full autopsies in order to collect tissues for further studies and reveal possible causes of death and to document food composi-*

tion. The information shall be made available in an international database'. In addition, the CMP also states that '*Information shall be provided to the general public in order to ensure support for the aims of the agreement in general and to facilitate the reporting of sightings and strandings in particular; and to fishermen in order to facilitate and promote the reporting of bycatches and the delivery of dead specimens to the extent required for research under the agreement*.'

Marine Strategy Framework Directive and Good Environmental Status

The MSFD, formally adopted by the European Union in July 2008, requires Member States to develop marine strategies that apply '*an ecosystem-based approach to the management of human activities while enabling a sustainable use of marine goods and services, priority should be given to achieving or maintaining good environmental status in the Community's marine environment, to continuing its protection and preservation, and to preventing subsequent deterioration*'.

Each Member State is required to develop a marine strategy by 2012 that ensures '*integration of conservation objectives, management measures and monitoring and assessment activities*' with the conservation element focused on protected areas. It is expected that these strategies will be developed in coordination with other MS within the same marine region or subregion. For the North Atlantic region this is achieved through OSPAR (see below). The marine strategies must include '*an assessment of the current environmental status and the environmental impact of human activities thereon*' and the establishment '*of a series of environmental targets and associated indicators*'. OSPAR has stated that '*the Quality Status Report 2010, a comprehensive evaluation of the state of the environment of the North-East Atlantic, will provide an excellent basis to assist Member States with producing their initial assessment for national marine strategies required by the European Commission for 2012*' under the MSFD. By 2014, establishment and implementation of a monitoring programme for ongoing assessment and regular updating of targets is required.

Indicators of GES have yet to be determined. However, the first descriptor of GES is biodiversity. EC have recently indicated that this descriptor will, in part, require individual species assessments for which the 'three criteria for the assessment of any species are species distribution, population size and population condition.' It has been indicated that these monitoring requirements will not exceed those of the Habitats Directive. Annex III of the MSFD also identifies pressures such as physical disturbance through underwater noise, contamination by hazardous substances and biological disturbance such as bycatch that need to be included within the national marine strategy. These are pertinent to all cetacean and pinniped species occurring in European waters.

The Convention for the Protection of the Marine Environment of the North-East Atlantic (OSPAR)

The Convention for the Protection of the Marine Environment of the Northeast Atlantic replaced both the Oslo and Paris Conventions, with the intention of providing a comprehensive and simplified approach to addressing issues associated with maritime pollution. Additionally, OSPAR also provides for the '*protection and conservation of the ecosystem and biological diversity of the maritime area*' in Annex IV and lays down 'criteria for identifying human activities for the purpose of Annex V' in Appendix 3. In 2004, OSPAR agreed a list of threatened and declining species that included the marine mammals: blue, bowhead and northern right whales and the harbour porpoise. For each of these species, a case document was prepared that outlines basic biology and ecology, trends in abundance and distribution, the IUCN status assessments, known threats to the species, existing management measures and actions to be undertaken by the relevant Contracting Parties. These documents were updated most recently in 2010.

As part of this revision, the harbour porpoise background document now includes a recommended list of monitoring requirements for the OSPAR area (Table 7). Initially, baseline monitoring is required that includes visual surveys of abundance and distribution, reporting strandings and bycatches. Additionally, acoustic surveys are required for areas known or suspected to host high densities of harbour porpoise or to be breeding, birthing, or rearing grounds. The monitoring should be enhanced when a population is considered to be endangered, or when a population has shown statistically significant declines.

This additional monitoring includes bycatch reporting on all vessels in fisheries known or suspected to have a porpoise bycatch; aerial surveys of national areas at least every three years, preferably every year (or increased sighting surveys in areas of known or suspected problems, semi-annually or quarterly as well as the use of passive acoustic monitoring); collection of tissue samples of bycaught and stranded animals and necropsies (post-mortem examinations) of a sample of these animals; this should include the examination of all organs including brain, the inner ear, analysis of pollutants in tissues, and immune function tests.

In addition to the list of threatened and/or declining species, OSPAR has also developed a number of Ecological Quality Objectives (EcoQOs) for the North Sea associated with marine mammals. These include one for harbour porpoise bycatch and two for seal population estimates. Specifically for bycatch, it was agreed at the fifth North Sea Conference in 2002 that an Ecological Quality Element relating to harbour porpoise bycatch in the North Sea would be given the objective: "*annual bycatch levels should be reduced to levels below 1.7% of the best population estimate*". OSPAR 2006 adopted the agreement on the application of the EcoQO system in the North Sea which required the first assessment of the application of the EcoQO system in 2008.

In 2008, the ICES Working Group on Marine Mammal Ecology tried to evaluate progress to date with this EcoQO on a North Sea wide basis (ICES, 2008b). It was quickly apparent that many of the fisheries suspected to have the highest bycatch levels are conducted without bycatch observer programmes as these are not a requirement of EU Regulation 812/2004. Consequently, it is not possible to evaluate whether or not the EcoQO has been met. Until such observer programmes are implemented it will not be possible to assess overall progress with this EcoQO.

The two seal EcoQOs are related to population size or pup production estimates. For harbour seals, there should be no decline in population size of >10% as represented in a five-year running mean. For grey seals there should be no decline in pup production of >10% as represented in a five-year running mean. In general, recruitment of grey seal pups in the North Sea has increased, but there have been dramatic declines in harbour seals numbers along the east coast of the UK (Lonergan *et al*., 2007). The EcoQO has thus probably been met for grey seals for all subunits of the North Sea population, while the harbour seal EcoQO has not been met in some areas. There have been disease outbreaks affecting seals in some other areas (Harkonen *et al*., 2008) but the cause of the observed declines in other areas are unknown (SCOS, 2008; 2009).

Council Regulation 812/2004

This regulation lays down measures concerning incidental catches of cetaceans in fisheries and also amends regulation (EC) No. 88/98.

The pertinent measures include:

- the coordinated monitoring of cetacean bycatch through compulsory on-board observers for given fisheries;
- the mandatory use of acoustic deterrent devices ('pingers') in certain fisheries.

EC Regulation 812/2004 requires that sampling should be geared to achieve a bycatch estimate with a coefficient of variation (CV) of less than 0.3. This can only be achieved if at least one bycatch event is observed. In the absence of any observed bycatch, and assuming continued monitoring is needed, Northridge and Thomas (2003) suggest the using the 'pilot study' levels of 10% and 5% for the various fishery segments as the most appropriate approach to setting monitoring requirement levels.

Note that the Common Fisheries Policy and Habitats Directive are in some respects contradictory and, because national governments have transferred competence for legislating on fishery issues, for all waters beyond the immediate coastal zone (outside 12 nautical miles), to the European Union, they now lack the legal competence to fully implement the Habitats Directive in instances where fishing is demonstrably adversely affecting the status of protected marine mammal species (Proelss, 2010).

North Atlantic Marine Mammal Commission (NAMMCO)

The NAMMCO Agreement was signed in 1992 by Norway, Iceland, Greenland and the Faroe Islands. It provides a mechanism for cooperation on conservation and management for all species of cetacean and pinniped in the region, providing scientific advice and conservation/management recommendations; including stock assessment, sustainable harvest levels, bycatch and marine mammal/fisheries interactions. The assessments cover stock structure, basic biology and ecology, distribution and abundance trends, current management including hunting takes, threats, status and outlook.

2. North American legislation for monitoring in the ICES area

Marine Mammal Protection Act

The US Marine Mammal Protection Act (MMPA) was enacted in 1972 because some marine mammal species or stocks were considered to be in danger of extinction or depletion as a result of human activities and marine mammals were proven to be resources of great international significance. All marine mammals are protected under the Act, which prohibits, with certain exceptions, the deliberate taking of marine mammals in US waters and by US citizens on the high seas, and the importation of marine mammals and their products. The MMPA was amended substantially in 1994.

Under the 1994 amendments of the MMPA, the National Marine Fisheries Service (NMFS) and the United States Fish and Wildlife Service (USFWS) were required to generate stock assessment reports (SAR) for all marine mammal stocks in waters within the US Exclusive Economic Zone (EEZ). Each SAR contains: (1) a description of the stock, including its geographic range; (2) a minimum population estimate, a maximum net productivity rate, and a description of current population trend, including a description of the information upon which these are based; (3) an estimate of the annual human-caused mortality and serious injury of the stock, and, for a strategic stock, other factors that may be causing a decline or impeding recovery of the stock, including effects on marine mammal habitat and prey; (4) a description of the commercial fisheries that interact with the stock, including the estimated number of vessels actively participating in the fishery and the level of incidental mortality and serious injury of the stock by each fishery on an annual basis; (5) a statement categorizing the stock as strategic or not, and why; and (6) an estimate of the potential biological removal (PBR) level for the stock, describing the information used to calculate it. The MMPA also requires that SARs be updated annually for stocks which are specified as strategic stocks, or for which significant new information is available, and once every three years for non-strategic stocks.

The Species at Risk Act (SARA).

The Canadian SARA was enacted in 2002. It is a key federal government commitment to prevent wildlife species from becoming extinct and secure the necessary actions for their recovery. It provides for the legal protection of wildlife species and the conservation of their biological diversity. It also manages species which are not yet threatened, but whose existence or habitats may be in jeopardy.

SARA designates the Committee on the Status of Endangered Wildlife in Canada (COSEWIC) to identify threatened species and assess their conservation status. With respect to the ICES area, the only pinniped covered is the North-west Atlantic population of walrus. The North Atlantic cetacean species currently listed are fin, blue, grey and northern right whales. Each status report contains information on the basic biology of the species, its distribution in Canada, population sizes and trends, habitat availability and trends, and threats to the species. Every ten years, or earlier, if warranted, COSEWIC reassesses the species designated in a category of risk with an update status report. As necessary, COSEWIC may also reassess other species previously found Not at Risk or Data deficient with an update status report.

3. Reference

Harkonen, T., Bäcklin, B.M., Barrett, T., Bergman, A., Corteyn, M., Dietz, R., Harding, K.C., Malmsten, J., Roos, A., and Teilmann, J. 2008. Mass mortality in harbour seals and harbour porpoises caused by an unknown pathogen. The Veterinary Record 162(17): 555–556.

Proelß, A. 2010. Protection of cetaceans under national and European law: Impact of the Habitats Directive on the Common Fisheries Policy. Invited talk, European Cetacean Society conference, Stralsund, Germany, March 2010.

SCOS. 2008. Scientific Advice on Matters Related to the Management of Seal Populations: 2008. Sea Mammal Research Unit. 98pp. http://www.smru.st-andrews.ac.uk/pageset.aspx?psr=411.

SCOS. 2009. Scientific Advice on Matters Related to the Management of Seal Populations: 2009. Sea Mammal Research Unit. 205pp. http://www.smru.st-andrews.ac.uk/pageset.aspx?psr=411.

Annex 2: Current monitoring schemes for cetaceans

Current monitoring of cetaceans: scope, time-period, who is doing it (G=government funded, V=voluntary sector, P=project funding, I=institutional funding). Aside from national or regional schemes, please also mention any important short-term or small-scale monitoring.

Country	Strandings	Sightings[1]	Fishery bycatch/directed catches[2]	Biological data[3] (diet, life history, pathology, contaminants)	Other[4]
Belgium	Part G, part I; all stranded animals are collected for research purposes (or for whales: investigated on the stranding site); good coverage of the coast due to short coastline, dense human population and easy access.	Monitoring density and distribution: aerial surveys; additional information from seabird surveys; Part G, part I, part P (offshore windfarms); Next to this also reporting of opportunistic sightings by the public through dedicated Internet sites set up by NGOs.	No directed catches; Obligation to report bycatch taken up in legislation (but follow-up by fishers weak); Fisheries cooperate on voluntary basis in projects assessing the impact of different fishing gears.	Part G, part I; research only on bycaught and stranded animals, no biopsies taken.	Assessment of distribution and density is carried out mainly in the framework of offshore windfarm projects: partly project (construction applicant) funded, partly I; dedicated monitoring through aerial surveys, passive acoustic monitoring (C-PoD).

1 Monitoring of abundance, distribution, movements, behaviour.

2 Where relevant.

3 From carcasses of stranded or bycaught animals or from biopsies.

4 For example, tagging and photo-ID studies, monitoring related to ship strikes, disturbance, naval exercises, seismic surveys, wind farms, etc.

Country	Strandings	Sightings[1]	Fishery bycatch/directed catches[2]	Biological data[3] (diet, life history, pathology, contaminants)	Other[4]
Canada	Network in place since early 1980s for St Lawrence Estuary (G). Full necropsies of beluga (I+G); recovery and necropsy more variable for other sp (G). No real structure for Gulf of St Lawrence or Nova Scotia. In Newfoundland, mainly through University and Gov. in earlier years, now mainly Gov + NGO	Regular monitoring for St Lawrence beluga (G); systematic surveys much less frequent for other species (G); network for opportunistic observations in Estuary and Gulf of St Lawrence and Newfoundland + Labrador operated by NGOs; more formal initiative to collect opportunistic sightings (G).	Monitoring programme in place via observers on board of vessels, with obligation to log n and sp of m. mammals (G). True for Estuary and Gulf of St Lawrence and Newfoundland/Labrador. Not sure for Maritimes if similar system exists (but yes for Bay of Fundy). All G	Pathology done fully on St Lawrence beluga (I+G), but not other species. Tissues for diet, life history and contaminants collected for St Lawrence beluga (G). Tissues not collected systematically for all other species. Depend on current projects (P+G+I).	- Photo ID programmes for St Lawrence beluga (I), right whales (I+G), the four rorquals (minke (I), blue (I), fin (I), humpback (I)), bottlenose whale (I), killer whales (G), and to a lesser extent sperm whales (I). - Tagging of beluga (G), narwhal (G), blue whales (G), fin whales (I). - Obligation to report collisions in the Saguenay St-Lawrence marine park (G). - Effect of noise and whale-watching on St Lawrence beluga and blue whales; similar study on humpbacks in Newfoundland

Country	Strandings	Sightings[1]	Fishery bycatch/directed catches[2]	Biological data[3] (diet, life history, pathology, contaminants)	Other[4]
Croatia	Since 1990 all stranded death animals are collected by the Faculty of Veterinary Medicine University of Zagreb for post-mortem examination (G + P). Since 1990 only one live stranding has been reported.	Reporting of opportunistic sightings by the public	Bycatches are reported to the Faculty of Veterinary Medicine University of Zagreb (G+P).	Postmortem examinations and cetacean tissue bank at the Faculty of Veterinary Medicine University of Zagreb (G+P).	Photo-ID project (P)

Country	Strandings	Sightings[1]	Fishery bycatch/directed catches[2]	Biological data[3] (diet, life history, pathology, contaminants)	Other[4]
Denmark	G: Denmark is divided into several districts that are obliged to report any marine mammal stranding (species, sex, size and location). An annual report is made in Danish.	G: In relation to bird surveys marine mammals are also recorded. From 2011 systematic annual surveys are planned in all harbour porpoises NATURA2000 sites. Every 5 years there will also be a large-scale survey in the Danish Straits and every 10 years hopefully a SCANS like survey. P: many small-scale surveys are conducted in relation to wind farms and bridge EIAs and constructions. V: Private initiative where sightings and strandings can be reported on www.hvaler.dk	G: No bycatch estimate is available from the North Sea (see Vinther and Larsen 2004[5]). No bycatch estimate exists for the Danish straits (Kattegat, Belt seas and the Baltic sea).	P: Diet and contaminants are regularly examined in various projects. Pathology and life history are currently not studied.	G/P: Satellite tagging and deployment of data loggers (depth, 3D movements and acoustic) of harbour porpoises have been carried out on almost 100 animals since 1997. P: Acoustic monitoring of harbour porpoises (PODs and towed array) have been carried out in relation to habitat areas, wind farms and ship routes to determine the presence of animals and the effect of human disturbance.
Finland					
France					

[5] Vinther, M., and Larsen, F. 2004. Updated estimates of harbour porpoise (*Phocoena phocoena*) bycatch in the Danish North Sea bottom-set gillnet fishery. J. Cetacean Res. Manag. 6(1): 19–24.

Country	Strandings	Sightings[1]	Fishery bycatch/directed catches[2]	Biological data[3] (diet, life history, pathology, contaminants)	Other[4]
Germany	G = monitoring system of dead and live stranded marine mammals funded by 3 German states (Lower Saxony, Schleswig-Holstein, Mecklenburg-Western Pomerania) P = resulting projects from stranding network I = more animals are examined than paid for.	G/P = several projects	G/P = bycatches and observations by fishermen in the Baltic (Schleswig-Holstein+Mecklenburg Western Pomerania); stranding network	G/P = stomach content: in the past; still collecting samples, but no funds at the moment Age determ. for porpoises Pathology: all stranded cetaceans Contaminants: in the past; currently done at international institutes.	
Iceland	G= Monitoring of live and dead strandings by the Marine Research Institute (MRI) and the Institute of Natural History (INH). The Research associated with strandings by the MRI.	G=Large scale surveys (NASS) conducted every 5-7 years. Sightings data collected on a small scale in several different projects (G, P, V).	G: Catches reported yearly. Bycatch reporting mandatory, but reporting efficiency low. Improvements are being investigated (MRI).	G: Most animals from the hunt are examined regarding life-history parameters, genetics, feeding ecology and energetics. Opportunistic studies on other aspects. Strandings sampled to the extent feasible.	Various research projects ongoing using techniques s.a. photo-id, acoustic recordings, behavioural observations and satellite tracking (G, P, V).
Ireland	(Irish Whale and Dolphin Strandings Scheme).	Marine mammal monitoring Broadhaven Bay, Mayo since 2002 (P).	No official bycatch reporting programme for marine mammals.		Acoustic monitoring of delphinids in Broadhaven Bay, Mayo since 2007 (P)

Country	Strandings	Sightings[1]	Fishery bycatch/directed catches[2]	Biological data[3] (diet, life history, pathology, contaminants)	Other[4]
Italy	CIBRA (University of Pavia) and the Museum of Natural History of Milan maintain the official national stranding database granted by the Italian Ministry of the Environment. A network to monitor cetacean stranding along the Italian coasts has been operating since 1986 by Centro Studi Cetacei, resulting in annual stranding reports as well as scientific outputt. CE.TU.S.: and Centro Ricerca Mammiferi Marini also monitor strandings.	Monitoring in Italy is split between a large number of organizations. Weekly monitoring of density and distribution (yearly or from May to September): network of systematic surveys with fixed transect approach in Tyrrhenian and Ligurian sea; additional information from seabird surveys in Tyrrhenian sea. Part G, part I, part V.			

Country	Strandings	Sightings[1]	Fishery bycatch/directed catches[2]	Biological data[3] (diet, life history, pathology, contaminants)	Other[4]
Netherlands	Collected by different organizations (V, G); animals are brought together at the University of Utrecht for further analyses.	Different projects: land-based observations, ESAS observations IMARES: aerial surveys, passive acoustic surveys, ship board surveys Others (see ASCOBANS report). Rugvin (V): monthly ferry surveys Hook of Holland (NL) to Harwich (UK). Rugvin (V): boat surveys plus acoustic monitoring Oosterschelde estuary (southwest Netherlands).	IMARES (G): short-term monitoring and pilot projects e.g. using camera systems to monitor bycatch (Bram Couperus).	University of Utrecht (G).	
Norway	Irregular, V	Every year, G	Catches yearly, G	Irregular, P	
Portugal			In northern Portugal, on-board, interview and voluntary reporting (V/P).		

Country	Strandings	Sightings[1]	Fishery bycatch/directed catches[2]	Biological data[3] (diet, life history, pathology, contaminants)	Other[4]
Azores (Portugal)	Azorean Cetacean Stranding Network, 1996–present, G/I	Department of Oceanography and Fisheries of the University of the Azores (DOP/UAç), 1999–present, P Whale-watching operators, 1993–present, private funding Nova Atlantis Foundation, 2004–present, I	Tuna-fisheries observer programme, DOP/UAç, 1998–present, G/P Short-term monitoring programmes of several fisheries, DOP/UAç, 1990–present, P	DOP/UAç, 1999–present, P	Photo-id, DOP/UAç, 1999–present, P Photo-id, whale-watching operators 2000–present, private funding Photo-id, Nova Atlantis Foundation, 2004–present, V Disturbance from Whale-watching, DOP/UAç, 1998–2006, P Disturbance from Whale-watching, Nova Atlantis Foundation, 2004, I Tagging, DOP/UAç, 2008–present, P Acoustic, DOP/UAç, 2007–present, P

Country	Strandings	Sightings[1]	Fishery bycatch/directed catches[2]	Biological data[3] (diet, life history, pathology, contaminants)	Other[4]
Spain	Patchy coverage e.g. Galicia: since 1990 (V), separate NGOs run strandings monitoring schemes on different islands in the Canaries (V).	Patchy coverage Large-scale: SCANS II, CODA (P); Small-scale (V) (e.g. in Galicia); long-term medium scale monitoring of abundance and distribution (northern Alborán Sea) (P) Long-term cetacean monitoring scheme off La Gomera (Canary Islands) conducted by the German NGO MEER e.V. Data collected opportunist-ically year-round from whale watching vessels since 1995.	Some on-board (G). In Galicia, on-board, interview and voluntary reporting (V/P)	In Galicia, age, maturity, diet data routinely collected and samples collected for other analyses (P/V).	- Photo-ID programe for northern Alborán Sea, long-term (P)
Sweden	Continuous reporting G	Irregular P.	Irregular P.	Pathology G, P	

Country	Strandings	Sightings[1]	Fishery bycatch/directed catches[2]	Biological data[3] (diet, life history, pathology, contaminants)	Other[4]
UK	I- Since 1913 Natural History Museum has collected ad hoc data on UK stranded cetaceans. G- Since 1990, centralized funding (by Defra and the Devolved Administrations) of the systematic recording and investigation of UK strandings by the Cetacean Strandings Investigation Programme (CSIP, www.ukstrandings.org). V- local collation of records by some voluntary schemes.	Large-scale (P) Small-scale (G, V, P),	On-board (G, P))	G- Detailed standardized necropsies routinely carried out by CSIP since 1990. Approximately 2800 have been conducted to date (19 species). Pathology, life history, diet etc. data are routinely collected. Contaminant data (PCBs, metals, brominated flame retardants, perfluorinated compounds) has been generated on several hundred harbour porpoises plus limited data on other species. A national cetacean tissue archive and a web-accessed strandings/pathology database is maintained by CSIP partner organizations. P- additional funding for some aspects of above.	

Country	Strandings	Sightings[1]	Fishery bycatch/directed catches[2]	Biological data[3] (diet, life history, pathology, contaminants)	Other[4]
USA	National Programme operated under NMFS Letter of Authorization and NMFS regional coordinators – Sampling protocol and data reporting are standardized, and annual regional stranding workshops are held. Tissue samples support a wide range of research, and also used to assist in management of anthropogenic impacts. Individual stranding organizations are partially funded by NMFS and private sources. Highest priority on east coast is necropsy of right whales. An important component of the programme is the disentanglement of large whales.	NMFS conducts year-round aerial surveys in support of North Atlantic right whale science and management. Center for Coastal Studies conducts a parallel survey in Cape Cod Bay in spring long term. NMFS conducts coast-wide (Virginia to Maine) summer cetacean survey (ship & aircraft) every 3 to 5 years. NMFS conducts aerial and shipboard summer harbour porpoise abundance survey in Gulf of Maine and western Scotian Shelf – every 3 to 5 years. NOTE- Recent funding by US Minerals Management Service will expand all survey effort to year-round and expand survey range from Florida Keys to Maine/Canadian border from the beach to the offshore EEZ.	NMFS conducts a coast-wide observer programme, sampling priorities are based on fisheries with known bycatch problems (e.g. northeast and mid-Atlantic sink gillnets; northeast trawl fishery; pelagic longline fishery; Atlantic herring purse-seine fishery.	Necropsies of fishery bycaught and high quality strandings; particularly right whales and mass stranding are routinely conducted. Tissue samples are collected for a wide range of ecological and health assessment studies. Various GOV organizations and Cornell University established a long-term right whale passive acoustic (T-POD) study on Stellwagen Bank and adjacent high-use habitats. NMFS and WHOI conduct large whale (primarily for right whale) ecological studies; annually in spring biannually in summer. All right whale focused studies collect photo-ID data which is submitted to the New England Aquarium.	

Annex 3: Current monitoring schemes for seals

COUNTRY	STRANDINGS	SIGHTINGS	BYCATCH/PREDATOR CONTROL	DIET	OTHER
Belgium	Similar as for cetaceans	Similar as for cetaceans, but no dedicated aerial surveys due to low density of seals in Belgian waters; only 1 – 3 haul out locations used during a part of the year by in total 5 to 15 animals (common seals). Irregular presence of grey seals.	No predator control; stranded pups are taken to rehabilitation centre and released thereafter;	Plans to investigate stomach content of stranded and bycaught animals in the near future.	
Canada	Recovery and necropsy variable and project related (P).	No	Monitoring programme in place via observers on board of vessels, with obligation to log n and sp of m. mammals (G). True for Estuary and Gulf of St Lawrence and Newfoundland/Labrador. Not sure for Maritimes if similar system exists. All G	Pathology not done. Tissues not collected systematically for all other species. Depend on specific projects (P+G+I).	- No PhotoID - Tagging harbour, grey, harp, hooded and ringed seals. - Effect of disturbance at haul-out sites in St Lawrence Estuary (G).

COUNTRY	STRANDINGS	SIGHTINGS	BYCATCH/PREDATOR CONTROL	DIET	OTHER
Denmark	G: Denmark is divided into several districts that are obliged to report any marine mammal stranding (species, sex, size and location). An annual report is made in Danish.	G: In relation to bird surveys, marine mammals are also recorded. All seal haulout sites are surveyed 1-3 times both during pupping and moulting. P: many small-scale surveys are conducted in relation to wind farms and bridge EIAs and constructions. V: Private initiative where sightings and strandings can be reported on www.hvaler.dk	G: No bycatch estimate is available from the North Sea. No bycatch estimate exists for the Danish straits (Kattegat, Belt seas and the Baltic sea). 10–20 harbour seals are shot every year in relation to conflict with fixed gear fishery.	P: Diet and contaminants are regularly examined in various projects. Pathology and life history are currently not studied.	P: Satellite tagging and deployment of data loggers (FastLoc GPS, depth, 3D movements) of harbour seals and grey seals have been conducted in various project in the past 10 years.
Finland					
France					
Germany	G = monitoring system of dead and live stranded marine mammals funded by three German states (Lower Saxony, Schleswig-Holstein, Mecklenburg-Western Pomerania). P = resulting projects from stranding network I = more animals are examined than paid for.	G/P = several projects	G/P = bycatches and observations by fishermen in the Baltic (Schleswig-Holstein+Mecklenburg Western Pomerania); stranding network.	G/P = stomach content: in the past; still collecting samples, but no funds at the moment Age determ. for seals Pathology: only well-preserved seals Contaminants: in the past; currently done at international institutes.	

Country	Strandings	Sightings	Bycatch/predator control	Diet	Other
Iceland		Regular (every 3–5 years) aerial coastal surveys of grey and common seals. Land based surveys at a local scale annually (G, P, V).	G: Catches reported yearly. Bycatch reporting mandatory, but reporting efficiency low. Improvements are being investigated (MRI).	Variable proportion of animals from the hunt are examined regarding life-history parameters and feeding ecology, Opportunistic studies on other aspects.	
Ireland		National census of harbour seal and grey seal (2003; 2005 respectively) (G); Regional/local counts at main seal colonies infrequently since 1980s (G, P);		Seal diet studies University College Cork infrequent since 1990s (P)	GPS/GSM tagging of both harbour seal and grey seal in southwest Ireland since 2006 (P)
Netherlands	Stranded animals from north/west autopsied parallel to Porpoise(I)- others in rescue center (I). Stranded seals from north/west autopsied at the Seal Rehabilitation and Research Center (SRRC), Pieterburen (I).	Monitoring Wadden Sea pup and moult counts coordinated with German and Danish counts (G). In south different groups survey monthly (G), pup and moult (G) other (V). Aerial monitoring of seals in the Wadden Sea, pup and moult counts by SRRC (I).	Relatively high occurrence of probable bycatch in southern Netherlands,(V,I collect data) no controle	Scat samples collected parallel to tracking efforts (I), stomachs from north/west collected from strandings (I). Other ?	Satellite telemetry in relation to contract research N~10–50/year (I). Obs in relation to disturbance.
Norway		Every 5 year, I	Estimates reference fleet, P/yearly, G	Irregular, P	Ecological seal studies, P
Portugal	Seal strandings monitored by Azorean Cetacean Stranding Network, 1996–present, G/I				
Spain					

Country	Strandings	Sightings	Bycatch/predator control	Diet	Other
Sweden	Approx 100 seals per year examined. G	Annual Surveys G	Irregular studies P	Irregular P	
UK	Started in Scotland; not in Eng, Wales??, yes in N Ireland. Irregular. Poor in 2002 PDV in UK.	Yes, UK wide Annual surveys some areas Approx every 5 years in Scotland	Some control of seals as predators Trying to get bycatch assessed	Sporadic, mostly very local studies. Grey diet in 1985 and 2002; 1st Scottish-wide harbour diet study started	Long-term studies at two grey seal colonies looking at female condition/breeding success/pup growth (Scot).
USA	National Programme operated under NMFS Letter of Authorization and NMFS regional coordinators – Sampling protocol and data reporting are standardized, and annual regional stranding workshops are held. Tissue samples support a wide range of research, and also used to assist in management of anthropogenic impacts. Individual stranding organizations are partially funded by NMFS and private sources.	NMFS conducts seasonal monitoring of harbour seal and grey seal haul-out sites off Massachusetts coast. Annual monitoring of three grey seal pupping colonies. NMFS conducts harbour seal abundance survey- every 5–8 years along coast of Maine during the pupping period. NOTE- Recent funding by US Minerals Management Service will expand all survey effort to year-round and expand survey range from Florida Keys to Maine/Canadian border from the beach to the offshore EEZ.	NMFS conducts a coast-wide observer programme, sampling priorities are based on fisheries with known bycatch problems (e.g. northeast and mid-Atlantic sink gillnets; northeast trawl fishery; pelagic longline fishery; Atlantic herring purse-seine fishery	Necropsies of fishery bycaught and high quality strandings – particularly right whales and mass stranding are routinely conducted. Tissue samples are collected for a wide range of ecological and health assessment studies. Intermittently, scat samples are collected at major Cape Cod haul-out sites	Small-scale behavioural study of major grey seal pupping colony initiated in 2009; University of New England.

Annex 4: List of participants

NAME	ADDRESS	PHONE/FAX	EMAIL
Sophie Brasseur by correspondence	IMARES Institute for Marine Resource & Ecosystem Studies Department of Ecosystems PO Box 167 1790 AD Den Burg The Netherlands	Phone: +31 Fax: +31	Sophie.Brasseur@wur.nl
Karsten Brensing	Whale and Dolphin Conservation Society in Germany Altostraße 43 D-81245 München Germany	Phone: +49 89 4581 9943 Fax: +49 89 6100 2394 Skype: karstenbre	karsten.brensing@wdcs.org
Malcolm Clarke Invited expert	Cachalotes e Lulas Museu Rua do Porto, 18 9939 430 Sao Joao Lajes do Pico Acores Portugal www.whalesandsquids.org.uk	Phone: +351 292673278 / 267	dotmacclarke@yahoo.co.uk
Callan Duck	Sea Mammal Research Unit Scottish Oceans Institute University of St Andrews East Sands St Andrews Fife KY16 8LB UK	Phone: +44 334 467281 Fax: +44 1334 463443	c.duck@st-andrews.ac.uk cdd1@st-andrews.ac.uk
Steve Geelhoed	IMARES Institute for Marine Resource & Ecosystem Studies Department of Ecosystems c/o Bevesierweg 4 (Fort Harssens) NL-1781 CA Den Helder The Netherlands	Phone: +31 317482537	steve.geelhoed@wur.nl
Jan Haelters	Royal Belgian Institute of Natural Sciences 3de en 23ste Linieregimentsplein B-8400 Oostende Belgium	Phone +32(0)59700131 Fax: +32(0)59704935	j.haelters@mumm.ac.be
Ailsa Hall by correspondence	Sea Mammal Research Unit Scottish Oceans Institute University of St Andrews East Sands St Andrews Fife KY16 8LB UK		ajh7@st-andrews.ac.uk

Tero Härkönen	The Swedish Museum of Natural History Höga 160 SE-442 73 Kärna Sweden	Phone: +46 303 22 69 27 Fax: +46	tero.harkonen@nrm.se
Ilka Hasselmeier	Christian-Albrechts-Universität zu Kiel Research and Technology Centre Hafentörn 1 D-25761 Büsum Germany	Phone: +49 4834 604 103 Fax +49 4834 604 199	ih@ftz-west.uni-kiel.de
Thierry Jauniaux	Royal Belgian Institute of Natural Sciences Management Unit of the North Sea Mathematical Models (MUMM) Gulledelle 100 B-1200 Brussels Belgium	Phone: +32(0) 27732113	t.jauniaux@mumm.ac.be
Paul Jepson Invited expert	Institute of Zoology Zoological Society of London Regent's Park London NW1 4RY UK	Phone: +44 20 7449 6691	Paul.Jepson@ioz.ac.uk
Mart Jüssi	Environment Committee Parliament of Estonia Lossi pl 1A EE15165 Tallinn Estonia	Phone: +372 6316602 GSM: +372 5053732	mart.jussi@riigikogu.ee mart.jussi@gmail.com
Santiago Lens	Instituto Español de Oceanografía Centro Oceanográfico de Vigo PO Box 1 E-36200 Vigo (Pontevedra) Apdo 1552 Spain	Phone: +34 986 462946 Fax: +34 986 492111	santiago.lens@vi.ieo.es
Veronique Lesage	Fisheries and Oceans Canada Maurice Lamontagne Institute PO Box 1000 850 Route de la Mer Mont-Joli, QC G5H 3Z4 Canada		veronique.lesage@dfo-mpo.gc.ca
Mike Lonergan Invited expert	Sea Mammal Research Unit Scottish Oceans Institute University of St Andrews East Sands St Andrews Fife KY16 8LB UK	Phone: +44 334 467281 Fax: +44 1334 463443	mel5@st-andrews.ac.uk

Sinead Murphy Chair	Sea Mammal Research Unit Scottish Oceans Institute University of St Andrews East Sands St Andrews Fife KY16 8LB UK	Phone: +44 334 467281 Fax: +44 1334 463443	snm4@st-andrews.ac.uk
Kjell Tormod Nilssen	Institute of Marine Research PO Box 6404 N-9294 Tromsø Norway		kjelltn@imr.no
Graham Pierce	Instituto Español de Oceanografía Centro Oceanográfico de Vigo PO Box 1552 E-36200 Vigo (Pontevedra) Spain		g.j.pierce@abdn.ac.uk graham.pierce@vi.ieo.es
Eunice Pinn	Joint Nature Conservation Committee Inverdee House Baxter Street AB11 9QA Aberdeen Scotland UK	Phone: +44 1224 266580	eunice.pinn@jncc.gov.uk
Rui Conde de Araújo Brito Prieto da Silva	University of the Azores Cais de Santa Cruz PT-9901 862 Horta Portugal	Phone: +351 292 200 400 Fax: +351 292 200411	rui@portulano.org rprieto@uac.pt
Ricardo Serrão Santos	University of Azores (DOP) Cais de Santa Cruz PT-9901-862 Horta Azores Portugal		ricardo@uac.pt
Giuseppe Notarbartolo di Sciara Invited Expert by correspondence	Téthys Research Institute Milan City Aquarium Milan Italy AS MAILING ADDRESS PLEASE USE: Via Benedetto Marcello 43 I-20124 Milano Italy	Phone: +39 Fax: +39 02 700518468	gn@disciara.net disciara@gmail.com
Mónica Silva	University of the Azores Centro do IMAR da Universidade dos Açores Cais de Santa Cruz PT-9901 862 Horta Portugal	Phone: +351 292200442 Fax: +351 292200411	monica@uac.pt
Tero Sipilä	Metsähallitus Natural Heritage Services 57130 Savonlinna Finland	Phone: +358 20564 5912 Fax: +358 20564 59100	tero.sipila@metsa.fi

Jakob Tougaard Invited expert	University of Aarhus National Environmental Research Institute PO Box 358 4000 Roskilde Denmark	Phone: +45 4630 1956 Fax: +45	jat@dmu.dk
Raúl Castro Uranga	AZTI-Tecnalia Herrera Kaia Portualde z/g E-20110 Pasaia (Gipuzkoa) Spain		rcastro@pas.azti.es
Gordon Waring	National Marine Fisheries Service North-east Fisheries Science Center 166 Water Street Woods Hole MA 02543-1026 United States	Phone +1 508-495-2311 Fax: +1 508-495-2258	gordon.waring@noaa.gov
Stefanie Werner	Federal Environment Agency Woerlitzer Platz 1 06844 Dessau-Rosslau Germany	Phone: +49-340-2103-2221 Fax: +49-340-2104-2221	stefanie.werner@uba.de

Annex 5: Agenda

Horta, The Azores, 12–15 April 2010

Monday, 12th April 2010

08:30 Start of meeting

09:00 Visit from the Secretary of the Environment of the Azores

09.30 Plenary session, setting up of Internet connection, adoption of agenda

11:00 Coffee break

11:30 Forming of subgroups and leads, setting up of work plan

13.30 Lunch break

15:00 Presentation by Paul Jepson; Pollutant exposure in UK top predators

15.30 Work in subgroups

16.30 Coffee break

16.45 Work in subgroups

19:00 Dinner (optional) place to be announced

Tuesday, 13th April 2010

08:30 Start

10.00 Plenary session; update from leads of ToRs

11:00 Coffee break

11:30 Work in subgroups

13.30 Lunch break

15:00 Presentation by Jakob Tougaard; Effects of wind farm construction and operation on marine mammals in Danish waters

15.30 Presentation by Mike Lonergan; Target population sizes for marine mammals

the PBR approach?

16.00 Work in subgroups

16.30 Coffee break

16.45 Work in subgroups

20:00 Working group dinner

Wednesday, 14th April 2010

08:30 Start

10.00 Plenary session; review of material from ToR E, ToR C, ToR D

11:00 Coffee break

11:30 Work in subgroups

13.30 Lunch break

14.30 Tour of old whaling factory

15:00 Presentation by Ilka Hasselmeier; The German marine mammal strandings network

15.30 Presentation by Tero Harkonen; HELCOM Sea Expert Group

16.00 Work in subgroups

16.30 Coffee break

16.45 Plenary session; review print outs of available first drafts

19:00 Dinner (optional) place to be announced

Thursday, 15th April 2010

08:30 Start

10.00 Plenary session; review of material from ToR G, target population sizes and seal consumption

11:00 Coffee break

11:30 Work in subgroups; finalizing reports

12.00 Plenary session; review of material from ToR A

13.00 Lunch break

14.00 Plenary session; review of material from ToR C, and ToR B

16.30 Coffee break

16.45 Plenary session; review material from all other subgroups

19:00 Dinner (optional) place to be announced

Annex 6: WGMME Terms of Reference for the next meeting

The **Working Group on Marine Mammal Ecology** [WGMME] (Chair: Sinéad Murphy, UK) will meet in Berlin, Germany from xx March to xx March 2011 to:

a) Outline and review the potential negative impacts of tidal farms (construction and operation) on marine mammals and provide advice on research needs, monitoring and mitigation schemes;

b) The effectiveness of marine spatial planning management practices, such as Marine Protected Areas, and their role in the conservation of marine mammals;

c) Review and report on any new information on population sizes, population/stock structure and management frameworks for marine mammals;

d) Update on development of database for seals, status of intersessional work.

WGMME will report to the attention of the Advisory Committee (ACOM).

Supporting Information

Priority:	
Scientific justification and relation to action plan:	
Resource requirements:	No specific requirements beyond the needs of members to prepare for, and participate in, the meeting.
Participants:	The Group is normally attended by some 20–25 members and guests.
Secretariat facilities:	None.
Financial:	No financial implications.
Linkages to advisory committees:	WGMME reports to ACOM
Linkages to other committees or groups:	
Linkages to other organizations:	

Annex 7: Recommendations

Recommendation	For follow up by:
1. With regard to wind farm developments, establishment of means for efficient dissemination of results of common interest and means of making previous EIA reports and previously collected baseline data available for subsequent studies and assessments.	OSPAR, EC, ICES, and respective countries
2. Encourage multinational studies and encourage management decisions regarding offshore wind farms to be based on appropriate populations and/or management units for the relevant marine mammal species, irrespective of national borders.	OSPAR, EC, ICES
3. As the development of offshore wind farms extends further offshore and into new waters, monitoring should be extended to include all commonly occurring marine mammal species and marine mammal species of particular concern.	OSPAR, EC, ICES, and respective countries
4. Geographical location of offshore wind farms should consider the distribution of marine mammals throughout the year, time of day and under typical weather and hydrographical conditions.	OSPAR, EC, ICES, and respective countries
5. Increase efforts to develop common measurement standards for both noise and marine mammal abundance.	OSPAR, EC, ICES
6. Increase the effort to characterize sources of underwater noise related to the construction and operation of offshore wind farms. As part of this, common standards for measurement and characterization of underwater noise should be developed.	OSPAR, EC, ICES
7. Develop methods to assess cumulative effects on marine mammals of the underwater noise level caused by the simultaneous construction and operation at nearby sites.	OSPAR, EC, ICES
8. Step up research on the behaviour of marine mammals as a consequence of increased underwater noise levels, in particular on how changes ultimately affect population parameters.	OSPAR, EC, ICES
9. Increase efforts to characterize fundamental properties of the auditory system of marine mammals and the way noise affects physiology and behaviour.	OSPAR, EC, ICES
10. With regard to marine mammals to work towards common accepted tolerance limits for acute noise exposure and the development of common guidelines for mitigation in relation to pile driving.	OSPAR, EC, ICES
11. To undertake studies to develop better marine mammal acoustic deterrent devices, including realistic trials in the field to demonstrate their effectiveness.	OSPAR, EC, ICES
12. Attention should be given to improve efficient means of real-time detection of marine mammals during pile driving operations.	OSPAR, EC, ICES
13. Undertake other measures to prevent the exposure of marine mammals to high levels of underwater noise. This includes limiting the radiated energy during pile driving and the development of alternative methods for installation.	OSPAR, EC, ICES
14. Research should be continued and expanded to assess trends in contaminant exposure (PCBs and newer contaminants), population structure and health and reproductive effects in marine mammal species of highest risk (e.g. killer whales, St Lawrence belugas, polar bears, bottlenose dolphins, and Baltic marine mammals). The use of biopsy techniques would allow for simultaneous sampling for genetics and contaminant exposure.	EC, ICES, OSPAR
15. In order to better detect future contaminant-related population level effects, there is a need for more robust population estimates for some marine mammal populations with low abundance and high pollutant (esp. PCB) exposure (e.g. killer whales and bottlenose dolphins).	EC, ICES

16. Adoption of an adaptive monitoring and surveillance framework for marine mammals under which objectives, monitoring (inlcuding surveys, strandings and observer bycatch programmes) and outcomes are regularly reviewed and updated by a Steering Group composed of representatives from all relevant bodies. While adaptive monitoring has the advantage that the monitoring programme can respond to changing requirements and constraints, the value of consistently collected long-term datasets should be taken into account. Further, this approach will improve the mechanisms for translating monitoring findings into appropriate management action for marine mammals.	EC, ICES
17. Adoption of a coordinated international approach to developing a single assessment for each marine mammal species at an appropriate biological scale when such assessments are required (e.g. the FCS reporting at 6 yearly intervals).	EC, ICES
18. To further facilitate international coordination of monitoring, we recommend creation of ICES area/Europe-wide networks (e.g. for strandings, sightings, bycatch monitoring) and common databases and sample banks such as the European Marine Mammal Tissue Bank, and under which the unit of monitoring will be the natural population or (minimally) broad-scale spatial divisions that take into account the transboundary nature of most marine mammal populations (see also Recommendations 26–27 below).	EC, ICES
19. The WGMME again strongly recommends immediate action by the Spanish and Portuguese governments in monitoring and conserving the Iberian harbour porpoise population.	Spanish and Portugusese Governments
20. Based on the newly described harbour porpoise Management Units, the WGMME recommends to ASCOBANS the establishment of a separate conservation plan for the harbour porpoise Inner Danish Waters MU.	ASCOBANS
21. The WGMME also recommends to ASCOBANS to take into account the existence of the two newly designated harbour porpoise Management Units in the North Sea, Northeastern North Sea & Skagerrak and Southwestern North Sea & Eastern Channel, within their harbour porpoise North Sea conservation plan; with the inclusion of the Shetland Islands, Skagerrak and northern Kattegat within the Northeastern North Sea MU.	ASCOBANS
22. The WGMME agrees with the actions of the Finnish Government, and recommends a ban on fishing within the area 15.4–30.6 in Lake Saimaa from mid April to mid June.	ICES, Finnish Government
23. ICES to encourage a move away from implicit and automated conservation targets for marine mammals and towards the explicit definition and justification of target population sizes and management objectives.	ICES
24. Conduct systematic cetacean surveys of the Azores archipelago every 3 to 5 years – implement survey design established in Faustino *et al*.	Regional Government of the Azores
25. Existing Azorean observer bycatch monitoring programmes should be expanded to increase observation effort of some Azorean fisheries (e.g. demersal) and allow monitoring of other fisheries (e.g. the swordfish fishery).Further, bycatch monitoring of the European deep-water longline fleet that fish in the outer 100 nm of the Azores archipelago needs to be implemented	EC, ICES, SGBYC, Regional Government of the Azores
26. Ensure compliance of existing whale-watching regulations in the Azores archipelago through the establishment of an efficient law-enforcement scheme	Regional Government of the Azores
27. Develop and promote the European Marine Mammal Tissue Bank and recognize its relevance in different fora.	EC, ICES

28. As part of the European Marine Mammal Tissue Bank; (a) identify laboratories and institutions involved in the post-mortem investigation (full necropsy and tissue sampling) of marine mammals in the North-east Atlantic, (b) collate information on the availability and location of samples, (c) develop bilateral collaborations between laboratories and institutes to fulfil the objectives of a tissue bank, including the establishment of a steering committee to manage sample loans and data exchanges, and (d) develop a website and meta-database for the EMMTB, with links to national websites and databases.	EC, ICES

Annex 8: Technical minutes from the Protected Species and Mammals Review Group

- RGPROT/MAM
- By correspondence, 10 May 2010
- Participants: Nicole LeBoeuf (USA, Chair), Henrik Skov (Denmark), Paul Thompson (UK), Mette Bertelsen and Michala Ovens (ICES Secretariat)
- Working Group: WGMME

Protected Species and Mammals Review Group (RGPROT/MAM) dealing with EC request on 'Status of small cetaceans in European waters'.

Review of

- Section 7.1 of ICES Report of the Working Group on Marine Mammal Ecology (WGMME) 2010
- Section 1.1.1 of ICES Report of the WGMME 2009

The WGMME was asked to review and report on any new information on population sizes, population/stock structure and management frameworks for marine mammals: European Commission request on EC Regulation 812/2004. The Protected Species and Mammals Review Group (RGPROT/MAM) was asked to review Section 7.1 of ICES Report of the WGMME 2010 and Section 1.1.1 of ICES Report of the WGMME 2009. Two Reviewers offered comments on the two relevant documents.

The Review seems extensive and thorough, doing an excellent job highlighting the paucity of information up-to-date available on the distribution and abundance of comment on striped dolphins within European waters. Still, the Reviewers noted two areas of concern regarding the scope of the review which may have limited the work of the WGMME, as well as the potential capacity of that work to contribute to the overall objective of EC Regulation 812/2004 to provide protections for cetaceans within European waters.

First, the reviewers noted that the WGMME chose to interpret the European Commission request to imply that the Commission request was seeking information on whether measures taken by the EC Regulation 812/2004 have had a noticeable effect and whether there are any areas in particular where further consideration of the measures is required. While the Reviewers believe that this is a reasonable interpretation of the Commission's request, without further guidance that this was the intent of the Commission, the Reviewers wonder if this interpretation may have limited the WGMME's inclination to offer guidance to the Commission regarding where additional research and monitoring efforts may be placed by the Commission as a way to increase the effectiveness of Regulation 812/2004.

In particular, Council Directive 92/43/EEC (cited within Regulation 812/2004 as a reason for the Regulation's adoption) requires "Member States to undertake surveillance of the conservation status of [cetaceans]...Member States should also establish a system to monitor the incidental capture and killing of these species, to take further research and conservation measures as required to ensure that incidental capture or killing does not have a significant impact on the species concerned." The lack of up-to-date assessments noted by the WGMME makes this an even more important consideration. With Member States charged with surveying the conservation of these species, Member States would likely benefit from any guidance ICES could provide on where they could prioritize their efforts. So, while the WGMME strictly followed the letter of the EC's request with regard to providing "any new information on population sizes, population/stock structure and management frameworks for marine mammals", an opportunity for facilitating more comprehensive and strategic basis for implementation of Regulation 812/2004 may have been missed.

Second, the Reviewers noted that the WGMME focused its review only on the two primary target species while the Regulation covers all cetaceans. The WGMME chose to prioritize common and striped dolphins with regard to their review of the Mediterranean Sea and the harbour porpoise and common dolphin for their review of the Northeast Atlantic. The WGMME chose these species because they are the main species bycaught and found stranded with evidence of bycatch within these regions. While the Reviewers recognize the utility of this approach, Regulation 812/2004 applies to all cetaceans within European waters. By not reviewing other species and/or noting a lack of available information on these species, the WGMME may have disregarded some new information on bycatches of rarer species in the two regions and may have sent a message to the Commission that a broader assessment of cetacean species and cetacean bycatch in European waters is not needed.

For example, in the Mediterranean Sea the driftnet fishery based in Morocco, which is suspected to be responsible for the largest number of bycatches of dolphins, entails the bycatch of a diversity of cetacean species (including minke whale, fin whale, sperm whale, pilot whale and bottlenose dolphin). Yet, the sampling campaigns from this fishery mainly report on the capture of striped and short-beaked common dolphins. Given the gear type involved, this likely underrepresents the full complement of cetacean species impacted by this fishery. Without more comprehensive assessment of all cetacean species and monitoring of their bycatch in this and other fisheries in European waters, the Commission will be unable to determine whether Regulation 812/2004 is being complied with or, if it is, whether its provisions prove effective over time for all cetaceans.

With respect to the concerns expressed regarding the scope of the WGMME Review, the Reviewers point out that both of the choices made by the WGMME do not diminish the quality of the work that was conducted. Still, such choices may have limited the usefulness of the work by providing an incomplete assessment of the populations impacted by implementation of Regulation 812/2004. Indeed, the WGMME correctly notes that "the nature of population trends that would have occurred in the absence of the regulation is clearly unknown." To make this statement, however, fails to recognize the entire assemblage of species that may be interacting with these fisheries. It also does not take these impacts into account with the limited reported implementation of Regulation 812/2004 found within the Report of the SGBYC. These two considerations together emphasize the importance of communicating the need not only for better assessment of all cetaceans in European waters, but also for improved monitoring of the impacts of fisheries on their populations.